ASSESSMENT OF CARBONACEOUS $PM_{2.5}$ FOR NEW YORK AND THE REGION

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ASSESSMENT OF CARBONACEOUS $PM_{2.5}$ FOR NEW YORK AND THE REGION

Final Report

Volume II: Technical Report

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ABSTRACT AND KEYWORDS

This report, containing two volumes, presents an assessment of the carbonaceous fraction of ambient fine particulate matter (PM2.5) in New York State. Volume I includes an executive summary and a synthesis of major findings, while Volume II contains a three-chapter comprehensive report and seven related appendices. Carbonaceous components comprise a significant fraction of ambient levels of fine particulate matter (PM2.5) in many areas in the Northeast, and they may play a critical role in observed adverse human health effects associated with PM2.5 exposure. In order to design future policies based on sound scientific and technical knowledge that will reduce carbonaceous PM2.5 levels in New York, this project synthesized available and emerging information in three key areas: (1) atmospheric science and measurement methods, (2) human health effects, and (3) control technologies and strategies.

Keywords: aerosols, air quality management, atmospheric processes, control strategies, control technologies, elemental carbon, emissions, exposure, fine particulate matter, health effects, organic carbon

PREFACE

The New York State Energy Research and Development Authority is pleased to publish this “Assessment of Carbonaceous PM2.5 for New York and the Region, Volumes I and II.” The report was prepared by the Northeast States for Coordinated Air Use Management (NESCAUM), principal investigators Philip Johnson and John Graham.

This assessment focuses on a significant fraction of the ambient PM in New York State and synthesizes information in three key areas: (1) atmospheric science and emissions sources, (2) human health effects, and (3) control technologies and strategies. Volume I includes an executive summary and synthesis of major findings. Volume II contains a comprehensive technical assessment report.

This work was funded by the **New York Energy \$mart**^{SM} Environmental Monitoring, Evaluation, and Protection (EMEP) Program. This study is one of a broader portfolio of research projects characterizing particulate matter (PM), performing source apportionment on PM datasets, and addressing policy-relevant questions for PM control strategies in New York State.

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INTRODUCTION

Carbonaceous aerosols represent perhaps the most complex class of fine atmospheric constituents of particulate matter (PM). Unlike other major $PM_{2.5}$ species, like sulfates or nitrates, the organic fraction of carbonaceous aerosols consists of hundreds of different molecular species possessing a wide range of chemical and thermodynamic properties. Several logical divisions exist to characterize these aerosols, including elemental/organic, primary/secondary, and anthropogenic/biogenic. These simplifications, however, do not always result in clear divisions.

In this work, primary particles refer to the direct emissions of particles or vapors that quickly condense, while secondary particles are those formed in the atmosphere from reacted organic gases or heterogeneous processes or multiphase interactions. Elemental carbon (EC), sometimes referred to as black carbon (BC) or graphitic carbon, forms from combustion processes as a primary emission. Organic carbon (OC) can be emitted directly or formed *in situ* in the atmosphere. A clear-cut separation between these two carbon types does not exist, although generally speaking EC represents the more heat-resistant, stable carbonaceous particles. Even distinctions between anthropogenic and biogenic sources can be problematic. For example, combustion of wood could be considered as either depending on its genesis, with residential wood use being anthropogenic but forest fires biogenic. Mobile sources contribute significantly to primary and secondary anthropogenic carbon, whereas secondary organic production from terpenes and sesquiterpenes represents a dominant biogenic source for carbon aerosols.

To begin, the chapter reviews the current state of knowledge for major formation processes for carbonaceous aerosols. First, the phase dynamics and major chemical pathways are reviewed, as these provide the foundation for understanding their contribution to observed $PM_{2.5}$ levels. The visibility and climate implications are also discussed. Next, the chemical transport parameterizations of important carbon processes are outlined along with the application of receptor modeling techniques. The primary measurement approaches are examined. Finally, characterization of major carbonaceous emission sources is presented.

The second half of this atmospheric science chapter covers data analyses specific to New York State (NYS) and relevant research findings from recent monitoring campaigns. The results show carbonaceous aerosols represent a substantial fraction of the $PM_{2.5}$ mass, with a large contribution from sources located within the state. Both temporal and spatial variations are investigated. Based on the state of the science and current understanding of the role of $PM_{2.5}$ carbon in New York State, the report recommends avenues for further study.

ATMOSPHERIC CHEMISTRY AND AEROSOL PROCESSES

In the 1980s and early 1990s, atmospheric chemists focused on photochemical smog in urban and regional atmospheres, building knowledge of ozone chemistry and the complex interactions of volatile organic carbon (VOCs) and nitrogen oxides (NO_x) (Atkinson 2000). Additional concern about acid rain prompted an effort to establish the scientific underpinnings of inorganic (sulfate and nitrate) aerosol formation. Research focus transitioned in the mid-1990s to organic aerosols as it became clear that a knowledge gap existed in understanding organic aerosol sources and their formation mechanisms.

Organic aerosols comprise a substantial fraction of the fine particle burden in the atmosphere. Successful attainment of the fine particle National Ambient Air Quality Standard (NAAQS), particularly in urban areas, will depend on an increased understanding of organic aerosol behaviors. The current knowledge deficit directly results from the complex array of precursor VOCs and their varied atmospheric chemistry. These same complexities exist for smog chemistry, where VOCs and NO_x react in the presence of sunlight to form ozone. Additional complications arise in understanding organic aerosols since the reactions leading to their formation produce a diverse array of individual molecular species that are often analytically difficult to identify and quantify. Beyond the complicated chemistry, phase-dynamics dictate the extent to which organic reaction products partition into the aerosol phase. Theories for this gas-particle partitioning and secondary organic aerosol (SOA) yield descriptions put forth in the mid-1990s have provided the foundation for research over the last decade (Pankow 1994a, b; Odum et al. 1996; Bowman et al. 1997). The next two sections outline these approaches and highlight research that further develops the understanding of the complex dynamics at play.

Gas-Particle Partitioning

The complex oxidation chemistry and diverse collection of organic precursor compounds provide a wide variety of semivolatile organics (SVOCs) in the atmosphere. Nearly all major organic compound classes (e.g., alkanes, PAHs, acids, esters) contain SVOCs that can exist in both the gas and particle phase, depending on the state of the atmosphere. The associated vapor pressures of these compounds span orders of magnitude at ambient temperatures; they depend in large part on molecular size and compound class, with larger molecules or those with more functional groups having lower vapor pressures (Seinfeld and Pankow 2003). Assessment of the contribution of organic aerosols to fine particle levels requires a detailed understanding of the factors that govern their partitioning behavior.

This section reviews the development and current understanding of gas-particle partitioning of SVOCs. Both adsorptive and absorptive processes are important and depend on a number of factors, including temperature, vapor pressure, and solute-solvent interactions, where the solvent is either an aqueous or organic phase.

Over the last decade, gas-particle partitioning research has progressed substantially. Prior to work by Pankow (1994a, b), the primary mechanism for particle formation from gas phase organics (be they primary or secondary in nature) was thought to be adsorptive processes. Adsorption refers to uptake of the gas phase to a solid surface, in this case, an existing aerosol particle. The absorption process recognizes the existence of liquid layers of aerosols into which gases may dissolve. Compounds were thought to partition to the particle phase only after their atmospheric concentrations had reached saturation vapor pressure (Gelencser 2004). Pankow (1994a, b) theorized the importance of absorptive processes and laid the foundation for the current understanding of gas-particle partitioning of organic aerosols. Experiments have confirmed the importance of absorptive processes of organic aerosols, even at very low (2%) organic contributions to particulate levels (Mader and Pankow 2002). Although the process applies equally to primary and secondary species, much of the discussion in the literature is focused on secondary organics.

The basic equilibrium equation used to discuss partitioning is:

$$K_p = \frac{P}{G}$$

K_p is the partitioning coefficient, where P is OC mass of the SVOC (e.g. particle phase) divided by the SVOC gas phase (G) concentration in mass units. (Literature also uses F/A, because particles are captured on a filter (F) and the gas phase on an adsorbent (A). Often the results are normalized by the organic carbon mass fraction or total particulate for absorption and adsorption, respectively). The partitioning constant can be expanded into two terms, one each for adsorptive and absorptive processes, as derived by Pankow (1994a, b).

The derivation shows several important factors that determine the partitioning behavior. The compound-specific vapor pressure influences partitioning behavior the most, with both adsorption and absorption terms inversely proportional to it. Adsorption is also proportional to the surface area and density of active sites on the adsorbing media. Absorption, on the other hand, is inversely related to the activity coefficient, which is a measure of how the actual behavior of the compound deviates from the ideal behavior, and the average molecular weight of the absorbing organic layer. Both of these properties vary as the composition of the organic absorbing layer changes, implying the gas-particle equilibrium for each compound constantly adjusts.

Although K_p varies, experimental calculations are often conducted at constant temperature and composition. Such evaluations limit the utility of those K_p determinations for real-world situations. To better understand the dependencies of partitioning on temperature and composition, researchers isolate the major variables to assess their significance. The vapor pressure temperature effects appear to be most important (Chandramouli et al. 2003), since these span orders of magnitude depending on the specific compound. For comparison, the activity coefficient ranges in value from 0.3 to 3 for most organic species of interest (Pankow et al. 2001).

Sheehan and Bowman (2001) studied the temperature dependence of K_p for both aromatics and α-pinene, noting the exponential relationship of vapor pressure and temperature through the Clausius-Clapeyron equation. The enthalpy of vaporization (H_v) represents another important parameter influencing gas-particle partitioning, with higher enthalpies increasing the temperature sensitivity of partitioning. The results show that a 10 °C temperature decrease can lead to a 150% increase in aerosol yields, depending on the choice of H_v. Therefore, ambient diurnal temperature variations will directly impact the gas-particle partitioning along with the levels of total SVOC. At fixed levels of SVOC, the partitioning will be driven by variations in temperature with lower nighttime temperatures driving the equilibrium toward the condensed phase. Similar temperature-based partitioning differences occur on a seasonal basis.

Recently, estimation methods for calculating compound-specific vapor pressures and activity coefficients have been reviewed (Bowman and Melton 2004; Camredon and Aumont 2006). Predictive models for both parameters are generally found to agree within a factor of three when compared directly with measured values. One approach used to estimate both vapor pressures and activity coefficients for organic compounds is the Universal Quasi-Chemical Functional Group Activity Coefficients (UNIFAC) method, which relies on structural subgroups (e.g., CH_2, OH, CHO) within molecules and the interactions among subgroups to calculate the desired property (Asher and Pankow 2006). Methods such as this, however, require substantial computational resources if they are to be implemented within atmospheric models (Bowman and Melton 2004).

Given the obstacles for directly calculating the very low vapor pressures and activity coefficients, an alternative approach has been suggested by Finizio et al. (1997). They derive the relationship between K_p and K_{OA}, the octanol-air partition coefficient, which defines the partitioning of a SVOC between liquid octanol and air. The relationship established replaces the vapor pressure term and relies on a ratio of activity coefficients, with the advantage that the ratio of octanol to different SVOC activities varies less than direct comparison of individual SVOC activities (Finlayson and Pitts 1998). Nonetheless, substantial errors in partitioning behavior for polar compounds may arise when using the K_{OA} approach (Chandramouli et al. 2003).

Additional complexity arises when the role of water vapor is considered. The previous discussion assumed absorptive processes between SVOC and a distinct organic phase as much of the experimental work has focused on dry aerosol. Partitioning behavior, however, will also be influenced by dissolution into the aqueous phase and its associated inorganic ion concentrations (Turpin et al. 2000; Cocker et al. 2001a, Pankow 2003). In many cases, there may be multiple liquid phases in ambient PM, with one primarily organic (nonpolar) in nature and the other aqueous (polar), leading to multiple interfaces for mass transfer (Pankow 2003; Erdakos and Pankow 2004). The overall effect of aqueous interactions with SVOCs depends upon the specific circumstances.

Seinfeld et al. (2001) investigated theoretical dependence of K_p on relative humidity and concluded for certain monoterpene and cyclohexene oxidation products, increased relative humidity would increase partitioning to the aerosol phase. Increased aerosol water content would lower the average molecular weight of the absorbing mass. The interaction of the activity coefficient and humidity revealed four possible scenarios, with hydrophilic compounds tending to show positive associations and hydrophobic ones negative dependencies between increased humidity and partitioning into PM (Seinfeld et al. 2001). Model results investigating the interactions among species lumping mechanisms, absorbing aerosol properties, and relative humidity predicted an increased aerosol yield for hydrophilic species with increased humidity and no influence on yields for hydrophobic species (Bian and Bowman 2005). Contrary results showed that overall organic aerosol yields were reduced with aqueous seed aerosol relative to dry aerosol (Cocker et al. 2001a). Further study is required to determine the influence of humidity levels on organic aerosol yield (Jacobson et al. 2000).

Research over the past fifteen years has improved the understanding of gas-particle partition for organic aerosols. However, more studies are needed to better define the properties of individual SVOCs so their partitioning behavior can be predicted.

Gas-Phase Chemistry

The use of the partitioning constant underlies the fractional aerosol yield description proposed by Odum et al. in 1996. Their expression relates the mass of aerosol created to the mass of parent VOC reacted as a simple ratio, or fractional aerosol yield. This quantity permits an easy assessment of the relative ability of different precursor species to form aerosols. The yield incorporates the partitioning coefficient to account for the reaction products in both gas and aerosol phases:

$$Y_T = \sum_{i=1}^{p} Y_i = M_o \sum_{i=1}^{p} \left(\frac{\alpha_i K_i}{1 + K_i M_o} \right)$$

where Y_T represents overall yield, Y_i is yield of an individual product, M_o is the mass of aerosol formed, α_i is the stoichiometric mass factor for product i and K_i is the partitioning coefficient of product i. The limiting behavior of Y for very low organic mass, M_o or very small K (e.g. a highly volatile species) reveals the yield to be proportional to the mass. At very large mass or K (e.g. nonvolatile species), yield depends on the stoichiometic mass factor and is independent of M_o.

Research has shown this empirical model to be a useful characterization of organic aerosol formation, where p = 2 represents the optimal number of products to describe the system (Odum et al. 1996). One product is insufficient, while more than two does not provide an improved fit. This fit determines two pairs of parameters, α and K, that describe a composite of the more and the less volatile reaction products, respectively. In reality, many more than two products will form from a parent VOC. In addition, the model describes conditions at a specific temperature, and not the temperature continuum found in the ambient atmosphere (Donahue et al. 2005). Leungsakul et al. (2005) note that in real-world situations, the fitting parameters derived for specific reaction conditions may not apply, as the actual product distribution may be different. Despite its shortcomings, this idealized scheme has been successfully used to describe the results of reaction chamber experiments.

The reaction chambers have been employed extensively in the study of VOC oxidation reactions and the subsequent formation of organic aerosols. Given the highly complex nature of ambient processes, chamber studies attempt to minimize variables to properly assess the various dependencies of aerosol formation. The considerations include reactant(s) (e.g., terpene or aromatic species of interest), oxidant(s) (e.g., O_3, OH, NO_3), environmental variables (temperature, humidity, light level), and seed aerosol. Sampling methodology and analytical methods also play a key role. More recently, flow tube reactors have been used to monitor aerosol formation processes (Jang et al. 2003; Seinfeld et al. 2003; Lee and Kamens 2005; Esteve et al. 2006).

A number of conditions impact the experimental results and the ability to extrapolate those findings to the ambient atmosphere. Additionally, numerous measurements are required to properly characterize the chemical reaction pathways and subsequent aerosol dynamics, including such things as gas and particle phase species measurements and aerosol size distributions (Cocker et al. 2001b). Important considerations include the chamber size, which should be sufficiently large (low surface area to volume ratio) to minimize wall loss; those losses should be accounted for in the determination of product yields. Particle-chamber interactions could also introduce artifacts. Second, the experiments are generally conducted at high concentrations, well above typical ambient levels, given analytical detection limitations. Often, aerosol mass yield is determined indirectly through particle volume measurements and assumed aerosol density (Presto and Donahue 2006). Many results assume a density of 1 g/cm^3 (Griffin et al. 1999; Presto et al. 2005b; Cocker et al. 2001a), with some experiments using a density as high as 1.4 g/cm^3. In some cases the densities may be determined based on the molecular composition of the aerosol (Kalberer et al. 2000; Spittler et al. in press) or through use of Aerosol Mass Spectrometry (AMS) mass and size distributions (Bahreini et al. 2005).

Chamber studies usually focus on one of the three main oxidants, OH, O_3, or NO_3. Both OH and NO_3 radicals may react with hydrocarbons through H-abstraction, with OH being an important daytime oxidant and NO_3 important only at night (Seinfeld and Pandis 1998). For alkenes, all three oxidants add to the double bond. The addition pathway dominates over H-abstraction pathways, which generally represent less that 10% of the overall reactions for alkenes (Seinfeld and Pandis 1998; Calogirou et al. 1999). For ozone-alkene reactions, the resultant Criegee biradical can either be stabilized through collision or decompose. This latter pathway is important as it also generates an OH radical (Seinfeld and Pandis 1998).

Studies try to isolate the effects of each oxidant through experimental design. Photooxidation studies often rely on OH production through photodecomposition of nitrous acid (HONO) (Ng et al. 2006). Nitrate oxidation reactions are generally conducted in the absence of light, with NO_3 radicals produced through thermal decomposition of N_2O_5 (Spittler et al. in press). Dark conditions are also used frequently for ozonolysis experiments to prevent reactions with photochemically produced OH (Presto et al. 2005a). Ozonolysis of alkenes, however, produces OH radicals, which require the use of an OH scavenger to prevent OH oxidation of the parent alkene. Iinuma et al. (2005) confirmed that OH affects aerosol yield in ozonolysis experiments, as SOA production was reduced when scavengers were employed.

Although scavengers may keep OH from reacting with the parent VOC or its primary oxidation products, they have been shown to influence aerosol yield indirectly. This effect depends on the specific scavenger used and the resultant peroxy radical balance. SOA yield was lower when cyclohexane instead of 2-butanol was used as the scavenger in ozonolysis of cyclohexene experiments (Keywood et al. 2004a). With β-pinene as the parent VOC, the opposite behavior was observed, with the cyclohexane scavenger increasing SOA yield relative to alcohol or aldehyde scavengers (Docherty and Ziemann 2003). The relative amount of hydroperoxy and alkylperoxy radicals affects the reaction pathways and product distribution (Jaoui et al. 2004), which determines the extent of gas-particle partitioning.

In addition to isolating the oxidant, most studies focus on an individual or related set of aerosol precursors. Researchers generally study a single parent hydrocarbon in each experiment to avoid the complication of sorting out contributions from different reactants. The species drawing the most attention include toluene and α-pinene, as these represent basic structures of aromatic (usually anthropogenic in origin) and terpenic (biogenic) aerosol precursors. Studies have focused on structurally similar compounds to understand how the aerosol formation potential is influenced by the structural elements of the parent VOC and its oxidation products. This knowledge, along with emissions inventory information, helps to predict which compounds will contribute most substantially to ambient SOA levels.

The size of the molecule represents the most basic piece of information that determines the relative formation potential among VOC species. Larger molecules are more likely to contribute to aerosol formation by virtue of their lower vapor pressures (Keywood et al. 2004b). Other important elements include the extent of unsaturation (e.g., number of rings, double bonds); location of double bonds (e.g., endocyclic, exocyclic, conjugated); and the location and type of substituent(s). Hoffman et al. (1997) reviewed the oxidation of biogenic hydrocarbons and determined that cyclic hydrocarbons were more likely to form SOAs than their structurally similar acyclic counterparts since the oxidation products of the acyclic compounds were often lower molecular weight compounds (higher vapor pressure). Likewise, polyunsaturated alkenes showed greater SOA production through secondary reactions of the first-generation oxidation products (Lee et al. 2006a; Ng et al. 2006).

The location of double bonds and methyl groups also influences aerosol yield. Comparisons of aerosol formation between terpene isomers with endo- and exocyclic double bonds showed higher yield for the endocyclic species (Keywood et al. 2004b). Oxidation of the exocyclic species resulted in higher molecular fragmentation than in the endocyclic compounds, which underwent ring opening. In addition, the endocyclic molecules had more polar functional groups than did their exocylic counterparts (Ng et al. 2006). The relative position of the double bond also effects SOA formation potential, as cleavage of the bond results in the loss of carbon. The vapor pressure of the oxidized molecule may be higher or lower than the parent, depending on the relative effect of mass loss through cleavage versus vapor pressure decrease due to the additional functional groups (Lee et al. 2006a). Keywood et al. (2004b) studied the effect of methyl group location on aerosol yield. The experiments showed that when the methyl group was situated at a double bond, the aerosol yield would increase. When it was not at the double bond location, the yield would decrease relative to a compound without the added methyl group. They hypothesize that the yield differences are related to polymer chemistry and the increased likelihood of enol addition reactions for the primary oxidation products.

Hydrocarbon-to-NO_x ratios constitute another factor that affects the aerosol yield of oxidation reactions (Odum et al. 1996). A constant SOA yield was shown for VOC-to-NO_x ratios above 15, but yield decreased substantially for lower ratios (Presto et al. 2005b). The results of Song et al. (2005) confirm increased SOA yield for low NO_x conditions. Contrary to conventional wisdom, even benzene oxidation can result in SOA formation under low-NO_x conditions (Martin-Reviejo and Wirtz 2005). The product distribution shifts for aromatic species depending on the NO_x level; ring retention is favored at low NO_x levels while fragmentation is the preferred pathway at high NO_x levels (Smith et al. 1999; Jang and Kamens 2001a). When NO_x levels increase relative to VOCs, peroxy radicals react with NO_2 to create peroxy nitrates. Under low NO_x conditions, formation of peroxides, hemiacetals and acids is promoted, leading to increased SOA (Presto et al. 2005b). Lim and Ziemann (2005), in contrast, revealed that for higher-carbon-number alkanes, organonitrates may contribute substantially to SOAs. Based on the available data, the SOA production potential clearly depends on specific levels of hydrocarbons, NO_x, and peroxy radicals.

Other environmental variables such as temperature and humidity play an important role in SOA formation as well. A number of experiments have shown that lower temperatures permit higher aerosol formation due to gas-particle partitioning dependence (Odum et al. 1996; Takekawa et al. 2003). The effect of water vapor on SOA formation is less clear. Some experiments show decreased aerosol yield as relative humidity (RH) increases (Na et al. 2006) while others show aerosol mass increasing as RH increases (Jonsson et al. 2006). Experiments conducted by Docherty et al. (2005) predict that RH may increase or decrease the SOA yield of β-pinene, depending on the OH scavenger used. The reaction of water with oxidation products may result in carboxylic acids prone to particle formation. Czoschke et al. (in press) point out that a negative RH dependency of SOA formation may be due to heterogeneous processes and aerosol acidity.

The significant impact of aerosol acidity on SOA formation has only recently been realized (Jang and Kamens 2001b; Jang et al. 2002). Seed aerosols were initially introduced into reaction chambers simply to facilitate particle growth by providing a substrate. In 2001, Jang and Kamens reported that acidic aerosols promoted the heterogeneous production of SOAs from aldehydes. Since that time, dozens of studies have observed similar enhancement of SOA under acidic conditions (Kroll and Seinfeld 2005; Jang et al. 2003; Iinuma et al. 2004; Jang et al. 2005). Much of the work has relied on inorganic acids, although other research reveals that weak organic acids can also raise SOA production (Kalberer et al. 2004; Gao et al. 2004).

Currently, researchers hypothesize that acid catalysis generates oligomers, which are similar in concept to polymers except they are made up of a limited number of repetitive structures (Baltensperger et al. 2005). The detection of high molecular weight species has been reported by a number of studies (Baltensperger et al. 2005; Gao et al. 2004; Tolocka et al. 2004; Kalberer et al. in press). Various formation mechanisms have been suggested to explain the presence of these oligomers, including aldol condensations and gem-diol reactions.

Recent evidence shows that small molecules such as isoprene and glyoxal may also form SOAs via these newly discovered pathways (Hastings et al. 2005; Limbeck et al. 2003; Claeys et al. 2004; Matsunaga et al. 2005). This represents an important finding since in the past isoprene was thought to be too small to form SOAs. In addition, other recent studies have shown that aqueous-phase reactions of water-soluble organics (including isoprene oxidation products such as pyruvic acid) can also lead to SOA production (Carlton et al. 2006; Lim et al. 2005). Given the substantial emissions of isoprene from the biosphere, its contribution to SOA formation could be quite large (Matsunaga et al. 2005; Kroll et al. 2005). Global modeling predicts a doubling of SOA when formation mechanisms involving isoprene are included (Henze and Seinfeld 2006). These predictions, however, may not accurately reflect ambient conditions, especially in light of observations that SOA mass decays rapidly under the low NO_x conditions encountered over most of the globe (Kroll et al. 2006).

Despite the emerging evidence of heterogeneous chemistry, a detailed understanding of the mechanisms involved is still under development. This lack of understanding prevents an overall assessment of the relative contribution of heterogeneous processes to SOA growth. In addition, the formation of second- (and later-) generation products and cross-products (i.e. oxidation by multiple oxidants) further complicates the overall understanding of SOA production. Models attempt to determine the relative importance of all these factors, but in their current form they lack the sophistication needed to fully represent SOA formation.

At this time, a number of other drawbacks exist in applying experimental results to the real world. The experiments are designed to isolate specific parameters that influence SOA formation (Cai and Griffin, 2006). Ambient conditions are never that clear-cut and are highly variable. Many of the studies are conducted using high levels of reactants to which the results are sensitive, from the standpoint of reaction pathways and partitioning dynamics. These high concentrations are needed to permit detection of the gas and particle phase species. Even so, issues with characterization and analytical method arise: complete characterization is not achieved (Smith et al. 1999; Hamilton et al. 2005), uncertainties are introduced through derivatization steps (Jaoui et al. in press; Koehler et al. 2004), calibration is uncertain due to lack of calibration standards (Jang and Kamens, 2001a; Forstner et al. 1997). For these reasons and more, additional research is needed to fill in the knowledge gaps.

Radiative Effects of Carbonaceous Aerosol

Atmospheric particles play a vital role in the energy balance of the Earth. Fine particles may scatter or absorb incoming radiation from the sun, directly affecting the Earth's radiation balance. These effects also occur in the visible frequency, leading to reduced clarity of view at the Earth's surface. Additionally, aerosols may serve as cloud condensation nuclei (CCN), which are particles that grow into cloud droplets in the presence of supersaturated water vapor (Seinfeld and Pandis, 1998). Such growth affects the properties of clouds in the sky, often referred to as the indirect effect.

Although the global climate impacts of organic aerosols extend beyond the scope of this report, some local effects may occur that influence local atmospheric organic chemistry. Given the large uncertainties, however, feedbacks to temperature, ultraviolet flux, and atmospheric stability, which are all relevant to atmospheric chemistry, cannot be reliably predicted. Nonetheless, an awareness of the major climate effects may provide another rationale for addressing the environmental impacts of carbonaceous aerosol emissions. In that light, the following section describes the substantial role carbonaceous aerosols play in the degradation of visibility in the northeastern U.S. Further, their effects on global climate are outlined and current research activities are highlighted.

Visibility The U.S. Congress recognized visibility degradation as a problem in 1977, setting a goal to reestablish pristine conditions in national parks as part of the Clean Air Act. In response more than twenty years later, the U.S. EPA promulgated the Regional Haze Rule, establishing a timeline to achieve the visibility objective by 2064 while preventing any backsliding on the current best visibility days. The northeastern states have sponsored studies of visibility conditions in the region over the last decade and are developing strategies to address the causes of poor visibility.

Visibility degradation results from scattering and absorption of visible light, primarily by small particles. An observer's view can be obscured by light being scattered from or into the sightline, or being removed by absorbing particles. Mathematically, the extent of this interference is often expressed as light extinction, with extinction being the sum of scattering and absorption by both particles and gases. Both scattering and absorption can be expressed as a product of aerosol mass and an associated mass efficiency, which allows the determination of the extinction contribution of different particle species based on mass measurements.

This approach forms the basis used in the national regional haze program to assess current visibility and apportion species contributions to degradation. Extinction properties of the major aerosol species (sulfates, nitrates, OC, EC, fine crustal material, sea salt, and coarse matter) have been determined, with recent changes being accepted by the Interagency Monitoring of Protected Visual Environments (IMPROVE) steering committee (IMPROVE, 2006). These changes incorporate dry scattering efficiencies for two fine particle size modes and, for some constituents, enhanced efficiency due to hygroscopic particle growth. Studies disagree on the role hygroscopic growth plays for organic carbon, with some research suggesting enhancement while other research shows a hydrophobic effect (IMPROVE, 2000). For U.S. EPA's haze rule, humidity effects are attributed only to sulfates, nitrates and salt.

In the northeastern Class 1 areas, sulfate aerosols dominate current poor (20% worst) visibility conditions, with an estimated contribution of nearly three-fourths (NESCAUM, 2006a). Organic carbon contributes measureably (~13%), while absorption due to elemental carbon plays a minor role (~4%). The contribution of carbonaceous aerosols to visibility extinction mirrors its relative mass contribution. Thus, the effects are more apparent in urban areas, with lesser relative impact in rural areas. On days with the best visibility, sulfates still dominate Class 1 areas, although on those days they represent half of the aerosol extinction. On the best days, the carbonaceous aerosol contribution increases to nearly 20% and 8% for OC and EC, respectively.

Climate/Cloud Condensation Nuclei The climate effects of aerosols have emerged as an area of intense research over the last two decades. Unlike greenhouse gases, whose impact on radiative transfer is fairly straightforward, the effects of aerosols are complicated by particle morphology (e.g., size, shape, and mixing state). Particles may exert positive (heating) or negative (cooling) effects, or forcing, on climate, depending on the type of particle and its atmospheric distribution. They may have direct, indirect, or "semidirect" effects. Direct effects refer to direct scattering or absorption of solar radiation by aerosols, while indirect effects are changes caused by aerosol-cloud interactions (Menon, 2004). Absorbing aerosols, such as EC (or black carbon) may also affect the radiative properties of the atmosphere through heating that leads to cloud evaporation or by changing the reflectivity of snow cover. These processes are called semidirect effects.

The composite effect of carbonaceous aerosols on climate remains uncertain (Kanakidou et al. 2005). For example, EC is considered the primary aerosol absorber in the atmosphere. However, predictions of its direct forcing depend on its mixing state (Jacobson, 2001; Bond and Bergstrom, 2006). When considered an internal mixture instead of an external mixture, its relative forcing is doubled due to an increase in cross-sectional area.[1]

Other recent evidence has emerged that indicates OC may also absorb ultraviolet radiation, particularly OC derived from biomass burning (Kirchstetter et al. 2004). The semidirect effects of heating in the atmosphere are very difficult to quantify, especially considering the lack of knowledge of the vertical distribution of the aerosols (O'Brien and Mitchell, 2003). In addition to cloud dissipation, which may increase radiation reaching the surface, heating aloft will affect atmospheric stability.

The indirect effects of carbonaceous aerosols are equally uncertain. Cruz and Pandis (1997, 1998) studied the cloud condensation nuclei (CCN) activation ability of pure organic and mixed organic-inorganic aerosols. The research showed that simple organic species readily contribute to CCN, and the particle growth can be modeled as predicted by modified Köhler Theory. Kanakidou et al. (2005) note that organic aerosols introduce competing effects for CCN activation: a reduction in surface tension potentially promotes CCN activation, while formation of organic surface films may slow growth. A recent review confirms the importance of organic aerosols as CCN, noting that bio-aerosols also contribute on a global basis (Sun and Ariya, 2006). Research continues on the activation potential of specific organic aerosols, including fatty acids (Broekhuizen et al. 2004), biogenics (VanReken et al. 2005; Huff Hartz et al. 2005), organonitrates (Twohy et al. 2005), and organic acids (Huff Hartz et al. 2006). As expected, substantial variation in CCN formation potential is observed.

Predictions of the climate forcing of carbon aerosols from global models show the warming effect of EC is offset in large part by the cooling effect of OC (Chung and Seinfeld 2002). Recent measurements, however, show that current models fail to produce the observed concentrations of organics in the free troposphere, which casts further doubt on the climate predictions of global models (Heald et al. 2005). Much fundamental work remains to model the climate impacts of carbonaceous aerosols, including improvements

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[1] Particles that are made up of a single component are considered external mixtures, whereas particles that have multiple components are considered internal mixtures. Internal mixtures may have a core of one species coated by another.

in the temporal and spatial distribution of emissions, atmospheric transformation schemes, and relevant cloud processes.

Chemical Transport and Receptor Models

Chemical transport and receptor models provide insights into the relationship between emission sources and the environment. Chemical transport models or emissions-based air quality models begin with emissions inventories and incorporate atmospheric physical and chemical process to predict downwind concentrations throughout the modeling domain. Once validated by comparison to ambient measurement data, these models can be used for apportioning ambient pollutant concentrations to specific sources, as well as for evaluating potential effects of future changes in emissions or meteorology. In contrast, receptor-based models begin with ambient measurement data at one or more receptor locations and work backward to identify sources contributing to historical ambient pollutant concentrations at the receptor locations. Receptor models cannot predict effects of future emissions changes. They can, however, be applied to long historical records, providing an indication of past source-receptor relationships that can be used to evaluate the effects of historical emissions changes. That analysis provides a valuable complement to emission-based models for determining effective future emissions control strategies.

The following discussion provides an overview of the current modeling approaches used in the air quality community for assessment purposes. The available models are described along with advantages and disadvantages of their applications. Specific examples from the literature demonstrate their use and the character of the results.

Chemical Transport Models Atmospheric chemical transport models (CTMs) exist in many different formulations. These have evolved from simple parameterizations to more complete representations of atmospheric processes as both the understanding of those processes and computational capabilities of computers have expanded. The most sophisticated models couple spatially and temporally refined emissions, meteorology, and chemistry to predict the evolution of atmospheric constituents throughout the model domain. The treatment of organic aerosols by transport models represents a small component of the whole that has only recently grown in its sophistication. Even so, substantial uncertainties remain given the complexity of organic aerosol formation and the difficulty in characterizing their thermodynamic properties.

Early models of organic particulates treated only primary emissions. Secondary organics, if included, were scaled directly to primary emissions, ignoring chemistry completely. The next generation of models followed the lumped SOA yield methodology developed by Pandis et al. (1992), which relied on fractional aerosol yields of specific VOCs. The chemical mechanism included VOC oxidation that led directly to condensable organic aerosols, where the saturation vapor pressure was zero or some nominally small value.

Although this approach does rely on oxidant level and parent VOC reactivity, it does not incorporate temperature, humidity or existing particle composition and level. Studies have noted the model dependence of SOA production on existing PM, and the possible enhancement of SOA production through interactions of biogenic and anthropogenic aerosols (Kanakidou et al. 2001; Schell et al. 2001). Further developments beyond a vapor saturation approach include treatments of adsorption, absorption/dissolution to organic or aqueous media and detailed reaction mechanisms (Seigneur 2001; Pun et al. 2002a; Dechapanya et al. 2003a).

The more sophisticated approaches to gas-particle partitioning rely on gas-phase chemical mechanisms to describe the chemistry that creates SVOCs in the model. A number of oxidant mechanisms currently available were developed initially to describe the chemistry of ozone formation. These mechanisms generally follow one of two primary formulations: lumped structure or lumped molecule (Jiminez et al., 2003). The most common lumped structure mechanism is called the Carbon Bond (CB) Mechanism. Examples of the lumped molecule approach include the Statewide Air Pollution Research Center (SAPRC) and the Regional Atmospheric Chemistry Mechanism (RACM). Surrogate organic species tracked in the carbon bond system include paraffin (PAR), olefin (OLE), aldehydes (ALD2), and aromatics (XYL), where individual molecules are broken into representative bonds (Dodge, 2000). Important major species such as isoprene, toluene, and ethene are treated explicitly. SAPRC and similar mechanisms group species based on expected reactivity and structural similarity and generally model more species, along with their specific chemical mechanisms, than the CB methodology.

Since the gas-phase chemistry was optimized for ozone modeling, additional pathways specific to organic aerosol production must be incorporated. Numerous investigators have tackled this task over the last several years and continue to make improvements (Barthelmie and Pryor 1999; Dechapanya et al. 2003a, b; Griffin et al. 2002; Pun et al, 2002a). Barthelmie and Pryor (1999) present one of the earliest attempts to incorporate chemical mechanisms based on smog chamber experiments of biogenics, including the oxidation of primary reaction products. Highlights focused on uncertainty in reaction pathways and rate constants, along with partitioning parameters. Similarly, Dechapanya et al. (2003a,b) implemented reaction kinetics and phase partitioning for both individual and lumped aromatics into an air quality model. They describe a lumping mechanism for both reactants and products based on the relative reactivity of those individual species. The approach tries to capture the complexity of SVOC formation while reducing the number of species tracked by the model.

Researchers built SOA functionality into the SAPRC mechanism to create the Caltech Atmospheric Chemistry Mechanism (CACM) (Griffin et al. 2002a, b, Pun et al. 2002a). The new model expands the SAPRC ozone chemical mechanism and organic oxidation products capable of SOA formation. Akin to other approaches, the surrogate compounds contain the average characteristics of their parents and are grouped based on molecular size, structural characteristics, reactivity, and SOA formation potential (Griffin et al. 2002a). A thermodynamic module for both hydrophilic and hydrophobic species was created that interfaces with the chemical mechanism and relies on UNIFAC method to develop specific vapor pressures and activities (Pun et al. 2002a).

This model was expanded to couple the hydrophilic and hydrophobic model (Griffin et al. 2003), recognizing that some SOAs may partition to both organic and aqueous phases. It was eventually simplified to drop the distinction between hydrophilic/phobic species in the surrogate partitioning (Griffin et al. 2005). Additional refinements included updates of aromatic oxidation chemistry and adjustments of stoichiometric and reaction rate constants. Subsequent changes were implemented for SOA formation from terpenes. Model tests revealed underprediction of SOA when compared to laboratory predictions; this required adjustments to partitioning parameters (decreasing compound specific vapor pressure) to improve performance (Griffin et al. 2005; Chen and Griffin 2005; Chen et al. 2006a). The modification was justified based on recent laboratory work that indicates the formation of low vapor pressure species from heterogeneous reaction processes (Jang et al. 2002, Kalberer et al. 2004). Pun et al. (2006) pursued different revisions to the CACM methodology by reducing SOA surrogates to improve computational efficiency. Here, the vapor pressure adjustments made to match model and experimental results were justified simply on the basis of uncertainties in the original vapor pressure estimates. Similar adjustments to account for discrepancies between modeled and measured yield were employed in an application using the Master Chemical Mechanism that purports to follow oxidation from parent VOC to final CO_2 and H_2O end products (Jenkin, 2004).

Application of chemical transport models for regulatory purposes requires validation that the models reasonably portray the evolution of ambient pollutant concentrations in the atmosphere. Acceptable models may be used to evaluate the potential effects of future changes in emissions or meteorology. Many recent modeling applications have investigated the ability of the current generation of models to accurately describe the atmospheric transformations of carbonaceous aerosols (Pun et al. 2002b; Pun et al. 2003; Seigneur et al. 2003; Tsigaridis and Kanakidou 2003; Morris et al. 2006; Vutukuru et al. 2006; Russell and Allen 2005). These studies indicate further development and refinements need to be pursued to improve model performance.

In one study, the Community Multiscale Air Quality (CMAQ) model developed by the U.S. EPA in coordination with the academic community (Binkowski and Roselle 2003) was used to simulate a pollution episode in the southeastern US. Researchers took advantage of the model design, which has a modular construction for major processes, by comparing the performance of three separate SOA modules. The results revealed substantial differences in predicted OC levels, with the more sophisticated mechanistic module simulating higher aerosol concentrations than the two empirical formulations based on laboratory data. The modules did agree that biogenic sources were dominant. The researchers concluded that uncertainties associated with the temperature dependence of aerosol processes were paramount in explaining differences in the model results (Pun et al. 2003). Improved understanding of gas-phase chemical mechanisms and resultant water-soluble SOA, along with better representation within the model, is necessary to enhance model performance.

More recently, Morris et al. (2006) compared results from two different models, CMAQ and the comprehensive air quality model with extensions (CAMx), with observations in the southeastern US. Both models relied on similar meteorological and emissions inputs, but they differed in their treatment of atmospheric processes. The models showed substantial underprediction of OC, with CAMx performing slightly better, attributed to lower levels of concentrations required for condensation to occur. (The parameters used in CAMx were based on older chamber measurement data and were two orders of magnitude lower than ones used in CMAQ. This order of magnitude difference leading to improved performance is reminiscent of "adjustments" made in other models to improve performance). Modifications were made to CMAQ, including the addition of polymerization reactions resulting in nonreversible condensation in the model and the inclusion of isoprene and sesquiterpene mechanisms leading to SOA formation. These enhancements significantly improved model performance, although other changes may be necessary to capture atmospheric SOA behavior.

Modeling SOA on a global scale has also been performed (Tsigaridis and Kanakidou 2003; Chung and Seinfeld 2002). Based on current understanding of global precursor emissions and model uncertainties, global SOA production is constrained with a factor of ten; globally averaged biogenic contribution is predicted to be 95% of the total, with the low end of the range at 50% (Tsigaridis and Kanakidou, 2003). The researchers argue more can be learned through sensitivity studies confined to one model rather than comparisons among models, because variables are minimized.

The understanding of ambient organic aerosol processes has evolved considerably in the last decade. The inability to incorporate the current understanding into state of the art atmospheric models in a computationally efficient manner remains a significant challenge. Given the complexities and uncertainties, new approaches may be required. One potential direction focuses on saturation concentration as the essential modeling parameter, removing the need to track individual species concentrations in favor of volatility bins (Donahue et al. 2006). This approach would easily incorporate primary OA emissions in gas-particle transformations in addition to other processes (gas/particle/heterogeneous) that may shift the bulk vapor pressure. Regardless of new approaches or refinements of existing ones, knowledge gaps remain that may preclude the successful modeling of organic aerosols in the near future.

Current research agrees on a number of areas requiring further study to address uncertainty within atmospheric models of OC. Considerations to improve models include:

- improved estimates of gas-aerosol partitioning variables (heat of vaporization, activity coefficients, translation of laboratory results to ambient conditions)
- irreversible gas-to-particle transfers
- hydrophobic versus hydrophobilic behaviors and related oxidative transformation of SVOCs
- improved mechanistic representations for secondary and later reaction products
- more complete inclusion of precursor species (including isoprene, sesquiterpenes)
- representation of acid-catalyzed or heterogeneous processes

In addition, improvements in emission inventories are necessary. Areas for development include:

- expansion of source profiles to provide speciated VOC composition
- enhancement of temporal and spatial information of primary and precursor emissions
- accurate mapping of emitted species to modeled species (Makar et al. 2003; Vutukuru et al. 2006).

Receptor Models Multivariate mathematical models and ensemble backward trajectory techniques represent two broad categories of receptor methods. Multivariate models, such as Chemical Mass Balance (CMB), Principal Component Analysis (PCA), Positive Matrix Factorization (PMF) and UNMIX, typically rely on the variations in multiple species data in multiple observations at one or more sites. Ensemble trajectory techniques, such as Cluster Analysis, Potential Source Contribution Function (PSCF) and Residence Time Analysis (RTA) are driven by large numbers of backward air trajectory calculations at one or more sites. These are sorted and aggregated as a function of measured or modeled pollutant concentrations at the receptor, or grouped as a function of similar upwind locations.

The CMB multivariate model requires input of measured source composition profiles. Successful application of the model relies on source profiles that accurately reflect the emissions of the modeled chemical species along with uncertainty estimates. Uncertainty estimates of the ambient measurements are also needed. The model must include profiles from all sources that impact the ambient pollution levels (Cass, 1998). In its most common implementation, CMB uses all chemical species provided (not just tracers) and estimates the uncertainty of the apportionment results through weighting the relative importance of the inputs based on their reported precision (Watson et al. 2002). Other model assumptions include: (1) emission source profiles do not vary over the period of interest (and the emitted species are chemically stable), (2) modeled chemical species do not interact, (3) modeled chemical species outnumber the modeled number of source profiles, (4) source profiles are linearly independent, and (5) reported uncertainties are random and normally distributed. Assuming the underlying assumptions are met, the model determines the contribution of each source to each ambient measurement, and the identified source contributions produced are unambiguous.

Unlike the CMB model, the PCA, PMF, and UNMIX models do not need source profiles as input, as the models determine the source factors. This introduces ambiguity, as the resulting "sources" require subjective interpretation by the modelers to identify what these source influences actually represent. Resulting sources may correspond to an individual point source, source category, source region, meteorological influence, measurement or data processing artifacts or various combinations thereof. The fixed or constant chemical composition of the modeled sources presents a drawback for their application. In reality, sources will have "fixed" primary emissions that may undergo atmospheric transformations. The variable rate of secondary production violates the model condition of depending on a constant relationship among the source profile components. Consequently, models like PMF and UNMIX may divide a source influence into multiple "source components," each with constant but different chemical composition, representing different degrees of secondary aerosol formation (Poirot et al. 2001). Because of the complexities in source interpretation, and the sole reliance on the measured chemical compositions, ensemble trajectory techniques based on meteorology provide further insights for interpreting and evaluating the multivariate model results.

Schauer et al. (1996) present the first application of CMB using the organic molecular level composition of particles to apportion measured $PM_{2.5}$ to its sources. Their work evaluated measured organic species for atmospheric stability to determine which species could be used in the model. Levels of these markers in source profiles and measurements from four sites in southern California were used for source apportionment. The analysis showed four dominant sources responsible for the primary $PM_{2.5}$ OC: diesel- and gasoline-powered vehicle exhaust, meat cooking, and wood combustion. An upper bound for SOA levels was determined by subtraction (total mass of OC minus primary OC as determined by CMB). The results showed SOAs can contribute no more than 18% of the OC at the urban sites and less than 31% downwind. Primary tracers were associated with the four major contributors. Most of the EC was allocated to diesel vehicles. Gasoline vehicles were associated with most of the PAHs. Oleic acid and nonanal were primary tracers for meat cooking, while retene and acids associated with oxidation of terpenes identified wood combustion. Table X provides further selected tracer species and associated references.

This same approach has been implemented in numerous studies. Houston $PM_{2.5}$ was apportioned and showed similar sources as those in southern California, although road dust was more prominent and wood combustion less important (Fraser et al. 2003b). This study also identified a significant contribution from oil combustion related to shipping operations, but only for the measurement site situated near Houston's shipping channel. SOA estimates were not provided, but upper bounds can be derived as CMB apportioned only 53-89% of the total mass. Tunnel measurements of $PM_{2.5}$ in Houston were used to show the ability of

CMB to differentiate between diesel and gasoline vehicles; apportionments were compared to tunnel traffic volume for confirmation (Fraser et al. 2003a). The measurements indicate the importance of including the appropriate emission profiles, considering the great variation that may exist in motor vehicle fleets due to maintenance, fuel use, and operating conditions. Unlike Houston, wood smoke and diesel exhaust were identified as the dominant sources of OC in the southeastern U.S. (Zheng et al. 2002). Road dust, meat cooking, and gasoline-vehicle exhaust were identified as minor contributors. Unapportioned OC mass was shown to be greatest in the summertime, which is consistent with anticipated SOA increases in warmer months.

Table 1-1. Molecular tracers for major sources of carbon aerosol.

Tracer Compound	Major Source	Reference
Levoglucosan	Biomass combustion	Fine et al. 2001, 2002, Hays et al. 2002
Guaiacol and substituted guaiacols	Wood combustion	McDonald et al. 2000, Schauer et al. 2001, Fine et al. 2001, 2002, Hays et al. 2002
Syringols	Hardwoods	McDonald et al. 2000, Schauer et al. 2001, Fine et al. 2001, 2002, Hays et al. 2002
Resin acids	Softwoods	Rogge et al. 1998, Schauer et al. 2001, Fine et al. 2001, 2002, Hays et al. 2002
Retene (resin acid derivative)	Softwoods	McDonald et al. 2000, Fine et al. 2001, 2002
Betulin	Paper birch	Fine et al. 2001
Juvabione, Dehydrojuvabione	Balsam fir	Fine et al. 2001
Yangambin	Yellow Poplar	Fine et al. 2002
Friedelin	White Oak	Fine et al. 2004-
Hopanes & Steranes	Vehicular Exhaust	Rogge et. al. 1993b, Schauer et al. 1999b, Zielinska et al. 2004
Isoprenoids	Vehicular Exhaust	Schauer et al. 1999b, 2002b
Tricyclic Terpanes	Vehicular Exhaust	Schauer et al. 1999b, 2002b
Cholesterol	Meat cooking	Rogge et al. 1991, McDonald et al. 2003
γ-Lactones	Meat cooking (Grilling, deepfrying)	Rogge et al. 1991, Schaeur et al. 2002a, McDonald et al. 2003
nonanal	Meat cooking	Rogge et al. 1991; Schaeur et al. 1999a
High molecular weight odd carbon # n-alkanes	Leaf abrasion	Rogge et al. 1993c

Source apportionment results from CMB have been compared to source-oriented models for performance assessment. Marmur et al. (2006) demonstrate substantial differences in source apportionment, which they ascribe to the inherent biases for each approach. Receptor models may capture the temporal variability in source impact at the measurement site. The spatial representativeness of the result, however, is limited by the representativeness of the underlying measurement. The CMAQ results may better determine the spatial impact of sources over longer time periods while failing to capture their variability. Uncertainties in temporal profiles for emissions, combined with large-scale meteorological features, prevent the proper temporal characterization of variable source impacts.

Other research has found significant agreement in day-specific apportionment from receptor and source-based models (Held et al. 2005). The air quality model did, however, identify as contributors several sources whose profiles were not used in CMB. The study demonstrates the utility of multiple tools for evaluating air quality impacts and confirms the necessity of inputting all major source profiles into CMB to achieve the best apportionment. As noted by Christensen (2004), CMB analysis can be performed when unknown sources of pollution impact a measurement. If some profiles are unavailable, an additional term can be added to CMB that may account for the unknown source(s). Otherwise, the model may mistakenly assign pollution to the available profiles.

The inability of receptor models to treat secondary aerosols directly represents another weakness. Robinson et al. (2006) point out that assumptions about molecular stability may affect CMB results. Their research indicates that during peak atmospheric oxidative periods, substantial apportionment errors can be introduced since the underlying assumptions of stable markers (e.g., hopanes) in source profiles may be violated. The model cannot account for variations in the ratios of species in source profiles from source to receptor. Adjustments to the model that would allow the atmospheric transformations may be possible. Source emission aging can be treated by adjusting the original source profile by "fractionation factors" that account for such effects as differential deposition and chemical transformation (Sattler and Liljestrand, 2005). Multiple source profiles can be used as inputs, with the original profile indicative of local or recent emissions and modified profiles for aged or transported emissions.

Factor models like PMF and UNMIX avoid problems relating to missing source profiles, as these tools derive factors meant to represent sources. Interpretation of the derived factors represents the biggest challenge of these models. Any number of variables can be used as input so long as uncertainty estimates are provided. Often individual species are used as indicators for contributions from certain sources (e.g., Ni and V for oil combustion, K for wood combustion). Temporal differences in factor strength, such as weekend-weekday or seasonal behavior, can be used to assist in identify the underlying source(s). Additionally, tools such as back trajectories or conditional probability function, which employs local wind data, point to possible region-specific source influence. Unlike CMB, which can be used to apportion a limited number of samples, PMF and UNMIX depend on variability inherent in hundreds of samples for their resolution power.

An application of UNMIX in Baltimore determined that six sources contributed to measured $PM_{2.5}$ (Chen et al. 2002). Traffic-related pollution was attributed to two factors that varied in composition and temporal character. One factor was associated with reasonably high concentrations of nitrate and peaked in the wintertime, while the other factor was dominated by OC and EC. Carbon monoxide was strongly associated with both factors, confirming their local combustion origin. Another factor that peaked in the wintertime also had high levels of K and OC, with some EC as well. Back trajectories calculated on days with high loading from this factor showed a source region consistent with an area known for residential wood combustion, corroborating the contributions from K and OC.

A number of recent efforts have relied on the OC- and EC-resolved fractions to improve model results, focused especially on differentiating diesel and spark-ignition factors (Kim and Hopke 2004a, b; Kim et al. 2004; Lee et al. 2006b). Researchers rely on differences in weekday and weekend behavior of profiles to support the assignment of diesel truck traffic and gasoline vehicle sources to their correct profile. A profile that shows differences between weekday and weekend levels are anticipated for diesel vehicles, while no substantial variation is expected for gasoline traffic. Other carbon-dominated factors were attributed to wood smoke, railroads, buses, and oil combustion. Wind directional analysis and elemental tracers corroborated these classifications. In most cases, an unattributable factor of sulfate and pyrolytic OC was derived. Researchers speculate this could be explained by heterogeneous formation of SOA via acid catalysis (Lee et al. 2006; Kim and Hopke 2004a).

These recent studies using temperature fractions of OC and EC reveal the complexity of interpreting factors derived by receptor models. Comparisons among diesel and gasoline factors across sites (e.g., Atlanta, St. Louis, Washington, D.C., and Brigantine, New Jersey) show considerable differences in the relative proportions of the various fractions at each site, although a consistent general character exists for high OC associated with gasoline and high EC with diesel. This same general behavior is seen for actual source profiles from the early 1990s (Watson et al. 1994; Lee et al. 2006b). The relative differences in contributions from the various fractions could be due to a number of reasons. The source profiles represent fresh emissions, while the ambient factors are mixtures of fresh and aged emissions. In addition, the ambient measurements may have other source contributions that co-vary, resulting in subtle shifts in proportionality among OC and EC fractional composition. Evidence supports the potential for misclassification or changes in C evolution time as a result of intermixtures of sources, filter loading difference, or oxidizing agents in the collected $PM_{2.5}$, as discussed below in the analytical techniques section. Simultaneous source testing may provide further insights to explain the observed variations in PMF results.

More sophisticated versions of PMF have been developed to accommodate the inclusion of factors beyond traditional pollution data, including meteorological variables (Paatero et al. 2003) and particle size information (Zhou et al. 2004b; Zhou et al. 2005; Larson et al. 2006). Another innovation permits the inclusion of data obtained on different timescales (e.g. 10-minute average, hour-long, and 24-hour) (Zhou et al. 2004a; Ogulei et al. 2005). Larson et al. (2006) present PMF results that derive three mobile-source related factors using size distribution, whereas the analysis without size information obtained only two factors. The previous diesel factor is split into two different factors. The first has less mass with a dominant small particle mode, which represents fresh emissions. The second profile more strongly resembles the original profile, with most of the mass and a larger size profile, indicative of aged particles.

SAMPLING AND ANALYTICAL METHODS

Recent advances in monitoring methods are rapidly expanding knowledge of carbonaceous aerosols. Much of the current understanding derives from bulk carbon measurements that lack the detail required to fully characterize the organic components. Traditional filter-based methods rely on time-integrated samples that provide limited temporal information. With great care and effort, detailed speciation analysis of these integrated filters can be completed. New in situ and on-line analytical approaches, however, provide an increased level of temporal and compositional detail. Further development and deployment of these new techniques will expand understanding of carbonaceous $PM_{2.5}$.

This section reviews the major sampling and analytical approaches used for carbonaceous aerosols. The benefits and drawbacks of different techniques are highlighted. No one approach yields all relevant information required to describe carbonaceous aerosols. Analyses can be limited to mass determination, or as specific as molecular scale composition. They may indicate limited source information directly, as in isotopic analysis, or provide inputs for source apportionment modeling. Combinations of various approaches are needed for the most complete description of the levels and properties of ambient carbonaceous $PM_{2.5}$.

Filter-based Sampling

Much of the existing knowledge base of carbonaceous aerosols relies on time-integrated filter collection of fine particles. Researchers have identified a number of key elements important to optimal aerosol characterization. These parameters include filter media, sample flow rate, and sampler configuration, which may include multiple in-line filters or gas-phase denuders or post-filter adsorbents. These sampling choices affect the extent to which measurements suffer from positive and negative artifacts.

Positive artifacts, for example, may occur on quartz filters, as they have a high surface area prone to adsorption of gas-phase species. The adsorptive artifact has an upper limit based on the saturation level of the filter and therefore is most problematic for short sampling times and at low ambient particle concentrations (Gelencser 2004). Adsorption also depends on face velocity, or the sampling flow rate divided by the exposed filter area. Increased face velocity lowers the artifact (McDow and Huntzinger 1990), indicating an increase in sample flow rate would produce a corresponding decrease in gas adsorption.

The positive artifact may be offset in part by negative artifacts due to evaporative loss from collected particles during sampling or in post-sampling transport and storage. Other complexities include changes in atmospheric conditions (e.g., temperature, humidity, or gas-phase concentrations). The filter may experience a wide range of conditions that affect the delicate gas-particle phase dynamics. Therefore, measurements may be most reflective of the conditions present during the final hours of sampling.

Different strategies have been used to address sampling artifacts. Depending on sampling configuration, the relative importance of artifact effects will vary. The simplest approach relies on a blank filter subtraction, providing the filter remains exposed to ambient air for a sufficient equilibration time. Alternatively, research has shown that an in-line pair of filters may be sufficient to correct for sampling artifacts, provided both filters reach an equilibrium state with gas-phase species (Kirchstetter et al. 2001; Subramanian et al. 2004). Kirchstetter et al. (2001) also showed that filter absorption capacity varied substantially by manufacturer or from different batches from the same supplier. These subtraction approaches assume the positive gas-phase adsorption artifact dominates over the negative "blow-off" or evaporative negative artifact. The use of a denuder upstream of the filter can minimize or eliminate the positive artifact, but it may increase the negative artifact by driving the partitioning equilibrium to the gas-phase. Several researchers have shown this negative artifact to be small relative to the collected ambient OC by quantifying the collected volatilized particles (Mader et al. 2001; Mader et al. 2003; Subramanian et al. 2004). For successful implementation of this sampler configuration, however, the denuder efficiency must be well-characterized.

Filter-based Analytical Methods

Three filter-based analytical methods are described in this report. The most widely used analyses, thermal-optical methods, provide the OC and EC mass of a sample; 24-hour sample collection periods are common in the U.S. Although the various analytical protocols used generally agree on the total mass of carbon, they often disagree in their apportionment of this total to the organic or elemental fraction. The second approach describes isotopic analysis of carbon, which allocates the carbon mass to modern (biomass) or fossil sources. The final approach, chromatographic-mass spectrometric analysis, offers a more detailed characterization of OC at the molecular level. Although this technique may yield source markers and is useful for source apportionment, proper analysis requires large sample volumes and careful sample handling procedures. It also fails to characterize a significant portion of the OC.

Thermo-optical Methods. The simplest approach to apportioning carbonaceous aerosols to their organic and elemental components differentiates the fractions based on temperature alone. The thermally evolved carbon may be converted to methane or carbon dioxide and detected by flame ionization detection or infrared methods, respectively. Generally, the split between OC and EC evolution occurs at a specified temperature and carrier gas, with inert gas and lower temperature evolution corresponding to OC and higher temperature evolution in the presence of oxygen to EC. Unfortunately, complications arise due to pyrolysis of organic carbon, referred to as charring. This effect will tend to assign more carbon to the EC fraction. Current practice relies on optical methods to correct for charring, either through reflectance or transmittance. Generally, the trasmittance measurement may better correct for pyrolysis since it measures through the entire filter, whereas reflectance depends on the filter surface. Different analytical protocols, however, produce different results. Further investigation is needed to better understand the differences due to variations in evolution temperature and rate, and optical correction technique.

A range of analytical protocols are employed around the world, although only two enjoy widespread use in the United States. The first method was developed by the Interagency Monitoring of Protected Visual Environments (IMPROVE) program for aerosol measurements in Class 1 areas, while the second is based on National Institute for Occupational Safety and Health (NIOSH) diesel particulate protocols modified for the U.S. EPA's Speciation Trends Network (STN). Both employ specific schemes, with IMPROVE relying on Thermal Optical Reflectance (TOR) to demarcate the transition from OC to EC, while STN employs Thermal Optical Transmittance (TOT).

The IMPROVE protocol incorporates a combination of temperature changes (known as "ramps"), and analysis air stream compositions (e.g., He and He/O_2) to determine the OC and EC concentrations. The approach relies on variable analytical time-steps that permit the evolution of seven discrete peaks[2]. Temperature changes occur once the signal returns to baseline levels. To account for pyrolysis, filter reflectance is recorded. The split between OC and EC is defined as the point during analysis at which the filter reflectance reaches its original value; generally, the reflectance will decrease as OC pyrolyzes and then increase as pyrolyzed organic carbon (POC) and EC are driven off the filter. Concentrations of OC are determined through summation of the numbered OC fractions and the pyrolyzed portion of the first EC peak. Likewise, EC concentrations are determined by summing the three EC peaks and subtracting out the pyrolyzed carbon.

The STN method uses a different set of temperature parameters combined with the inert and oxidizing atmospheres.[3] The peak temperature achieved during the inert phase far exceeds that used by the IMPROVE protocol. Additionally, to speed analysis time, no delay occurs with the temperature ramping as carbon is evolved; temperature steps occur for a specified time interval. Akin to the IMPROVE approach, the split between OC and EC occurs when the optical transmittance recovers to its initial value.

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[2] IMPROVE fractions are evolved as follows: Under inert He atmosphere, OC1 (ambient to 120 °C), OC2 (120 to 250 °C), OC3 (250 to 450 °C), OC4 (450 to 550 °C), add 2% oxygen (oxidizing atmosphere), EC1 (550 °C), EC2 (550 to 700 °C) and EC3 (700 to 800 °C).

[3] STN carbon is evolved under an inert helium atmosphere from ambient to 310 °C, and ramps to 480, 615, 900 °C, the oxygen is added with temperature dropping back to 600 °C then ramps up to 675, 750, 825, and 920 °C.

Comparative results of these two methods shows measured EC concentrations are lower for STN than for IMPROVE.

A number of researchers have studied how reported measurements of OC and EC are affected by protocol differences. Most studies show agreement in determination of total carbon (TC) but report substantial differences in the OC and EC apportionment. Interlaboratory comparisons of EC are especially poor given its generally small contribution to overall measured carbon, with differences ranging from a factor of 3 to 7 (Schmid et al. 2001; Watson et al. 2005a). Chow et al. (2001) compared IMPROVE and NIOSH protocol carbon measurements and found agreement for TC. EC measured by the NIOSH protocol was about half that from the IMPROVE method, with two factors identified to explain the difference: (1) evolution of EC at NIOSH high temperatures during the inert atmosphere phase assigned to OC, likely oxidized by mineral oxides in the particulate (rather than oxidation by the O_2 added later in the analysis); and (2) consistently higher pyrolysis correction for transmittance used by the NIOSH protocol. Later analysis revealed that reflectance corrections were consistent under varying evolution temperature protocols (Chow et al. 2004). This study also found substantial in-filter charring occurs, which affects transmittance through the filter but does not appreciably affect reflectance.

These methods fail to agree in part because the underlying assumptions for partitioning the TC into OC and EC fractions are not met. At least one of the two assumptions must hold: either (1) pyrolyzed carbon must evolve prior to EC or (2) the optical properties of the pyrolyzed carbon equal those of EC. Several studies tested these postulates and found neither to be true (Yang and Yu, 2002; Chen et al. 2004; Subramanian et al. 2006). In many cases, EC, which is deposited on the surface of the filter, evolves prior to the pyrolyzed carbon, which often forms within the filter matrix (Chen et al. 2004). Subramanian et al. (2006) determined that the attenuation coefficient of pyrolyzed carbon exceeded that of EC. Taken with the co-evolution, EC determined with transmittance was found to be an underestimate.

Note that these methods measure the mass of carbon in the aerosol. They do not directly account for other elements that may be bound to the carbonaceous species, such as hydrogen, nitrogen, oxygen, or sulfur. To fully account for the mass associated with organic carbon, other approaches must be used. These include mass balance approaches or different analytical methods; they are discussed below in further detail below, when specific monitoring results are presented for New York State.

Isotopic Methods. Isotopic analyses of the three major C isotopes (^{14}C, ^{13}C, and ^{12}C) provide an important component in understanding sources of carbonaceous aerosols. A number of approaches have been employed. When used together with other metrics, these isotopic methods can better describe the relative importance of different source types.

Isotopic ratios of ^{14}C to ^{12}C determined by accelerator mass spectrometry are used to differentiate between fossil fuel and modern carbon sources. The approach relies on the decay behavior of ^{14}C, which has a half-life of nearly 5700 years. The carbon in fossil fuels contains negligible quantities of ^{14}C compared to those found in today's atmosphere given its geological age. Cosmic beta-rays impacting nitrogen in the upper atmosphere provide a steady stream of ^{14}C that becomes CO_2. Vegetation takes up this $^{14}CO_2$. The isotopic content of carbon aerosols derived from combustion of these contemporary plant materials will reflect the isotopic ratio in the atmosphere at the time of uptake.

The fraction of modern carbon is defined as a ratio of ratios: aerosol sample isotopic ratio to the standard reference material (oxalic acid) isotopic ratio representative of 1890 (Hildemann et al. 1994, Klouda and Connolly 1995; Hidy et al. 2004). Minor adjustments are made to account for ^{14}C generated through decay of materials related to nuclear bomb related materials. That recent injection of ^{14}C created an isotopic signature that has gradually been decaying back to the reference level. This adds a layer of complexity for long-lived modern carbon sources such as trees, which incorporate carbon over decades. Their isotopic content represents a time-integrated average of recent atmospheric carbon (Lewis et al. 2004).

While fossil fuel carbonaceous aerosols are generally anthropogenic, modern carbon comes from a variety of sources that may be anthropogenic or biogenic in nature. Biogenic sources include secondary production from vegetative emissions, vegetative detritus and forest fires. Cooking and residential wood combustion are considered anthropogenic sources of modern carbon. Recent research has relied on isotopic measurements to assist in apportioning carbonaceous aerosols to clarify the potential for development of controls to reduce their sources.

Radiocarbon measurements in Texas revealed a substantial fraction of organic carbon to be of biogenic origin, likely from secondary aerosols or fires (Lemire et al. 2002). Analyses conducted on Look Rock, Tennessee, aerosols showed the seasonal variation in the fossil carbon fraction ranged from 10 to 70% (Tanner et al. 2004). Results from Nashville during summertime and Tampa in May were similar, with roughly 30% fossil carbon contribution (Lewis et al. 2004; Lewis and Stiles, 2006). Other results reported from the southeastern U.S. for fall and wintertime measurements displayed differences between urban and rural sites, with the relative contribution of fossil carbon in urban areas two to three times higher (Zheng et al. 2006). These urban-rural differences are not unique to the Southeast. Measurements in the Denver area are dominated by fossil carbon while those at Yosemite National Park are mostly biogenic in origin (Klinedinst and Currie 1999; Bench 2004). At Yosemite, researchers found a base level of fossil carbon contribution with virtually all the variation due to biogenically derived carbon.

Researchers in Tasmania also observed a small, constant background fossil component, which they attributed to transported pollution (Jordan et al. 2006). In Zurich, the fossil OC component was 30% year-round (Szidat et al. 2006). Other overseas research has focused on carbon in PAHs, with fossil carbon responsible for most of the measured PAHs in three sites in Europe (Mandalakis et al. 2005) and Tokyo (Kumata et al. 2006).

Stable carbon isotope (^{13}C) measurements using an isotopic ratio mass spectrometer (IRMS) have also been conducted on carbonaceous aerosols. These analyses can potentially be used to differentiate between biogenic C sources depending on their primary photosynthetic pathways (Simoneit 1997; Turekian et al. 2003). Other researchers have applied stable isotope analysis to OC and EC (Huang et al. 2006) and individual molecular species (Glaser et al. 2005) for source apportionment with mixed results. Wang and Kawamura (2006) report latitudinal gradients in δ13 as evidence of a kinetic isotope effect, or preferential reaction of compounds based on isotopic composition. They hypothesize that the observed relationship corresponds to latitudinal trends in insolation where increased exposure to sunlight leads to preferential photodegradation; this implies isotopic ratios may be used to assess photochemical air-mass age.

Chromatographic Methods. Chromatographic methods have been used for molecular level analyses of filter samples. Most samples are collected on filters using high flow rates over 24-hour time periods. Often, sample composites are required to improve detection of trace molecular constituents of the aerosol. Most approaches rely on solvent extraction and other processing to prepare the sample for chromatographic analysis. Although considerable information can be gained, the methods are labor-intensive and demand careful sample handling procedures to minimize contamination.

Gas chromatography-mass spectrometry (GC-MS) is the most frequently used approach given its suitability for low-molecular-weight analyses. However, many highly polar species present in organic aerosol fail to elute on GC columns. A number of derivatization schemes have been employed (Mazurek et al. 1987, Kalberer et al., 2000) to permit the analysis of these more polar species that include alcohols, ketones, and carboxylic acids, although the choice of scheme requires some prior knowledge of aerosol composition. Common approaches include conversion of acids to esters and alcohols to ethers. These procedures complicate both quantification of the parent compounds and mass spectrographic interpretation, not to mention the possible occurrence of other chemical alterations during derivatization. Recovery estimates can affect proper quantification, and in addition, different parent compounds could generate similar derivatized species.

More recently, liquid chromatography-mass spectrometry (LC-MS) has been used (Antilla et al. 2005; Yttri et al. 2005). LC approaches remove the need for derivatization since polar species are elutable. Thus, the specificity and sensitivity are improved, as the compounds of interest are measured directly and the potential compound loss and transformation uncertainty inherent in chemical derivatization reactions are eliminated. Additionally, both basic and acidic species can be measured, permitting the characterization of organonitrogen compounds. Despite these potential advantages, analytical difficulties currently prevent this approach from providing the same quantitative dependability of GC-MS methods (Mazurek 2007, personal communication).

The absence of traceable standards represents a substantial drawback of these molecular identification techniques. Quantification can be difficult without appropriate standards. Additionally, the mass spectral libraries lack the wide range of atmospheric relevant compounds. Tentative compound identification requires time-consuming analyses of individual spectra.

On-line Methods

On-line monitoring methods are among the most notable recent advances contributing to knowledge of carbonaceous aerosols. On-line methods have distinct advantages over the more traditional off-line approaches. Advantages include highly time-resolved data, minimized artifact problems, and “real-time” results (e.g., no delay for sample processing). These gains, however, are offset by their lack of molecular specificity.

Aerosol Mass Spectrometry. AMS offers many advantages over more traditional filter-based sampling and post-collection analytical approaches. The instruments provide highly time-resolved information on a wide range of aerosol components (e.g., nitrates, sulfates, OC). Although a number of different configurations exist, the instruments can be described by primary components covering the interface with the ambient air to the final analytical mass spectral analysis. These components include the inlet configuration, sizing mechanism, vaporization and ionization method(s), and mass analyzer type. Brief descriptions of commonly used devices follow, along with their advantages and disadvantages. Applications of different instrument designs reported in the literature are also described.

The inlet provides the interface between the ambient air and the instrument. Several inlet types have been employed, including the aerodynamic lens, nozzle, and size-selective inlets. Ideal inlet design allows efficient and reproducible transfer of aerosols into the instrument. Particles are focused into a concentrated beam, relying on their inertia to separate them from the bulk gas. One approach used by Mallina et al. (2000) relies on a variable pressure inlet. A succession of small chambers, or skimmers, attached to a vacuum to remove the bulk gaseous constituents (e.g., air), performs both particle concentration and size selection through nozzle pressure modulation. The smallest particles are removed with the gases, due to their small inertial mass, while larger particles impact and stick to the skimmer walls. Nozzle pressure variations influence which size particles transfer into the ionization chamber. A similar design, the aerodynamic lens (Liu et al. 1995a, b), consists of a multi-partitioned chamber with an in-line series of successively smaller orifices (Jayne et al. 2000). Vacuums at the inlet and outlet of the lens control the flow rate of the concentrated particle beam. Unlike the size-selective variable pressure inlet, which transmits a narrow band of particle sizes at one time, the aerodynamic lens transmits a wide range of particle sizes simultaneously.

The aerodynamic lens can be coupled with a mechanical chopper, a rotating device with radial slits that interrupts the particle beam transmission. Particles are accelerated through a fixed-distance chamber, which permits particle size determination through measured time of flight (TOF). A second approach to particle sizing relies on size-dependent scattering properties. This approach can suffer from low resolution, dependencies of particle composition-specific optical properties, and inability to characterize the smallest particle sizes due to inefficient scattering. Particle velocity is measured using multiple lasers, and the determined timing triggers ionization. Alternatively, particle size separation and determination occurs prior to introduction into the spectrometer through use of a differential mobility analyzer.

After the particles' size(s) are determined, they are vaporized and ionized. This can be accomplished in one step using a high-powered laser, a technique termed laser desorption/ionization (LDI). The reliance on a high-energy laser causes extreme fragmentation, which represents a primary drawback of the method. An advantage, however, is the ability to detect non-refractory (i.e. very stable, heat-resistant) aerosol components (Sipin et al. 2003). A variety of approaches can be used if the vaporization/ionization steps are decoupled (Sullivan and Prather, 2005). Vaporization can be achieved through impaction onto a heated surface, use of low-powered lasers, or through cooled collection and subsequent thermal desorption. Unlike LDI, these lower-energy vaporization methods maintain the integrity of the aerosol and improve quantification capability. Desirable properties of ionization techniques include high (or similar) ionization efficiency for all species, with a linear relationship between ions produced and molecular number (for quantification). Aerosol ionization is achieved through electron bombardment, or 'softer' techniques such as ultraviolet photo-ionization or chemical ionization. Soft ionization allows the characterization of the molecular ion due to limited fragmentation.

Once the aerosol has been ionized, the mass spectrometer (MS) separates the ions through electrical fields according to mass-to-charge ratios (m/z). For aerosols, the major spectrometer types employed are quadrupole, time-of-flight (TOF) and ion trap. The quadrupole MS has the disadvantage that it can only measure one m/z at a time. The TOF-MS can be configured to monitor both positive and negative ions in addition to collecting data from a range of m/z. Ion traps offer the ability for multidimensional MS (e.g., MS-MS) and can operate at higher pressures. They operate over a limited mass range, however, which can be a distinct disadvantage for complex aerosol mixtures.

Given the various choices available in instrument design, many different configurations have been reported in the literature. Instruments used for single-particle characterization include: aerosol time of flight mass spectrometer (ATOFMS) (Su et al. 2004), aerosol laser ablation mass spectrometry (LAMS) (Tan et al.

2002; Owega et al. 2004), particle analysis by laser mass spectrometry (PALMS) (Murphy et al. 2006), and the rapid single-particle mass spectrometer (RSMS) (Zhao et al. 2005). Several configurations for bulk aerosol measurements have been reported, such as the thermal desorption particle beam mass spectrometer (TDPBMS) (Tobias and Ziemann 1999), photoionization aerosol mass spectrometry (PIAMS) (Oktem et al. 2004), and Aerodyne aerosol mass spectrometer (Jayne et al. 2000; Allan et al. 2003a). Recently, the front end of the Aerodyne instrument was interfaced with a TOF-MS, thus permitting single particle characterization and nonrefractory particle detection (Drewnick et al. 2005).

Unlike many of the current research-grade instruments, Aerodyne produces a commercial-grade instrument used by a number of research groups. It comes in two different configurations, quadrupole and TOF. Some examples of analytical results obtained with the instrument (with an emphasis on NYS-based field campaigns) offer insights into the type of single-particle and bulk aerosol information that can be gathered through AMS. The Aerodyne instrument employs an aerodynamic lens to focus particles, followed by a mechanical chopper to determine the aerodynamic size. Efficient particle transmission occurs for sizes ranging from 60 to 600 nm in diameter (Drewnick et al. 2004a). Focused particles are then vaporized, ionized, and analyzed with either a quadrupole-MS or a TOF-MS. In the quadrupole design, the instrument operates in two alternating modes, one that scans all masses (up to 300 m/z) for all collected particles and the other measuring selected m/z to collect aerosol size-distribution data (Drewnick et al. 2004a). The method does not determine molecular level composition, but it does reveal information about particle composition by size through analysis of the relative contribution of ion fragments. It does not measure refractory, or heat-resistant, species such as EC or dust particles, as the vaporization temperature is maintained at 600 °C, a compromise for the optimal detection of sulfate, nitrate and organic particulate matter composition. A third analytical mode, Single-Particle TOF, may be used for the TOF-MS version of the instrument (Drewnick et al, 2005).

Analysis of data collected during the summertime in Queens, NY revealed a bimodal size distribution for organic aerosols, with one mode centered around 80 nm and the other about 350 nm. Similar bimodal results have been observed in other urbanized regions including Vancouver, Canada, and Manchester, England (Allan et al. 2003a; Boudries et al. 2004; McFiggins et al. 2005). Sulfate particle mass was mostly found in the accumulation mode, coincident with the larger particle size organic mode. (The accumulation mode covers particle sizes from 0.1 to 1 μm in diameter and accounts for much of the aerosol surface area and mass [Seinfeld and Pandis 1998]). Comparisons of these species-specific size distributions are used to assess the mixing state of the aerosol. In this instance, the small mode (80 nm) carbonaceous particles are likely externally mixed, since very little inorganic aerosol mass is present in that small size range. The larger particles in the accumulation mode may be internally mixed, as evidenced by the similar modal diameters and shape variations in time for organic and inorganic particles (Drewnick et al. 2004b; McFiggins et al, 2005). Single particle data collected with the latest TOF-MS instrument configuration confirm that similarity in the size distributions of different species indicate internally mixed particles (Drewnick et al. 2005). Wintertime data collected in Queens also showed periods with both externally and internally mixed particles.

The modal structure of the carbonaceous aerosol was observed to change from summer to winter in Queens (Weimer et al. 2006). During wintertime, the larger mode shifted about 200 nm toward the smaller mode, possibly merging with the small mode observed in the summertime. The apparent disappearance of the small mode could also be due to a corresponding shift to smaller sizes, which may not be detected by the AMS if the particles are too small (< 30 nm). A similar shift in mode size was observed in Manchester during wintertime (Allan et al. 2003a).

The small organic particle mode (centered about 80 nm) comes from local traffic sources, as supported by several other observations. For example, the mode varies along with other gaseous indicators of mobile source pollution (e.g., benzene, CO, NO_x). The intensity of the mode is also shifted in time, with a greater proportion of organic mass in the small mode during morning rush hour in Queens (Drewnick et al. 2004b). Measurements from remote areas generally lacked this small mode, unless directly impacted by urban air masses (McFiggins et al. 2005). In addition, the mass-to-charge fragment m/z 57 ($C_4H_9^+$) is very prominent in the organic small mode while m/z 44 is not. This implies the small mode particles are likely both primary and fresh in nature, having undergone little oxidation, since m/z 44 is dominated by CO_2^+ fragments that are derived from highly oxidized species such as poly-carboxylic acids. Temporal patterns support the characterization of m/z 44 as a marker of oxidized species, as the observed afternoon build-up resembles that of other photochemically derived pollutants such as ozone (Drewnick et al. 2004a).

In addition to direct analysis of temporal and particle-size data for these two fragments, m/z 44 and 57 have been used in source apportionment modeling to represent signature components of oxygenated organic aerosol (OOA) and hydrocarbon-like organic aerosol (HOA) (Zhang et al. 2005). The organic mass can be apportioned to these two broad categories, OOA and HOA, which explain most of the variance in the organic aerosol time series. The composite mass spectra associated with HOA and OOA resemble spectra from fresh mobile source (or combustion related) organic aerosol and aged organic aerosol, respectively. OOA, however, is not strictly secondary in nature, as it represents any aged organic aerosol, where the aging process implies oxidation (Zhang et al. 2005). The temporal and size variations in these two mass fragments provide useful tools in understanding the relative age of the aerosol and potential source influences.

Another useful application of AMS was demonstrated by installing the instrument on a mobile platform. By following vehicles operating on the streets of NYC, researchers captured tailpipe pollution plumes (Canagaratna et al. 2004). Particles in diesel exhaust plumes were centered about 90 nm in diameter, and their mass spectra showed a strong resemblance to that of lubricating oil (note these particles were similar to size to the smaller mode identified in ambient AMS measurements in Queens mentioned on the previous page). Plumes were identified using CO_2 as a tracer. Carbon dioxide levels were also used to determine actual emission rates from mobile sources through relationships among measured particle mass, CO_2 concentrations, and fuel consumption. This type of analysis provides invaluable information as current emission inventory data likely fails to represent actual emissions, since real-world variability in operating conditions (e.g., fuels, vehicle characteristics, ambient conditions) cannot be characterized by limited testing.

An Aerodyne AMS has also been used to study aerosols formed during chamber experiments (Bahreini et al. 2005), providing useful information on SOA particle density and fragmentation patterns. Comparisons of size distribution and mass determined by AMS with traditional sizing instruments (differential mobility analyzers) yielded SOA density, a parameter required for yield calculations. Many experiments assume a unit density for SOA in the absence of direct measurement, thus contributing to uncertainty in product yield estimates. Density measurements from experiments with and without seed aerosol showed higher particle densities in the latter case. Those particles that underwent nucleation also had higher levels of the m/z 44 fragment. The AMS spectra showed substantial concentrations of large organic fragments, suggesting that higher molecular weight species were present in the parent aerosols and may be oligomeric in nature. The researchers also note that in some instances, the m/z 57 fragment contributes substantially to the overall SOA signal, offering a caution in the use of this fragment as an exclusive marker for direct emissions.

Aethalometer. An aethalometer continuously measures the light attenuation of particles collected on a filter tape. In general, only EC attenuates the signal at the wavelength used (e.g., 880 nm), as any attenuation due to scattering of the filter is subtracted as background. Some instruments employ multiple wavelengths for measurement. Quantification is achieved through an assumed specific absorption for the sampled EC. The rate of change in attenuation due to deposited aerosol will be proportional to its ambient concentration. The site-specific absorption can be determined through comparison with collocated thermally measured EC.

Aethalometers provide highly time-resolved data and are both easy to deploy and easy to operate. They are also quite sensitive and may provide data that would be near the detection limit in remote locales. Aethalometer data have been used to illustrate diurnal behavior of EC (Venkatachari et al. 2006b). Additionally, sophisticated algorithms can derive different scales (local, neighborhood, or regional) of source influences (Watson and Chow 2001). One drawback is the reliance on a constant specific absorption. This can affect results, since the nature of the absorption may vary by season, humidity levels (Gelencser 2004), particle size or with changing mixture of sources, whose aerosols might have different absorptive properties. Additional uncertainty can be introduced through a loading effect. At high loading or optical saturation, the apparent absorption coefficient is reduced; because mass calculations rely on a constant coefficient, this leads to an underestimation of EC mass (Weingartner et al. 2003). As the filter tape advances, an apparent spike in concentrations may be observed (Goodwin 2005). A correction has been recommended that accounts for the scattering effects of the filter material and nonabsorbing particles (Arnott et al. 2005).

SemiContinuous Thermal-Optical Carbon Analysis Semicontinuous carbon analyzers operate on the same principal as traditional time-integrated filter-based methods and have been shown to provide comparable measurements for OC and EC (Bae et al. 2004). Comparisons at the Baltimore Supersite showed the semicontinuous EC and OC to be lower than collocated integrated filters, by 12 and 22 percent respectively (Park et al. 2004). Unlike traditional methods, the analysis is conducted in the field in real time. An inlet cyclone or similar approach is used to remove particles larger than 2.5 microns. In some configurations, the sample then flows through a carbon-coated denuder to remove organic vapors; this minimizes the positive adsorption artifact, though it may increase the negative volatilization artifact. Arhami et al. (2006) report the positive artifact as five times higher than the negative artifact, which justifies the use of a denuder. Other instruments may operate parallel collections with one measuring ambient particles and adsorbed gases while the other provides a dynamic blank of adsorbed gases with particles removed upstream by a Teflon filter (Lim et al. 2003). Particles are collected on a quartz fiber filter for the time period of interest, often an hour or two, and then vaporized using the NIOSH 5040 or a similar protocol. Laser diode transmission is often used to correct for pyrolysis of organic carbon. The evolved carbon can be oxidized to CO_2 and detected via enhanced non-dispersive infrared (NDIR) methods. Or it can be reduced to CH_4 and detected via flame ionization detection (FID), although the latter method is more cumbersome with the concurrent need for hydrogen gas.

The use of semicontinuous instruments offers substantial improvement over a 24-hour integrated filter measurement, because carbonaceous aerosol concentrations vary over a much shorter time-scale than one day. Short-term measurements avoid the potential bias toward the conditions at the end of the sampling period that may occur for 24-hour samples. The smaller sample volumes, however, may increase the chance of below detection, especially for EC in remote areas. Although less specific than AMS data, OC and EC data from these instruments can be used to discern changes in source influence and to estimate secondary versus primary contributions to the OC (Polidori et al. 2006).

Other Measurement Methods for EC Several other methods exist for measurement of EC, including the particle soot absorption photometer (PSAP), integrating plate or integrating sphere methods, Raman spectroscopy, and photoacoustic spectrometry (Galencser 2004). The first three methods, similar in principle to the aethalometer, measure transmission through a filter. Raman spectroscopy specifically measures the graphitic light-absorbing component of the aerosol. The photoacoustic method does not rely on filter collection, instead relying on absorptive heating of light absorbing aerosols. The heating increases the pressure in the collection chamber, which generates waves detected with a sensitive microphone (Arnott et al. 1999, 2005).

Emissions Inventory and Source Characterization

Many sources exist that emit carbonaceous aerosols or their precursors. These include natural sources such as vegetation, bioaerosols, soils, and the ocean as well as anthropogenic sources such as biomass burning and fossil fuel combustion (Gelencser 2004). While EC is derived from direct emissions (primary), OC is composed of both primary and secondary components. That characteristic complicates the development of emissions inventories for carbonaceous aerosols. The complexity precludes an in-depth discussion of emission inventories for carbonaceous aerosols in this report. Nonetheless, readily available inventory details specific to NYS are provided in the control technology chapter. Inventory information presented here is discussed in relationship to other topics in this chapter.

Emissions inventories (EI) represent the basis for much air quality work. Currently, they can be used directly to assess primary emissions. They are a crucial input for chemical transport modeling, a fundamental tool used for evaluating air quality and control strategy effectiveness. They can also provide input for receptor models such as CMB, or comparison information for derived profiles from PMF based on ambient data. However, current inventories are inadequate for effective evaluation of OC and EC impacts on air quality. Although inventories may have been useful in their current form for past air quality management tasks, the significant improvements realized through controls of major pollution sources necessitate substantial refinements to existing emissions inventories if the proper decisions for continued forward progress are to be made (NARSTO 2005; Miller et al. 2006). Inventory development should be focused on providing the information needed for modeling and other purposes. This includes better temporal resolution, more complete characterization of specific species in addition to total mass emissions of OC and EC, better documentation of methods, and uncertainty estimates, to name a few.

Much of the available information on OC and EC emissions has come through efforts to provide source profile inputs for CMB receptor modeling. Other receptor models, such as PMF or UNMIX, rely on these profiles to interpret the derived factors. Motor vehicle emissions represent a substantial fraction of the available profiles, along with wood combustion and food preparation. A complexity arises due to the highly variable nature of these emission sources. For motor vehicles, important factors include such things as fuel burned, engine size, vehicle age (Harley et al. 2005; Schaeur et al. 2002), maintenance record, and operating conditions (Fraser et al. 2002). For wood combustion, variables include wood type, combustion conditions, and burn device (e.g., wood stove or fireplace). Even if all of these variables are incorporated, other information is needed, such as when and where the emissions occur. Equally important are the target species (e.g., OC, EC, VOCs, elements, or major ions) and the sampling (e.g., temperature, dilution, filter or adsorbent type) and measurement methodology (e.g., OC/EC protocol).

Considerable effort has focused on developing motor vehicle emission profiles, since these sources are many and ubiquitous. Work has shown the importance of capturing different modes of operation, including start-up and "hot stabilized mode," which is essentially normal driving (Fitz et al. 2004; Brandenberger et al. 2005), as the emission rates and species ratio (e.g., EC-to-OC) may be quite different for different operating conditions. In addition, research has shown that much of the emissions may come from a small fraction of the vehicles, which are sometimes termed "smokers" for their visible exhaust plumes. The type of information reported varies, as well. For example, some studies may only report gaseous emissions (McGaughey et al. 2004). Some may report mass of major species such as OC and EC (Shah et al. 2004), while others report molecular level data such as individual PAHs, hopanes, and steranes (Phuleria et al. 2006; Spencer et al. 2006; Zielinkska et al. 2004; Fraser et al. 1999; Rogge et al. 1993a). Generally speaking, the more specific the measurements, the more useful they are. For example, Schaeur et al. (1999) report that ratios of hopanes, steranes, tricyclic terpanes,[4]and EC may act as tracers for diesel exhaust. They also report that the hopanes in the exhaust are derived from lubricating oil and not the diesel fuel itself. In the past, ratios of EC and OC mass emission rates have been used to differentiate between diesel and spark-ignition vehicles, where EC was treated as a primary tracer of diesel. Shah et al. (2004), however, showed that the EC-to-OC ratio of diesels operating at very low speeds resembled that of gasoline-powered vehicles. This points to the utility of measuring a broad array of species, since the geological biomarkers pristane and phytane are present in diesel exhaust but not gasoline (Schaeur et al. 1999).

Emissions from wood combustion constitute another important category studied. Like mobile sources, the level of characterization differs depending on the experiment, although many studies measure gas and particle phases (Fine et al. 2001, 2002; Fine et al. 2004; Hays et al. 2002; McDonald et al. 2000; Rogge et al. 1998; Schaeur et al. 2001). Chen et al. (2006b) report emission dependence on different fuels and flame conditions. Studies characterize fireplace emissions (Rogge et al. 1998; Schaeur et al. 2001; Fine et al. 2001, 2002), woodstoves (Fine et al. 2004), and biomass burning (Hays et al. 2002) of different tree species. Common tracer species for wood combustion include K, retene, levoglucosan, and related sugar anhydrides. Ratios of OC to EC are usually higher than many other emission sources. Fine et al. (2001) present evidence of species-specific molecular tracers as well (e.g., betulin from paper birch, juvabione from balsam fir).

Profiles have also been obtained from food cooking processes, including meat charbroiling (Rogge et al. 1991; Schaeur et al. 1999; McDonald et al. 2003) and deep-frying operations (Schaeur et al. 2002). Meat

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[4] Crude oil and products derived thereof contain a number of hydrocarbon species classes, including hopanes, steranes, and tricyclic terpanes and individual molecules such as pristane and phytane. These compounds are geological biomarkers and ratios among the species are used as identifiers in geochemistry to differentiate among crude oils.

cooking operations in Los Angeles were determined to contribute nearly one-fifth of the primary OC emissions (Rogge et al. 1991) and ambient OC concentrations (Schaeur et al. 1996), with similar contributions (~18%) to emissions in the tri-state New York City nonattainment area (this work), based on current emissions inventory estimates. Tracer compounds for meat cooking include cholesterol, alkanoic (fatty) acids, and nonanal. Schauer et al. (2002) identify two alkanoic acids as potential tracers for cooking with seed oils. This source was not included in source apportionment modeling for Los Angeles (Schaeur et al. 1996), but it may account for a portion of the fatty acids not accounted for in the modeling. Motor vehicle exhaust and wood smoke may also contribute fatty acids to the PM. Recent work by Mazurek (2006) shows high ambient levels of these fatty acids in New York. Source apportionment may allocate some of the measured mass to frying. This would expose a deficiency in the current emission inventory, which lacks estimates for this source category.

In addition to the major categories presented, a handful of other emission measurements have been performed. The characterized sources include mobile source related emissions (e.g., road dust, tire dust, and brake lining dust) (Rogge et al. 1993b), soils (Rogge et al. 2006), leaf abrasion (Rogge et al. 1993c), No.2 fuel combustion (Rogge et al. 1997a), and asphalt roofing tar pots (Rogge et al. 1997b). Other work has focused on biogenic emissions of OC precursors such as sesquiterpenes and terpenes (Helmig et al. 2006), which are especially important to the chemistry of forested regions. These studies have generated detailed compositional profiles that can be used to tease out each source's contribution to ambient concentrations of $PM_{2.5}$ using source apportionment tools.

AMBIENT MEASUREMENTS AND DATA ANALYSIS FINDINGS

In recent years a number of monitoring initiatives and studies have occurred in New York State, including both routine measurements and special studies. In addition, measurements in surrounding states help provide information on air quality in the broader Northeast region. Taken together, analysis reveals that a large fraction of the fine particle burden in the state is comprised of carbonaceous species. A substantial portion of that aerosol is generated from sources within the state.

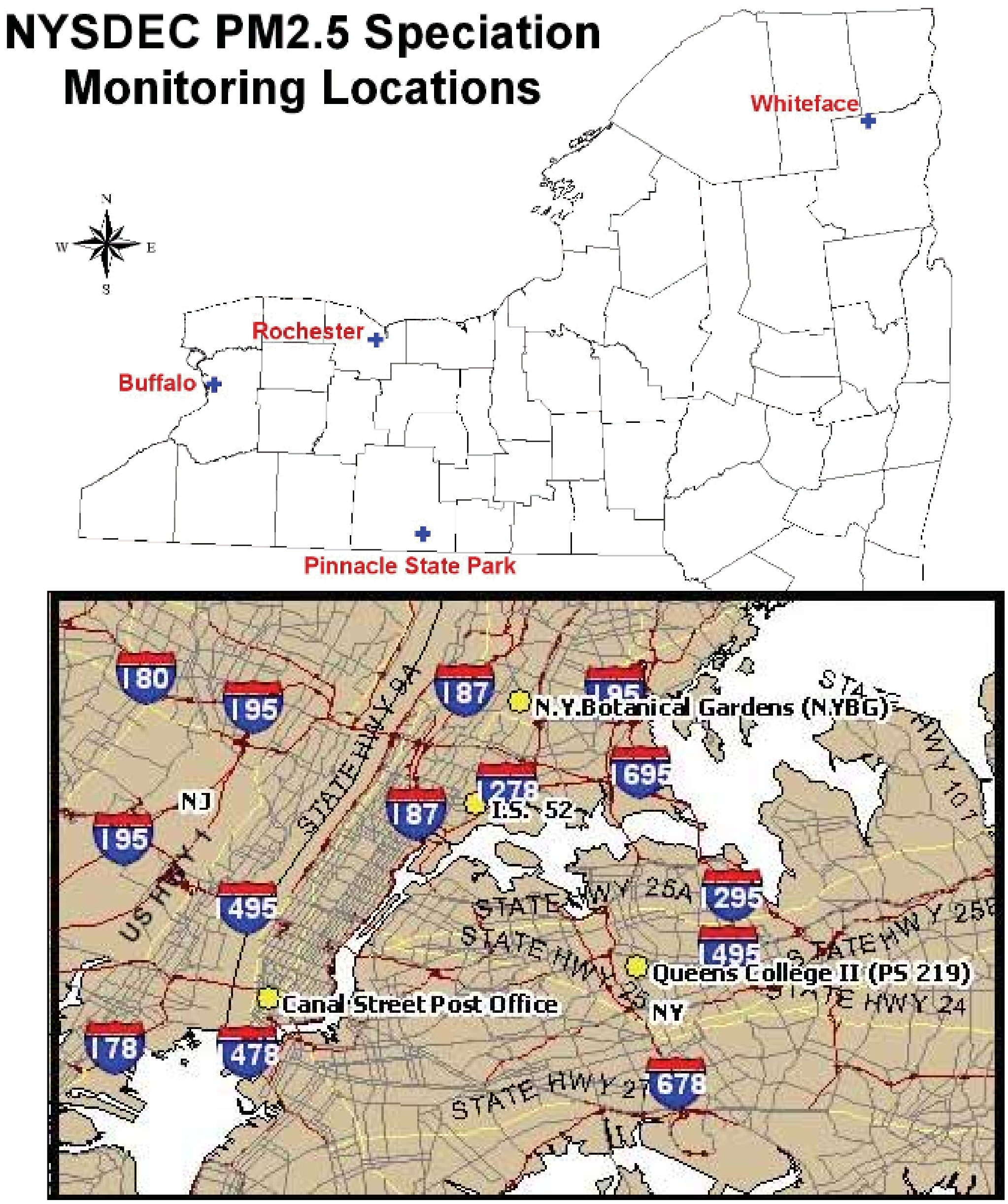
The following discussion presents and summarizes data analyses of carbonaceous aerosols in New York State. Much of the data come from the two routine monitoring networks that operate in New York and surrounding states: the STN, designed primarily for NAAQS $PM_{2.5}$ assessment, and the federal government's IMPROVE program, which focuses on regional haze. In both programs, samples are taken every third day as a 24-hour time-integrated filter and are analyzed for mass, OC/EC, major ions, and elements. These data are supplemented with results from two special studies: the U.S. EPA's Supersite program, designed for research and development of monitoring equipment and improving scientific understanding of aerosol processes, and Speciation of Organics for Apportionment of $PM_{2.5}$ (SOAP), intended to complement the research activities of the New York State Supersite. Additional relevant source apportionment and regional studies are included. These data sources provide the basis for the assessment of the sources, impacts, and behavior of carbonaceous aerosols in the State.

Filter-based Carbon Aerosol Measurements in New York State

This report relies on data from eight samplers (six R & P Partisol model 2300s and two MetOne Spiral Aerosol Speciation Samplers [SASS]) operated by New York State as part of the STN in support of the $PM_{2.5}$ monitoring program (Figure 1-1). Sampling began in early 2000 at the New York Botanical Gardens in the Bronx with the final site, Canal Street, coming online in August 2002 (Table 1-2). NYS Department of Environmental Conservation data analyzed for this report cover the period from the beginning of the sample record through mid-November 2005, with sample frequency every third day with the exception of the Buffalo site, which operated every sixth day. The current network has only seven sites, with Botanical Gardens ceasing operation at the end of 2005, although an additional monitor may be placed in Albany.

The analyses that follow describe data handling procedures and different approaches used to describe the carbonaceous $PM_{2.5}$ measured in NYS. The initial step reviews field and trip blank data to determine the applicable artifact correction of the carbonaceous aerosol concentrations. Subsequent analysis estimates the site-specific OC mass adjustment factor. The derived OC and EC mass data are combined with the other major $PM_{2.5}$ species to show their substantial contribution to the total measured mass across the State, on both an annual and seasonal basis. A more in-depth seasonal analysis of OC and EC reveals differences and similarities between urban and rural sites. EC levels at remote areas stay reasonably constant while wintertime increases occur at some urban sites. OC levels peak in summertime across NYS.

Figure 1-1. Upstate and NYC Region Speciation Monitor Locations.



Several techniques are used to investigate the data in further depth. The EC tracer method assesses the relative contribution of primary and secondary processes to OC. This method indicates a substantial secondary organic aerosol fraction, especially during the summertime, with the rural SOA fraction higher than the urban fraction. Comparisons of weekday and weekend data show the influence of local sources in urban areas. Further estimates of local and regional source signatures are determined through contrasting a rural site with urban sites. Both EC and nitrates are dominated by local sources in urban areas, while the sulfate signature appears to be mostly regional in nature. Organic carbon presents the biggest challenge given the complex nature of its sources, but it likely has substantial local and regional influences.

Table 1-2. Speciation Trend Monitor Information.

Site Name	Site FIP	County	Start Date	Sampler Type
Buffalo (BUFF)	36-029-0005	Erie	01/20/2002	R&P Partisol 2300
Whiteface (WHTE)	36-031-0003	Essex	05/25/2001	R&P Partisol 2300
Rochester (ROCH)	36-055-6001	Monroe	04/19/2001	R&P Partisol 2300
Pinnacle State Park (PINN)	36-101-0003	Steuben	02/06/2001	R&P Partisol 2300
Botanical Gardens (NYBG)	36-005-0083	Bronx	02/15/2000	MetOne SASS
IS 52 (IS52)	36-005-0110	Bronx	01/22/2001	R&P Partisol 2300
Queens College (QCII)	36-081-0124	Queens	04/04/2001	R&P Partisol 2300
Canal St (CANL)	36-061-0062	Manhattan	08/15/2002	MetOne SASS

Source: NYS Department of Environmental Conservation

Blank Correction for Carbon Aerosols. As noted above, quartz-fiber filter measurements of carbonaceous aerosols experience both positive and negative artifacts from adsorption of gas-phase species and volatilization of particulate species, respectively. Corrections for the positive artifact are often made through filter subtraction. Previously, artifact estimates for New York's STN data have been derived through a variety of methods (Civerolo and Sistla, 2006; Kim et al. 2005; Schwab, et al. 2004) and these results have informed the approach used in this work.

Civerolo and Sistla (2006) computed the average and standard deviation of organic carbon blanks using data through August 2005 for each measurement site. Using a more limited data set (through winter 2003), Schwab et al. (2004) pooled blank filters by sampler type to determine corrections for both elemental and organic carbon. Their analysis showed substantial differences in calculated EC blank levels between the two sampler types, which they attributed to the silicon impactor grease used in the R & P sampler. Studies have demonstrated that grease-related OC artifacts may be miscategorized as EC (Flanagan et al. 2003). This occurs because the transmission signal returns to the baseline (which is the definition of the split between OC and EC) before all of the carbon has evolved. Thus, carbon detected after the split time is assigned to the EC fraction.

In many instances, blank filter data may not exist or be available to the end user. One correction method relies on regression of measured organic carbon against concurrent $PM_{2.5}$ mass (Kim et al. 2005). In theory, the y-intercept of this regression represents the blank level, since the axis occurs where total $PM_{2.5}$ mass is zero. The result, however, seems to overestimate the true artifact; given the widespread scatter, its use should be viewed with caution.

The analysis conducted for this report includes three months beyond that used by New York's DEC (Civerolo and Sistla, 2006). Field and trip blank filters were pooled because no substantial difference was observed between their average values. In addition, sites were grouped into three sets: one urban pair of

SASS samplers (CANL/NYBG), one rural pair of R & P samplers (PINN/WHTE), and four urban sites using R & P samplers (IS52/QCII/BUFF/ROCH)[5]. Although individual corrections were calculated, the results were similar for the groupings and are shown in Table 1-3. Results for the two urban groupings are essentially the same, with the rural pair having a slightly lower calculated OC correction. The values here are more conservative estimates of the artifact, relying on the median value instead of the mean, as used previously (Schwab et al. 2004, Civerolo and Sistla 2006).[6] This minimized the number of negative blank-corrected OC mass values. In addition, the qualitative results compare well with both expectations that little to no elemental carbon should be found on blank filters and urban corrections should be greater than rural ones. This seems reasonable, as the artifact is derived from gas-phase organic adsorption onto the filter, and gas phase concentrations are greater in urban areas.

Table 1-3. Elemental and Organic Carbon Correction Factors as Determined by Field and Trip Blank Filters.

	Elemental Carbon (μg/m^3)	Organic Carbon(μg/m^3)
CANL/NYBG	0.01	0.94
PINN/WHTE	0.01	0.83
IS52/QCII/BUFF/ROCH	0.01	0.92

Organic Carbon Mass Adjustment Factor. After blank subtraction, the organic carbon values were analyzed to determine mass associated with organics. Historically, the OC mass values derived from thermal-optical methods are adjusted by a factor of 1.4 to account for the mass of non-carbon elements associated with the organic aerosol (e.g., oxygen, hydrogen, nitrogen, and sulfur).[7] Recent studies have revealed drawbacks to this approach, suggesting that both spatial and temporal variation should be incorporated into the mass adjustment factor (El-Zanan et al. 2005; Russell 2003; Turpin and Lim 2001). The bulk of the variation should be driven by the age of the aerosol, which reflects the extent of atmospheric processing or oxidation of the particles. In general, more-processed aerosol requires a larger adjustment factor to account for increased oxygen content. Spatially, areas dominated by primary or direct emissions (e.g., urbanized areas) likely need a lower multiplier than areas impacted by aged aerosols.

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[5] The Rochester sampler was moved partway through the study period. Data obtained from the two sites were pooled. Although differences between the two sites could potentially affect the results, the use of a longer data set outweighed these considerations.

[6] The median value is roughly equivalent to the geometric mean, as the distribution of blank measurements follows a log-normal distribution. Note also extreme values as defined by being outside two standard deviations of the ln mean were removed prior to calculating the correction factors.

[7] The adjustment factor represents the ratio of the average molecular mass to the carbon mass associated with the carbon aerosol. On a molecule specific basis, this value can range from just over one to greater than three (Huebert and Charlson, 2000).

Several different approaches have been employed to determine the appropriate adjustment factor. Turpin and Lim (2001) take a theoretical approach to provide limiting values by assessing average factors by compound class (e.g., alkanes, aromatics, acids). They also provide a review of derived values based on molecular scale measurements, which has the drawback of not fully characterizing the total carbon mass. Fourier transform infrared spectroscopy (FTIR) was used to develop a method independent of thermal-optical or GC-MS methods (Russell 2003). FTIR gives information on the functional group (e.g., carbonyl, acid, methyl) composition of the entire carbon aerosol that can reveal the proper adjustment factor. A comparison of two other techniques, mass balance and solvent extraction, was conducted with filters from the IMPROVE network (El-Zanan et al. 2005). The extract approach yielded a range of 1.58 – 2.58 for adjustment factors for the five sites studied. The mass balance approach[8] gave a similar range of values. The latter approach also revealed seasonality in the results, with a larger factor obtained for summertime as compared to wintertime. The mass balance approach has been used elsewhere as well (Bae et al. 2006; Frank 2006; Malm et al. 2005). Using STN data, researchers have applied a modified method that accounts for associated water and nitrate loss from the Teflon filter used for total $PM_{2.5}$ mass determination (Bae et al. 2006; Frank 2006). Sensitivity tests showed that water retained on the filter had the greatest impact on the factor determined through mass balance (El-Zanan et al. 2005).

The mass balance approach as described by Frank (2006) was used in this report since its successful application was demonstrated for a STN data set, including one of the New York State DEC sites (IS52). Sample-specific OC adjustment factors were calculated using the blank corrected carbon data along with sulfate, nitrate, and ammonium concentrations from the nylon filter and an estimate of fine crustal mass and $PM_{2.5}$ mass from the Teflon filter. Fine particle concentrations were adjusted for particle-bound water and seasonally adjusted nitrate loss. This water and nitrate adjustment resulted in a decrease in the mass factor adjustment of roughly 8%. Sample-specific factors spanned a wide range, in part due to substantial variation in OC concentrations, which are used as the denominator to determine the factor. Some site-specific seasonal dependence of the OC factor was observed, as might be expected due to seasonal differences in the relative contribution of secondary organics. The limited sample size and variability, however, precludes the use of seasonal factors. Instead, annual factors were determined for each site.

The annual carbon mass adjustment factors calculated for this report are shown in Table 1-4 and range from 1.3 to 1.6. These are fairly conservative values, with the trend consistent with lower multipliers calculated for regions more directly impacted by fresh emissions. Despite the conservative approach used here, the values compare reasonably well with reported results for Queens (1.27-1.59) and Pinnacle State Park (SP) (1.51- 1.87), which relied on a three-year time record of the same data set (Bae et al. 2006). That study concluded no substantial seasonal variations exist for the organic mass adjustment factor at the urban site,

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[8] The mass balance approach yields the mass from organic carbon aerosol by subtracting mass of sulfate, nitrate, ammonium, elemental carbon, and crustal mass from the measured $PM_{2.5}$ mass. This value is divided by the OC determined from the quartz filter to obtain the adjustment factor.

but the study did calculate a substantially greater factor during warmer months relative to the cooler months at the rural site. Using AMS results from wintertime in Flushing, NY, Venkatachari et al. (2006) determined multipliers ranging from 1.24 to 1.61. For the current study, a sensitivity analysis was conducted to assess variation in the adjustment factor due to the use of the water/nitrate mass correction versus no correction and the use of the average correction factor versus the median. The water/nitrate mass correction resulted in a decrease in the multiplier of ~10%; using the median instead of the average produced a ~20% decrease. Based on the adjustments used, some mass was always unaccounted for in the seasonal and annual speciated mass calculations. The "unaccounted" mass contains particle-bound water, although that fails to explain the entire difference. The rest of the difference is likely mass associated with OC. In some instances, the use of larger multipliers would have resulted in an overprediction of mass by speciated measurements.

Table 1-4. Organic Carbon Mass Adjustment Factors Determined Through Mass Balance Calculations.

Site Name	OC Factor		Site Name	OC Factor
Buffalo (BUFF)	1.5		Botanical Gardens (NYBG)	1.4
Whiteface (WHTE)	1.6		IS 52 (IS52)	1.4
Rochester (ROCH)	1.6		Queens College (QCII)	1.5
Pinnacle State Park (PINN)	1.6		Canal St (CANL)	1.3

Reconstructed Mass Data. Reconstructed mass data are presented for all eight sites, combined over the entire data record, as presented in Figure 1-2 and Figure 1-3. These speciated measurements provide an indication as to which species are important contributors to $PM_{2.5}$ levels. (Note the total STN $PM_{2.5}$ gravimetric mass from the Teflon filter exceeds that from the collocated Federal Reference Method (FRM) monitors by about 9%, averaged across the sites). For all sites, sulfates and OC play dominant roles, representing 25-37% and 26-32% respectively of the total $PM_{2.5}$ mass on an annual basis. In general, sulfates play a slightly greater role in rural areas and OC slightly less, as compared to urban areas. Elemental carbon and nitrates also contribute relatively more mass in urban areas, with unaccounted mass being more important in rural areas. On an absolute basis, sulfate levels are quite uniform across the state (about 4 $\mu g/m^3$), with the exception of Whiteface Mountain, which is both farthest from the source region and also at a higher elevation. Mass values for carbonaceous and nitrate aerosols are higher in urban areas.

Seasonal variation in particle levels and relative composition are also apparent as indicated in Figure 1-4 and Figure 1-5. Summertime concentrations (June through August) are universally greater than those in other seasons, as a result of elevated sulfate and OC levels. Summertime sulfate concentrations are twice as great as wintertime levels in urban areas, and three times as great at the two rural sites. The summer increases in sulfates and OC overwhelm the corresponding inverse nitrate trend, which reaches its lowest levels during the warm summertime period. Elemental carbon shows little seasonal variation, with the exception of slightly higher measured values in the more urban locales in winter.

Figure 1-2. Multiyear Average Speciated Fine Particle Concentrations at NYS STN Sites.

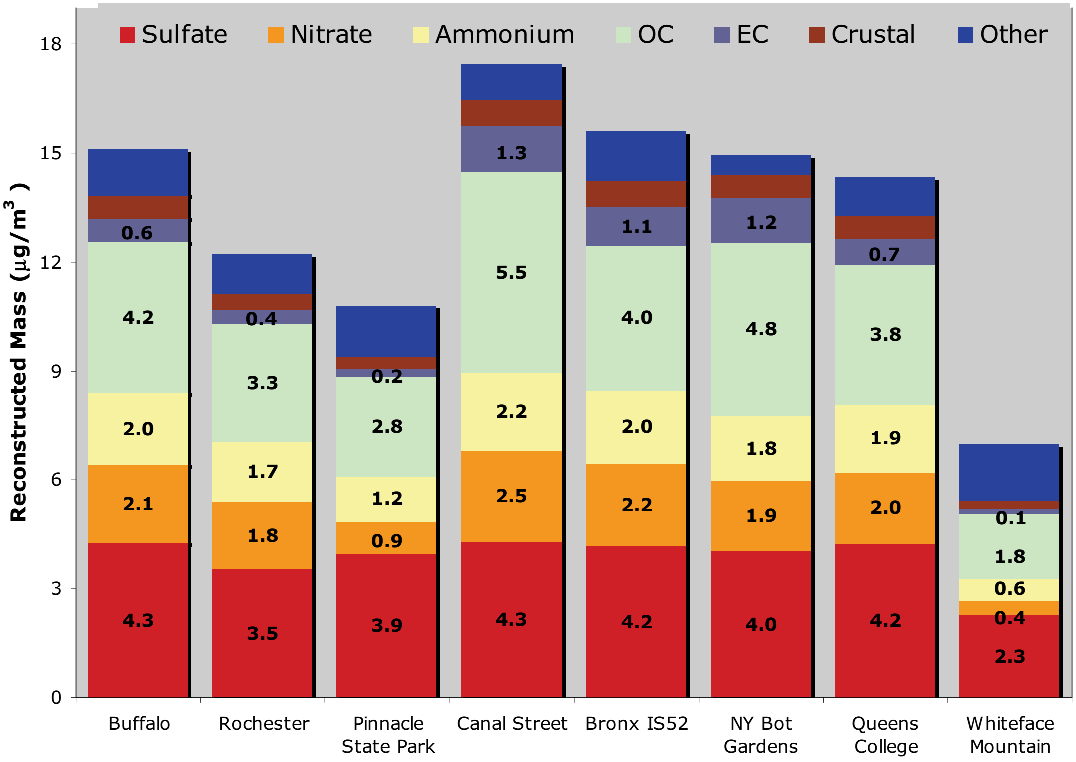


Figure 1-3. Overall Average Speciated Fine Particle Contributions at NYS STN Sites.

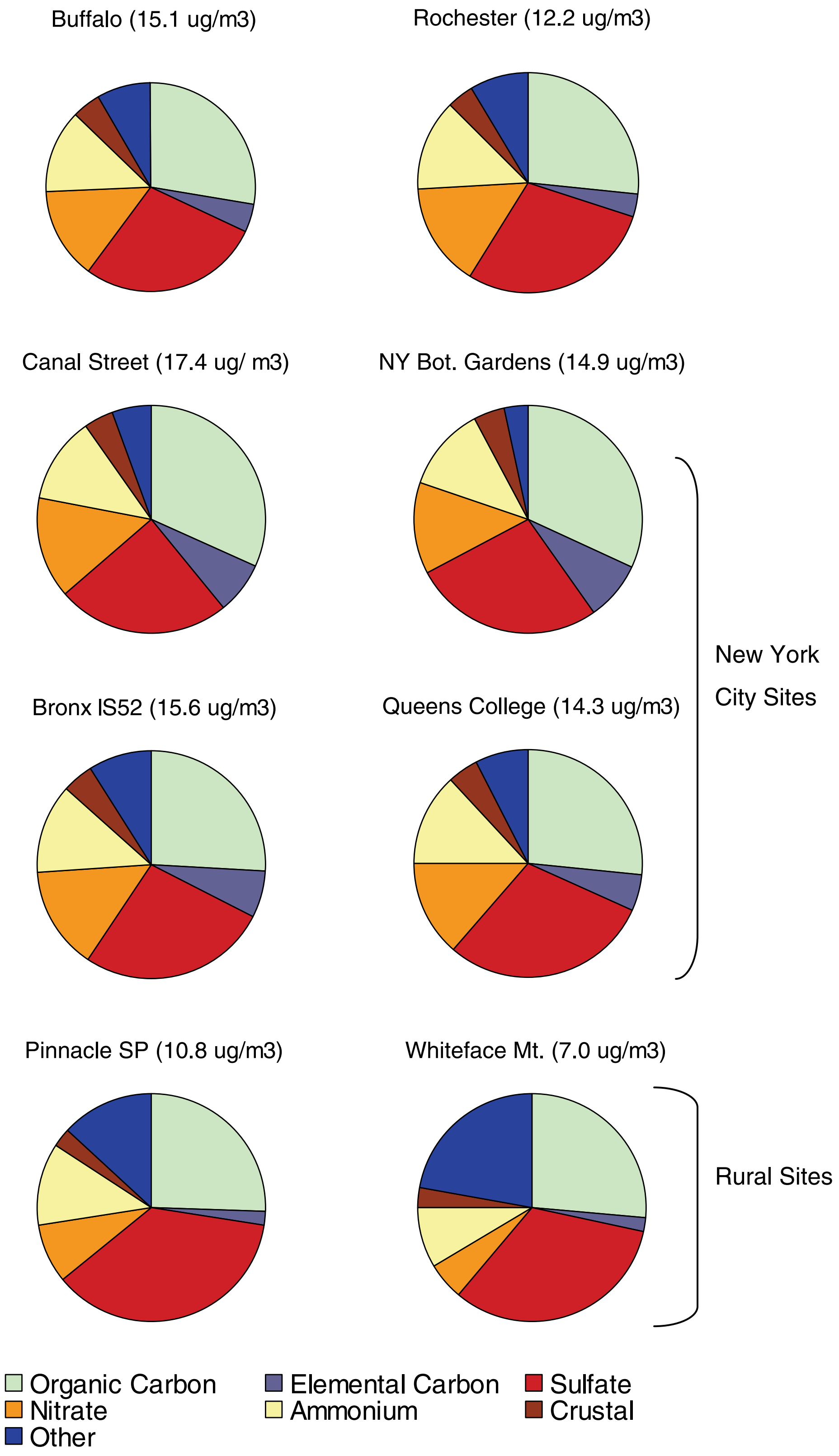


Figure 1-4. Seasonal Speciated $PM_{2.5}$ Concentrations ($\mu g/m^3$) for NYC STN Sites.

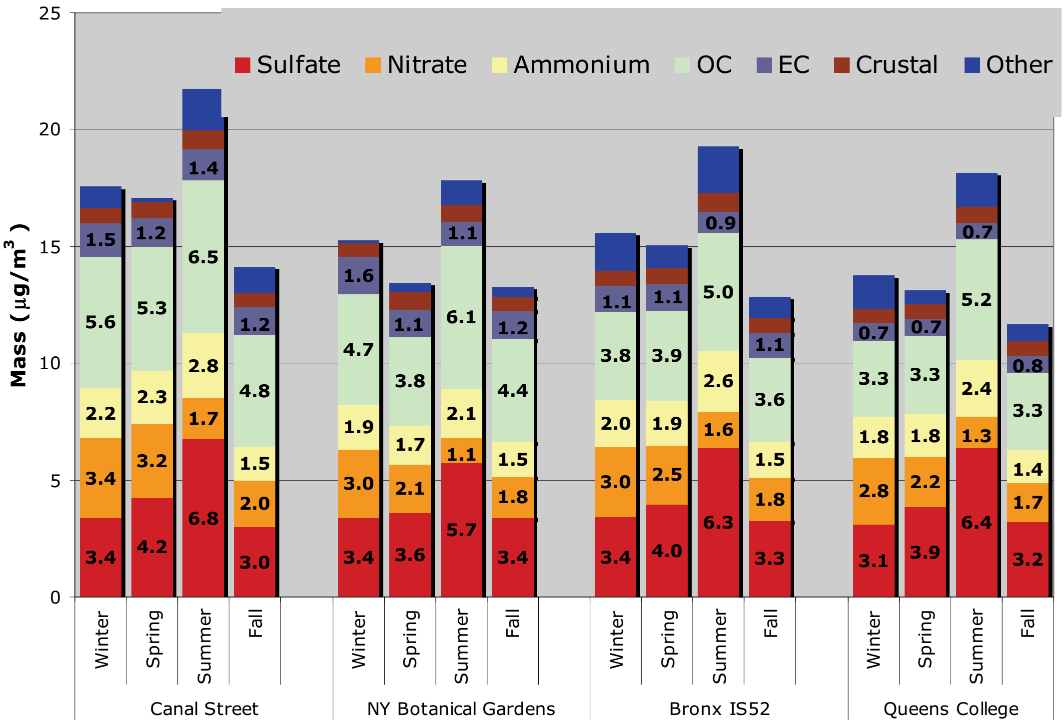
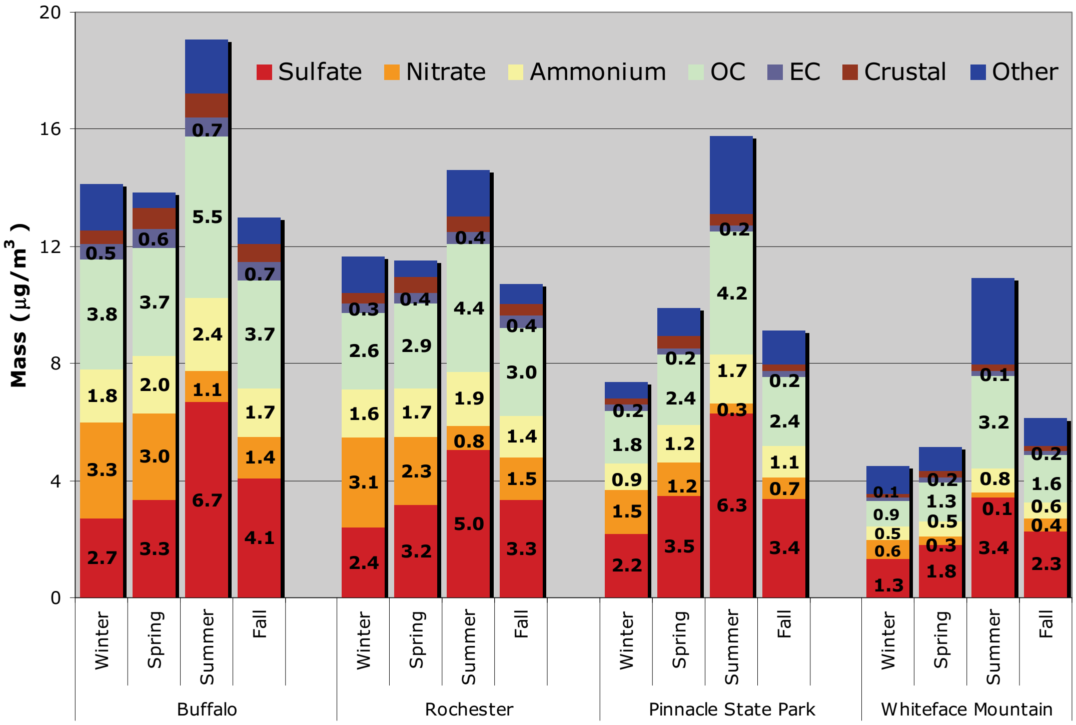


Figure 1-5. Seasonal Speciated $PM_{2.5}$ Concentrations ($\mu g/m^3$) for Upstate STN Sites.



Monthly Variation of Carbonaceous PM$_{2.5}$. Carbonaceous aerosol concentrations were investigated in more depth, looking at monthly variation and in addition, comparing the STN measured results to those from IMPROVE. For OC, values were self-normalized to the site-specific annual average and plotted to show monthly variations. The normalization permits all sites to be plotted together while showing site-specific variation. The absolute mass scale would partially mask variation of sites with the lowest overall mass measurements. The summertime maximum is readily apparent in Figure 1-6, being most pronounced for the two rural sites. The most urbanized sites show indications of bimodality, with a secondary wintertime peak. The summertime values are likely driven by secondary production of aerosols given the substantially increased photochemical activity in the warmer months. For the urban sites in winter, presumably cooler temperatures that promote condensation and suppressed vertical mixing drive observed increases. Normalized OC levels for some NESCAUM-area rural IMPROVE sites show similar results, with levels increasing in May, peaking in July, and returning to lower levels in September (not pictured).

Figure 1-6. Monthly Averaged Normalized OC Trends for NYS STN Sites.

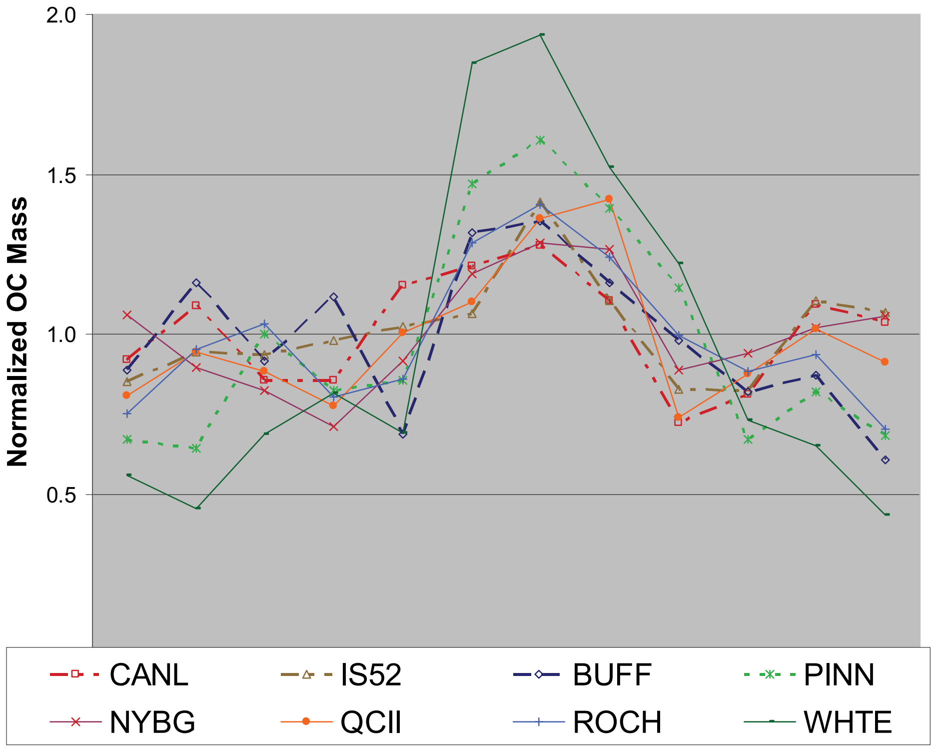


Figure 1-7. Monthly Average EC Trends for NYS STN Sites.

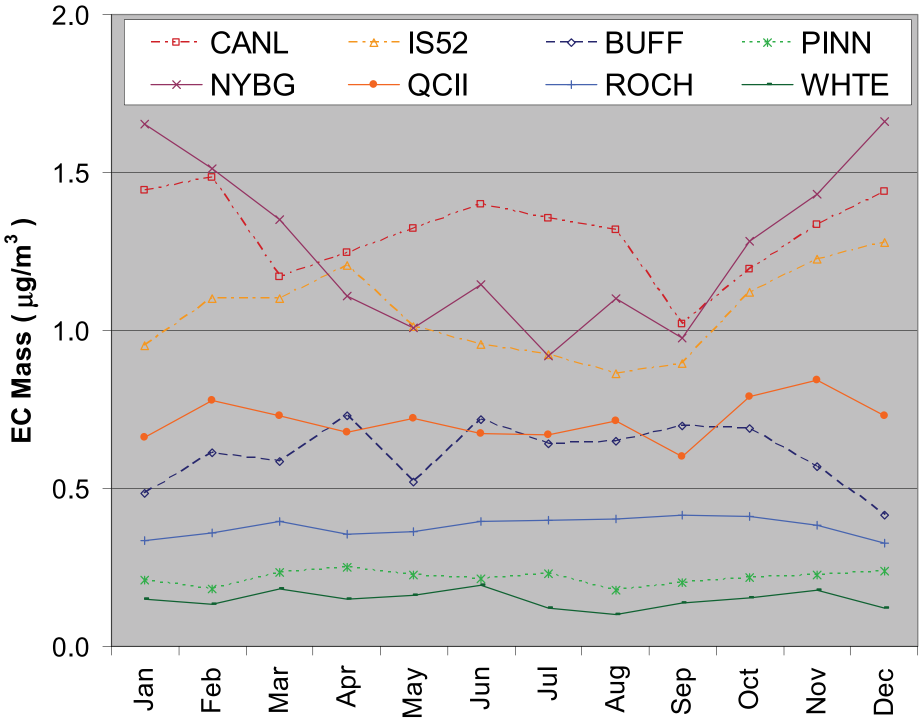
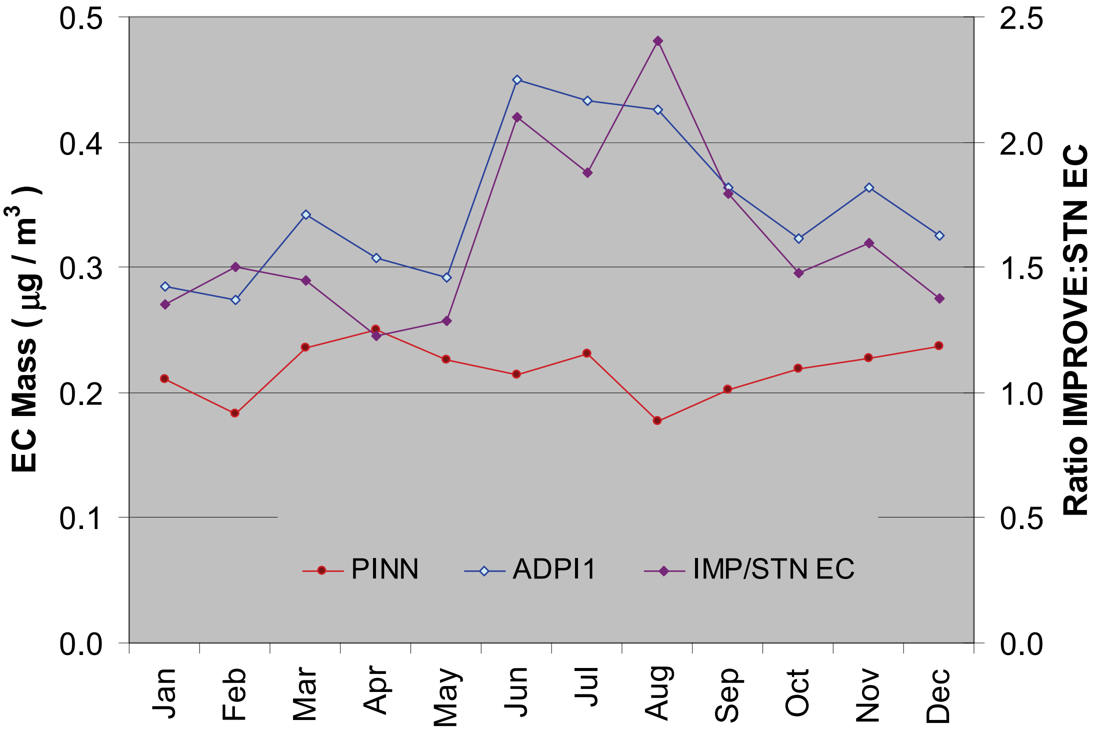


Figure 1-8. Comparison of EC Measured at rural monitors located at Addison Pinnacle State Park by STN (PINN) and IMPROVE (ADPI) and the ratio of IMP EC to STN EC.



The EC concentrations fail to vary much for most STN sites, although the most urbanized areas reveal wintertime peaks as depicted in Figure 1-7. Curiously, this behavior does not compare well with IMPROVE results for EC. To investigate these differences, the data from collocated STN and IMPROVE monitors at Pinnacle State Park were reviewed. The analysis of these data indicate no obvious seasonal trend in EC levels as measured by STN, while the IMPROVE summer-to-winter ratio equals nearly 1.5, as can be seen in Figure 1-8. EC values from IMPROVE average about 60% greater than STN values for EC at this site, while TC is reported about 10% lower by IMPROVE. Although scattered, plots of the IMPROVE EC/STN EC ratio show a marked increase in the summertime months. Given the substantial increase in OC predicted by both networks during the summer, IMPROVE likely classifies some pyrolyzed carbon as EC, while STN records it as OC. Although the "true" EC measurement remains unknown, a seasonal increase in the summertime would require that source strengths or source mixtures impacting the site have changed.

EC Tracer Method. The monthly carbon data can also be used to assess the relative contribution of primary and secondary aerosols using the ratio of OC to EC (Turpin and Huntzicker, 1995, Cabada et al. 2004, Yu et al. 2004). The STN OC/EC ratio plots shown in Figure 1-9 give fairly stable values within each site for all months except May through September. Peak values are observed in July, when photochemical oxidation should be maximized. An additional trend is noted between the most urbanized to rural sites, with the OC/EC ratio least at the monitoring locations most directly impacted by fresh emissions. Even in wintertime, the OC/EC ratio for sites far removed from direct emissions are as high or higher than summertime ratios at urban sites, indicating air masses have aged, or undergone oxidation, en route to the remote areas. This behavior could also indicate an impact from wood burning in the remote area, which has a high OC/EC signature. Table 1-5 provides a site-specific cold-warm season estimate of primary versus secondary organic aerosols, using EC as a tracer of primary emissions and assuming an average primary OC/EC emission ratio of 3.7.[9] In the cooler months, SOAs represent about half of the organic aerosols in the non-New York City sites. The average of warmer months is as high as 80% secondary aerosols by this estimate in the rural areas, with a range of 15-40% predicted for the New York City area. In July, the method estimates even higher SOA percentages.

The July estimate compares well with the 40% SOA estimate determined based on VOC and OH data from July 2002 (Tang 2006, Demerjian and Tang 2005). That research also estimates a small (~10%) wintertime

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[9] This ratio represents the average wintertime OC/EC ratio for data from the four NYC area STN sites and is applied to the non-NYC sites. For the four NYC monitors, site-specific ratios were used, since these may better reflect the primary emissions impacting those locations. An urban composite may be appropriate for sites removed from direct emissions. Note this approach differs from others that estimate primary OC by regression, which can suffer from errors due to uncertainty and interdependency in both EC and OC measurements (Saylor et al. 2006). Here, the OC/EC ratio is a composite of many different primary emission sources: wood combustion or meat cooking may have a very high OC/EC ratio while diesel emissions or residential combustion of fuel oil would be expected to have a much lower than average OC/EC ratio. Seasonal variations in these sources would therefore affect the primary OC/EC ratio. Contrasting to this work, hourly data from Flushing, New York, indicate a primary OC/EC ratio of 1.3 during wintertime (Venkatachari et al. 2006), which could be due to increased wintertime use of fuel oil for heating.

secondary OC component based on limited measurements in February 2004. Employing an ordinary least-squares regression of OC and EC data for the urban STN data set yields a comparable 5-10% contribution of SOA in the wintertime and 25-45% in summertime (Cabada et al. 2004). A similar analysis of semicontinuous data from IS52 predicted 26-35 % SOA in winter and 50-57% in summertime (personal communication, NY DEC). This regression method was used as a quick and easy estimate, despite the uncertainties in OC and EC (Saylor et al. 2006). By this approach, the primary OC/EC ratio multiplier is a third lower (2.5) with a constant of 0.9 (e.g., Primary OC = 2.5 * EC + 0.9). The constant represents primary OC sources that do not have an appreciable EC contribution, or sources with substantially higher OC/EC primary emission ratios.

The method does offer an estimate for secondary organic aerosols but is limited by the assumption that emission ratios are constant throughout the year. In reality, the relative contributions of different carbon emission sources varies, which will affect the primary OC to EC emission ratios. The filter data themselves show this occurs on a day to day basis as well, given the scatter of OC regressed with EC. Nonetheless, broad seasonal averages provide useful information on the relative contribution of SOA to total OC mass.

Figure 1-9. Monthly Average OC/EC Ratio for NYS STN Sites.

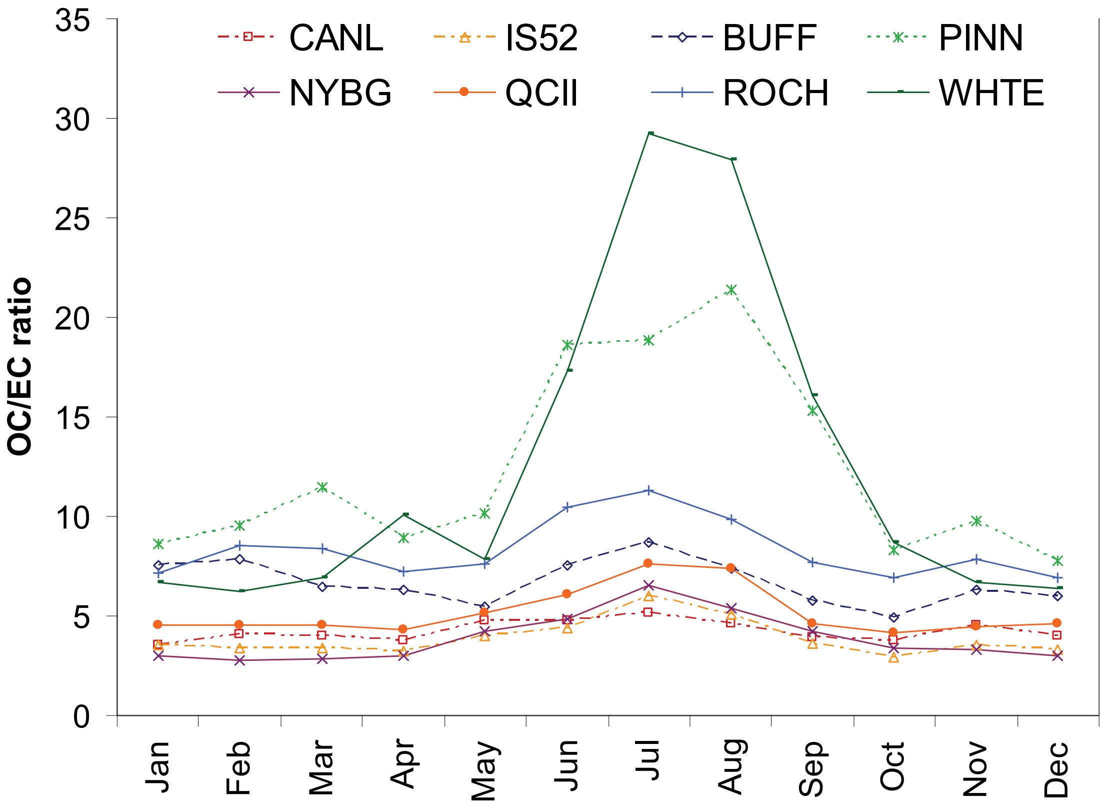


Table 1-5. Estimated Percentage of Organic $PM_{2.5}$ of Secondary Origin.

Site	Cooler Months	Warmer Months	Site	Cooler Months	Warmer Months
Buffalo	42%	47%	Pinnacle State Park	60%	78%
Canal St.	0%	15%	Queens College	0%	29%
IS52	0%	27%	Rochester	51%	61%
NY Botanical Gardens	0%	39%	Whiteface	50%	80%

Cooler months include October, November, December, January, February, March and April.

Weekday-weekend Analysis. The carbon data can be parsed to compare differences between weekdays and weekends. Differences indicate anthropogenically driven emission changes and can be interpreted as a lower bound of anthropogenic contribution. Often weekday-weekend variations are attributed to differences in mobile source contributions. Such differences were demonstrated in New York City using diurnal profiles of black carbon data collected with aethalometers, which showed rush hour peaks in BC (Rattigan et al. 2005).

Using the STN filter data, annual weekend and weekday average of OC and EC were computed along with a confidence limit based on the data variability. (Limits were based on standard deviation and sample number, but are only guidelines of data distribution as the distribution is not normal). Plots of these data for EC (Figure 1-10) and OC (Figure 1-11) for the eight sites show substantial weekday-weekend differences measured in urban areas, with virtually no difference at the two rural sites, Pinnacle State Park (PINN) and Whiteface Mountain (WHTE). Average urban weekday EC concentrations ranged from 25-60% higher than weekend concentrations. Very minor differences were observed for OC. This implies local mobile sources contribute more substantially to the overall EC burden than they do to the OC.

Seasonal averages of weekday-weekend EC data exhibit similar behavior. Figure 1-12 plots typical seasonal results for three sites. The Canal Street shows the widest weekday-weekend contrast while Pinnacle SP shows little seasonal variation. The Queens site shows higher weekday EC levels for all seasons, although not to the extent seen at Canal Street. The relatively decreased weekday-weekend differences at Canal Street in wintertime may indicate increased contribution from domestic heating sources.

Figure 1-10. Weekday and Weekend Averages for Elemental Carbon From STN Monitors for data from start of measurement period (site specific) through November 2005.

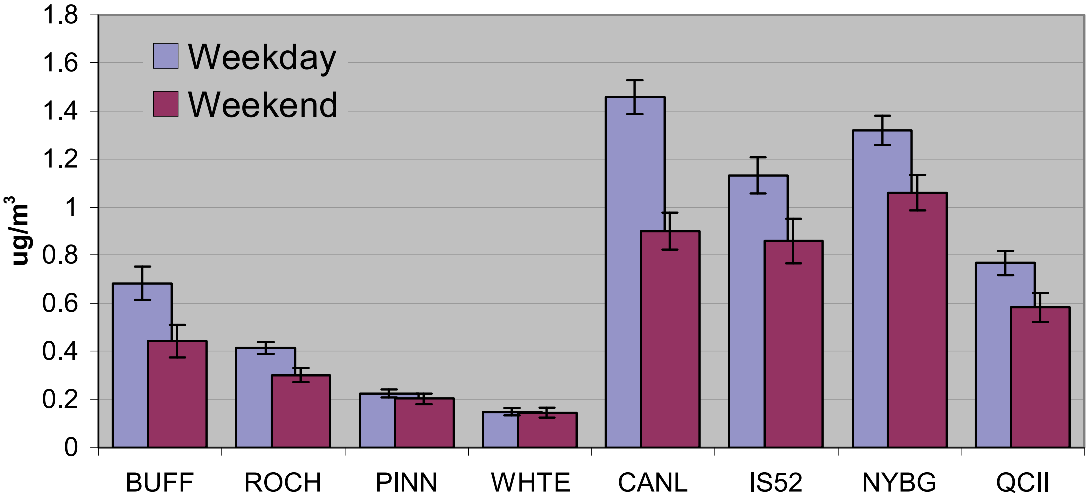
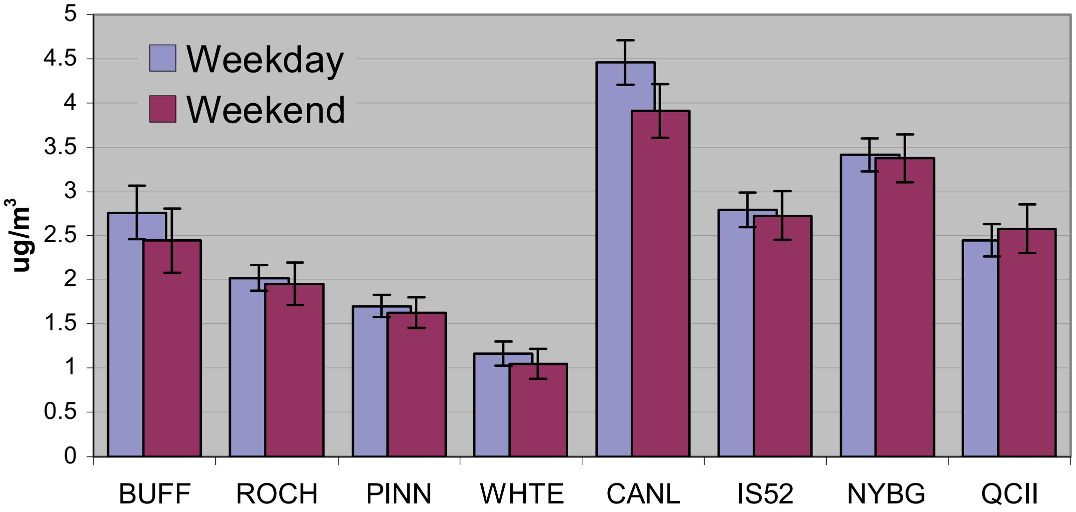
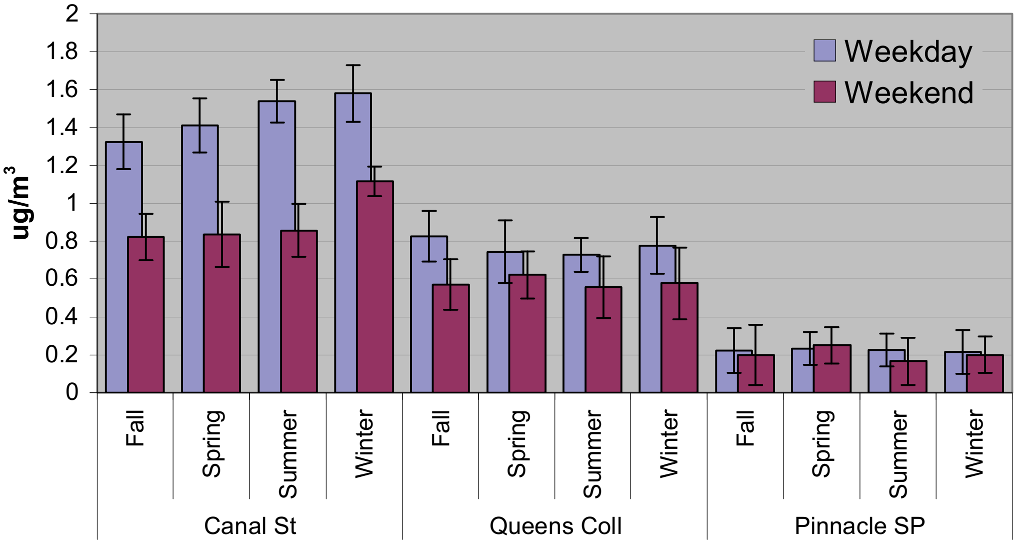


Figure 1-11. Weekday and Weekend Averages for Organic Carbon From STN Monitors for data through November 2005.



Local versus Regional Analysis. Within-site weekday-weekend differences provide one approach to assess local pollution influences. Similarly, between-site comparisons can yield information on the relative strength of local versus regional sources. Such estimates help guide the focus of control strategies. The STN data provide an opportunity to evaluate intra-urban and urban-rural pollution differences for the metropolitan NYC area. Intra-urban differences should represent the lower bound of very local source influence, where local implies a scale of up to ten miles and regional a scale of hundreds of miles. Rural monitors are generally sited to represent a broad region and should not reflect very local pollution influences. As such, they can provide a first-order estimate of regional-scale pollution. This assumption holds for primary pollutants such as EC, but it may not be as definitive for OC, where a significant secondary component may exist. Nonetheless, an isolated site should give a reasonable estimate of regional pollution levels. Note that the comparative analyses presented below may be biased by differences in sampler types. This observation is most likely a factor for semivolatile species like OC, but should be less of an issue for EC, sulfates and nitrate (collected on a nylon filter).

Figure 1-12 Seasonal Weekday-Weekend Differences for Elemental Carbon



In an attempt to provide the best possible comparison, data from the four NYC sites and the regional site, Pinnacle State Park, were reduced to a common set of sample days, which totaled 63 to 85 data points per season, depending on the pollutant. Seasonal and annual averages reported in Figure 1-2, Figure 1-3, Figure 1-4, and Figure 1-5 relied on all available data. Comparisons across sites are not truly one-to-one for that data set since different time frames and days are represented; given a sufficiently large data record, however, relative differences should be reasonably representative of true differences.

Table 1-6 summarizes the comparative results of matched sampled data. Intra-urban values were determined from the difference between the city monitor maximum and minimum, divided by the maximum. This represents a conservative estimate of the local contribution to observed levels at the most-polluted urban site[10]. Urban-rural ranges reflect the range of lower and upper bounds of local contribution as determined by the difference between the average (seasonal or annual) urban and rural concentrations, divided by the urban site concentration (i.e., four calculations, one for each urban-rural pair, were conducted, and the reported range equals the minimum and maximum of these four calculations). As noted,

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[10] Note that differences in the sampling instrumentation flow rates may affect the results, especially for organic carbon measurements, which are most impacted by filter face velocity.

this second calculation may not truly reflect local contributions for a number of reasons. Although the rural site represents a regional signature, it may not always represent the same regional signature that impacts New York City. Given the general west-to-east circulation patterns, however, the rural site used in this analysis likely provides a reasonable estimate. There may be some periods of more southerly winds during which the New York City area would be impacted significantly by upwind urban areas (e.g., Philadelphia, Baltimore, and Washington, D.C.) that do not influence pollution observed at Pinnacle State Park, and correspondingly, Pinnacle may be impacted more directly by some urban areas than the NYC region (e.g. Pittsburgh, Erie, PA). Nonetheless, the trajectory analysis for OC and EC in this work, and for sulfates from regional haze efforts (NESCAUM, 2006b), all show westerly transport associated with the worst pollution in NYC.

Table 1-6. Estimate of Local Source Impact in New York City for Major $PM_{2.5}$ Constituents From Intra-urban NYC and Urban NYC-Rural (Pinnacle State Park) Differences.

Pollutant	Comparison	Winter	Spring	Summer	Fall	Annual
Sulfates	Intra-Urban	5%	15%	10%	5%	10%
	Urban-Rural	30-35%	10-25%	0-10%	-15 – (-5%)	5-15%
Nitrates	Intra-Urban	15%	35%	30%	25%	25%
	Urban-Rural	50-55%	50-65%	75-80%	55-65%	55-65%
Organic Carbon	Intra-Urban	40%	35%	30%	35%	35%
	Urban-Rural	40-65%	30-55%	10-35%	25-50%	25-50%
Elemental Carbon	Intra-Urban	50%	40%	50%	40%	45%
	Urban-Rural	70-85%	65-80%	70-85%	70-80%	70-85%

The analysis displays substantial differences among pollutants, with sulfates and EC representing the extremes of regional and local pollutants, respectively. Intra-urban sulfate differences are on the order of 10%. Urban-rural sulfate differences reach a maximum of 35% during wintertime. Increased residential fuel use in wintertime likely drives the seasonal behavior. Similarly high transported sulfate fractions were reported by other researchers (NARSTO 2003; Lall and Thurston 2005, 2006; Qin et al. in press). Substantially lower transported sulfate in Queens of ~50% was predicted by a different approach that relied on back-trajectory and residence-time analysis (Dutkiewicz et al. 2004). The same study, using the same trajectory apportionment method, determined only 60% of sulfate at Pinnacle State Park was regional in nature, which seems quite low for an area not expected to have substantial local sulfur dioxide emissions.

On the other extreme, most of the urban EC measured may be of local origin. A factor of two difference exists between urban sites (Canal and Queens College), while as much as 85% of the EC could be local when relying on the rural measurement to represent the regional EC component. When Sterling Forest, just north of the metropolitan area, was used as the background site, similar estimates were derived (Lall and Thurston 2005, 2006). This strong local component for EC seems reasonable given the high traffic density and other combustion activities in highly urbanized areas.

Nitrates and OC behave somewhere between the two extremes of sulfates and EC. Up to 35% of nitrate could be locally generated as determined by intra-urban differences. Greater estimates arise from urban-rural differences, especially in summertime, when three-fourths of the pollution may be local. Again, this seems reasonable given the high NO_X emission density in urban areas and the strong temperature dependence of nitrates that might require high nitric acid concentrations for aerosol formation. Wintertime nitrate stability (and 'transportability) is also evidenced by the five-fold increase of nitrate observed at Pinnacle as compared to summertime levels. The estimates from this analyisis slightly exceed that derived using Chester, New Jersey, as a background site (Qin et al. in press).

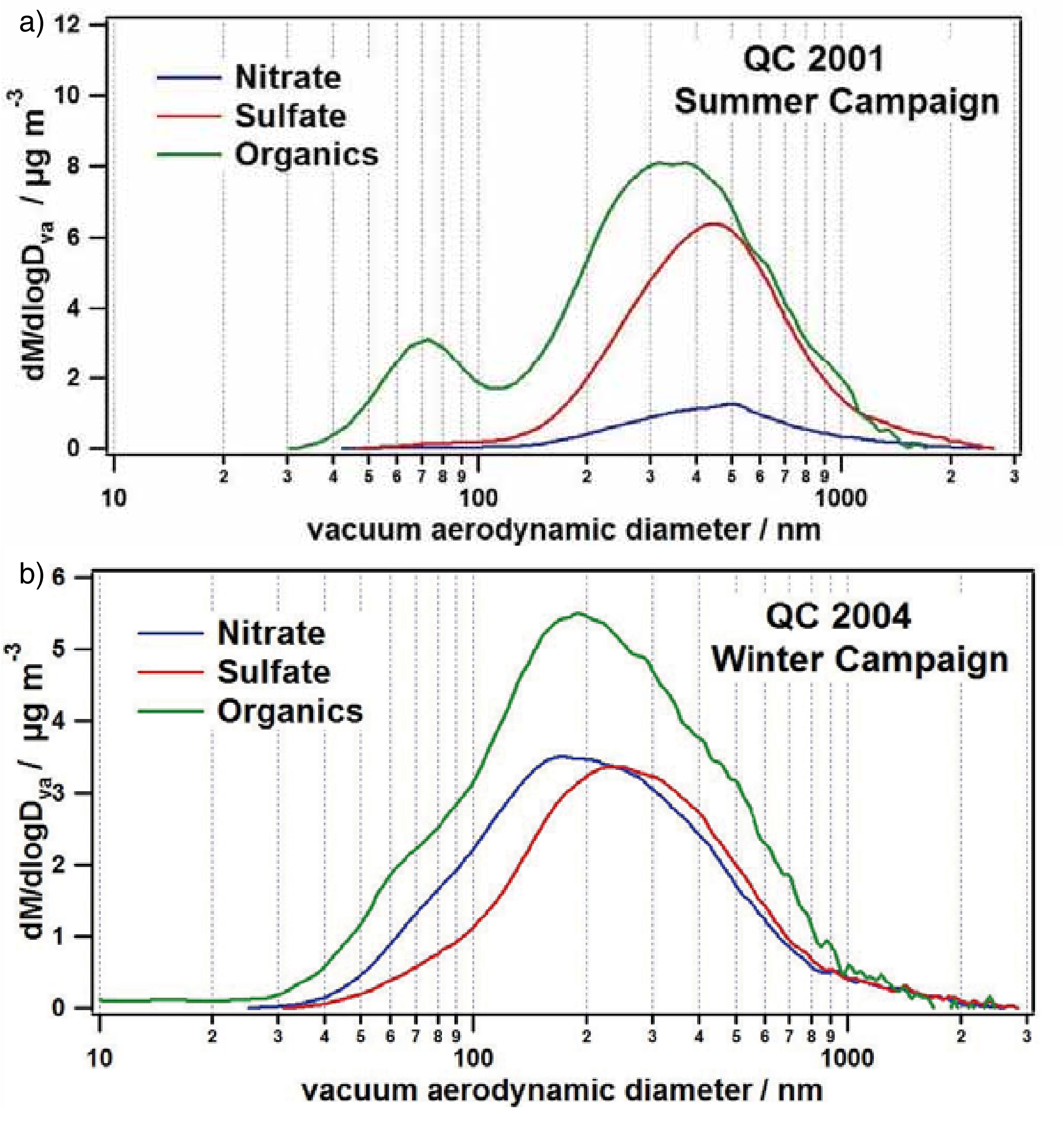
The relative local fraction of OC varies seasonally as predicted by this analysis. Intra-urban differences predict a 30-40% local OC component for summer and winter. The range for urban-rural differences reaches as high as 65% local OC in winter, but this analysis predicts a wide 10-35% local urban contribution in summer. The wintertime estimate may be more accurate than the summertime, in part due to differences in the underlying source strengths of SOA precursors and their spatial differences. This metric may underestimate local urban contributions if a large fraction of the OC measured at the rural site comes from biogenic sources that may not be as prominent in urban locales. Nonetheless, similar seasonality and local urban source strength of OC was predicted by Lall and Thurston (2005, 2006).

To some extent, one might predict the local-regional tendencies based on the relative primary and secondary production of various pollutants superimposed on their emission densities. Spatial contributions of primary pollutants such as EC should directly reflect the source strengths. Secondary aerosols such as sulfates and nitrates may not form near their emission sources, depending on the atmospheric conditions. OC, with its dual primary and secondary aspects, poses the most complicated species behavior to predict. More complete understanding will require improved monitoring and analytical techniques, such as those tested in the field through the U.S. EPA's Supersite program.

New York State Supersite

This section presents results from the New York State supersite, one of seven established through funding from the U.S. EPA to foster research on aerosol processes. An overview of the supersite program and results from other supersites is presented later in the section titled "Fine Particulate Matter Supersite Results." Most of the research conducted as part of this study occurred in Queens, although two rural sites also provided supplemental measurements. The three main objectives of research included: (1) an assessment of the temporal and spatial distribution of $PM_{2.5}$, aerosol properties, and associated gas-phase pollutants; (2) tests of PM emission control technologies; and (3) developments of new measurement technology. Applications of AMS provided the most exciting findings that expand beyond the carbon component of $PM_{2.5}$. A description of the overall program goals and findings from the other six centers follows after the discussion of aerosol studies conducted in and around New York City.

Figure 1-13a-b. Aerosol Mass Spectrometer average PM composition as a function of size measurements for the Queens College 2001 (a) summer and (b) winter field intensive studies.

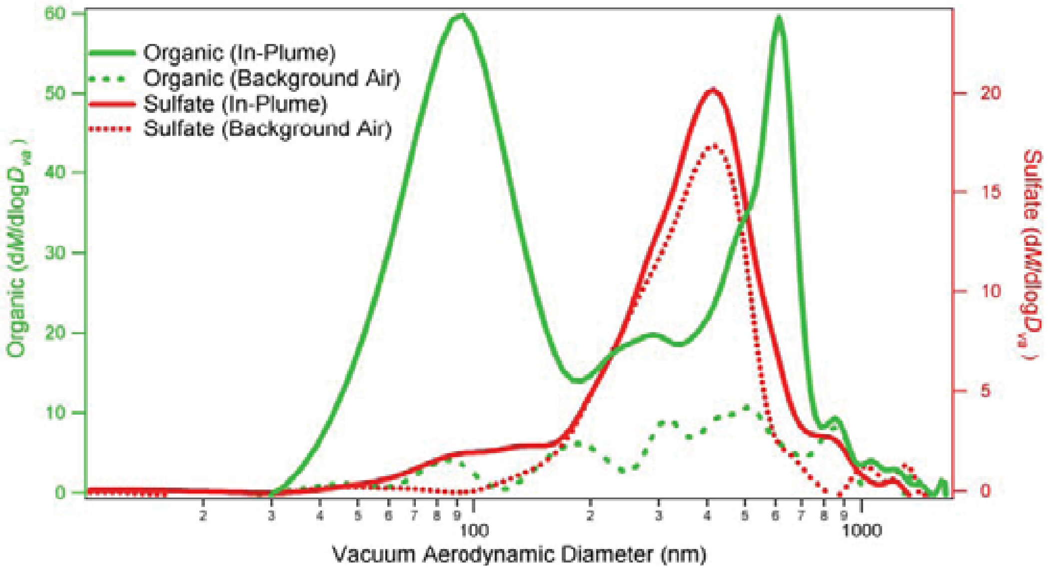


Source: Demerjian et al. (2006)

Data analyses conducted as part of the supersite in Queens found carbonaceous aerosol contributed 36% of $PM_{2.5}$ on an annual basis, with OC/EC splits of 28% OC to 9% EC in winter and 34% OC to 6% EC in summer (Demerjian 2005, Demerjian et al. 2006). As observed in previous analyses, OC levels increased in importance during the summer due to secondary processes, and EC played a larger role during colder months when inversions occurred more frequently. Two similar estimates (40% of total OC) of SOA production during summertime in New York City were obtained from the AMS OOA fragments and direct calculations based on reaction kinetics of measured OH and VOCs (Tang 2006).

The AMS also provided size information for OC. Most of the summertime OC mass (84%) was found in the accumulation mode with the rest of the mass found in particles smaller than 120 nm (Drewnick et al. 2004a). The distribution is pictured in Figure 1-13a. The small mode was greatest during the morning rush hour, indicative of fresh, traffic-derived emissions, as shown by vehicle chase studies (Canagaratna et al. 2004; Drewnick et al. 2004b). Figure 1-14 highlights the predominant small organic mode monitored in exhaust plumes. Researchers in Rochester, New York, found similar elevated numbers of small particles associated with local traffic sources (Jeong et al. 2004). At the rural sites, only the larger of the two modes was present in summer. This may indicate that these aerosols are not from direct emissions; rather they represent somewhat aged aerosols that have had time to grow. The primary size mode was observed to shift toward smaller particles in the wintertime, centered about 200 nm instead of the 350-400 nm observed in summer. The small mode was no longer apparent, likely being merged into the shifting primary mode as shown in Figure 1-13b. The possibility exists, however, that the small mode shifted to a size below the collection capability (30 nm) for the AMS (Weimer et al. 2005, 2006).

Figure 1-14. Typical diesel PM organic and sulfate exhaust plume and background measurements averaged over a chase event during summer 2001 field campaign in Queens, NY. The solid lines correspond to distributions averaged over time periods when the mobile laboratory was sampling the vehicle's exhaust plume while the dotted lines correspond to size distribution averages for ambient measurement conditions.



Source: Demerjian et al. (2006)

Other interesting AMS results included different diurnal patterns for the OOA and HOA fragment patterns (Drewnick et al. 2004a). The HOA showed morning increases that represent fresh traffic emissions. OOA levels peaked in the midafternoon, indicating their likely source as photochemical processes. Tang (2006) predicts similar bevavior relying on measured OH radical and VOC concentrations. During wintertime, strong correlations between AMS organics and NO_x indicated a substantial fraction of measured OC is derived from local sources (Weimer et al. 2006). Poor correlations between OC and EC during summer may be indicative of varying primary carbonaceous PM sources or variability in SOA production.

Vehicle chase data revealed significantly greater emission variability than what might be expected based on dynamometer testing (Canagaratna et al. 2004). This points to limitations in the current emissions inventories, which may not reliably reflect true emission variability from mobile sources. This also implies other inventory sources may require refined measurements to better characterize both annual-scale and shorter-term emission rates.

Semicontinuous measurements of OC and EC were also made, as were aethalometer measurements of BC. These showed substantial spatial and temporal variation, indicating urban EC measurements may only represent a very small area of at most a few kilometers (Venkatachari et al. 2006b). Morning rush hour peaks of EC appear consistently (Rattigan et al. 2004; Venkatachari et al. 2006a), as seen with HOA AMS data. Schwab et al. (2004) determined from 24-hour filter data that EC monitored at two rural areas did not show similar local source impacts. Regression of AMS data against OC obtained by other semicontinuous and traditional filter-based methods predicts a mass multiplicative factor ranging from 1.24-1.61 to account for the non-C elements associated with OC (Venkatachari et al. 2006a).

Speciation of Organics for Apportionment of PM2.5 (SOAP)

In anticipation of their likely nonattainment status for $PM_{2.5}$, New York, New Jersey, and Connecticut developed a monitoring plan for organic aerosols in the greater New York City area in the early 2000s. The research was designed to quantify the various regional and local emission source contributions for fine OC in the New York City metropolitan area relying on a chemical mass balance approach. This project, Speciation of Organics for Apportionment of $PM_{2.5}$ (SOAP), was designed to supplement and benefit from the research activities of the New York City supersite. Sampling occurred from May 2002 through May 2003 at four sites, one upwind of the city (Chester, New Jersey), one downwind (Westport, Connecticut) and two in the expected nonattainment area (Elizabeth, New Jersey, and Queens College, New York). This initial monitoring effort has been enhanced by similar, more recent measurements at Pinnacle State Park and IS52, over the course of 18 months spanning 2005 to 2007. Preliminary findings point toward the importance of cooking operations and mobile sources to OC levels in the region.

Samples were taken every third day, to correspond with the STN monitoring schedule. Each site was collocated with an STN monitor, which provides data on the suite of other fine PM constituents to supplement the mass balance analysis. Given the identified concerns with carbon blank issues, an independent analysis was conducted on the SOAP filters for EC/OC quantification. Organic speciation analysis yielded concentration data for a suite of molecular species, including n-alkanes, hopanes, steranes, PAHs, n-alkanoic acids, dicarboxylic acids, and aromatic carboxylic acids. Identification of compounds derived from wood combustion is underway (e.g., levoglucosan).

The monitoring campaign produced over 400 filter measurements, which were combined into seasonal composites of six to ten filters to improve detection capability. The approach maximized comparisons across the sites and seasons to illustrate spatial and temporal variation in OC molecular composition. Generally, molecular concentrations were greatest at the two metropolitan monitors, and also during the cooler months (Mazurek 2006). The n-alkanoic acids also showed increased summertime levels, indicating increased source strength for these species. Levels of hopanes, signifying oil combustion, were greatest at Elizabeth, as might be expected given its location near major roadways and toll plazas.

The detection of substantial levels of n-alkanoic acids, in particular the C16 and C18 homologs, may indicate the contribution of cooking operations using seed oils. The observation that peak levels occur in wintertime reinforces the belief of an anthropogenic source over something of biogenic origin. Measurements of these species have also been detected in motor vehicle emissions and wood smoke plumes. To determine the mix of sources responsible, source apportionment using improved source characterization for the New York City metropolitan area will be required.

The carbon preference index (CPI) presents a different approach for molecular marker data analysis. Naturally occurring species show the predominance of even n-alkanoic homologs. The lower molecular weight (C10-C18) species reflect levels of lipids (plant and animal) while the larger ones (C20-C30) indicate contributions from plant leaf surface waxes. The calculated CPIs reveal a substantial biological component for these compounds. Curiously, the large molecular weight compounds show maximum values in the most urban areas, while little seasonality is observed at the more rural monitoring sites.

The source apportionment results from this monitoring campaign should provide the best indication of the sources impacting levels of fine organic aerosol in the New York City area. The derived contributions will improve understanding and help point to appropriate control strategies to meet obligations for air quality improvement under the Clean Air Act. The receptor-oriented approach yields an independent assessment to augment existing emission inventory determinations of major carbon sources in the region.

Source Apportionment Studies in New York State

Source apportionment analyses provide estimates of source contributions to the $PM_{2.5}$ levels. They can highlight differences between sites or demonstrate deficiencies in emission inventories. In addition to source assignment, the analyses may also allocate the relative importance of local versus transported emissions. Work of this nature generates valuable insights for understanding the origins of pollution and presents an important tool for air quality planners.

Many researchers have conducted source apportionment analysis for New York City and the surrounding region, primarily through use of PMF in combination with trajectory analysis to identify source factors and regions (Liu et al. 2003; Ito et al. 2004; Kim and Hopke 2004a; Zhou et al. 2004a; Qin et al. in press). The studies most frequently rely on STN and IMPROVE data sets, although some more highly time-resolved data have been employed (Li et al. 2004; Lall and Thurston 2005). Generally speaking, the analyses identify a handful of source factors that commonly include motor vehicles (i.e., traffic), residual oil combustion, secondary aerosols (sulfates), and crustal material. Comparing across sites and studies, however, the derived profiles may appear to be very different, despite being attributed to a common source type.

One study identified only those four factors as contributing substantially to the $PM_{2.5}$ mass (Ito et al. 2004). Two years of STN data from three sites in New York City were analyzed by absolute PCA and PMF. Of the four factors, only the traffic factor contained appreciable OC and EC concentrations. This factor contributed 16-39% of the monitored mass, with PCA associating 39% and 36% of the mass with traffic at IS52 and Queens College respectively, while the PMF mass apportionment was 22% and 34% for the same sites. The factor of two difference between methods at IS52 shows the wide range of uncertainty associated with this type of analysis. Ito et al. (2004) observe a wide range of temporal correlation among factors between sites, suggesting a varying level of local and regional source impacts and recommending that caution be used when estimating population exposures.

Another study used data collected from Potsdam and Stockton, New York, over the course of two ozone seasons (May to September) in 2000 and 2001 (Liu et al. 2003; Zhou et al. 2004a). Daily samples were analyzed for $PM_{2.5}$ mass, its major and elemental constituents, and PAHs. PMF was applied, resulting in the identification of six common sources. From a carbon aerosol perspective, wood smoke represents the source of most relevance and was associated with 9% and 2% of the measured mass at Potsdam and Stockton, respectively. The profile was deemed to be wood smoke given the levels of EC, potassium, and the PAH retene, which are all associated with wood combustion. Two trajectory-based metrics, potential source contribution function (PSCF) (Liu et al. 2003; Zhou et al. 2004a) and residence time weighted concentration (RTWC) (Zhou et al. 2004a), were used to determine likely source areas for the wood smoke factor. Both methods suggested the Great Lakes region as being a possible source, originating from either residential wood combustion or forest fires. Interestingly, the zinc smelter factor identified at each location had nearly as strong a retene signature as did the smoke factor; likewise the predicted source region overlapped considerably for the smelter and woodsmoke.

One recent approach using the temperature-resolved fractions of OC and EC has shown promise in helping to better resolve two different mobile source types, spark-ignition and diesel engines (Kim and Hopke 2004a, b; 2005). PMF modeling results from Brigantine, New Jersey, relied on its long data record, including 10 years of data (1992-2001). Eleven factors were derived, with three containing much of the OC (secondary sulfates, gasoline vehicles, diesel emissions) and two with appreciable EC levels (incinerator, oil combustion). The two mobile source components represented 13% and 3% of the total $PM_{2.5}$ (totaling nearly 2 $\mu g/m^3$), while secondary sulfates accounted for another 6% from a combination of S and OC. The other two sources were each less than 2% of the total mass. Substantial seasonal but little weekday-weekend variation was observed for the mobile source factors, indicating the sources were not of local origin. Wind roses revealed the likely transport corridor is from the metropolitan New York City region. The sulfate-OC source appeared to originate from transported air masses from two forested regions, with PSCF highlighting forested areas in the southeastern U.S. and Hudson Bay in Canada. This factor was also observed from Atlanta and Washington, D.C. data (Kim and Hopke 2005).

Although PMF using thermal differenction of OC and EC separated two different mobile source factors (assigned to spark-ignition and diesel emissions), the relative contribution from the carbon temperature fractions was somewhat different across sites, especially for the diesel factor. Comparisons of the derived factors and source tests also indicated significant deviations from expectations. The source profiles of gasoline vehicles had relatively more OC1 and OC2 than OC3 and OC4, which was opposite of the weighting in PMF factors. For diesel, the source profile EC was dominated by the second fraction, whereas the PMF factor was mainly the first EC bin. Possible explanations include atmospheric processes changing the relative makeup of the carbon aerosols, and interactions between the carbon emissions and other ambient species affecting the temperature evolution of carbon relative to a pure mobile-source emitted particle profile.

Li et al. (2004) report source apportionment (PMF2) of data collected at Queens College in July 2001, including 6-hourly particle-into-liquid sampler (PILS) ion results and AMS data. Neither of these metrics measures EC. PMF2 did, however, identify three factors with appreciable OC portions: motor vehicle, secondary nitrate, and oil combustion. The motor vehicle factor displayed strong diurnal behavior with a morning peak likely associated with rush-hour traffic; its total mass contribution (mostly OC) was only 3%. The nitrate and oil combustion sources together represented about 15% of the mass, although much of that was noncarbonaceous material.

Two final studies attempted to apportion the local and regional source contributions to the metropolitan NYC area (Lall and Thurston 2005; Lall and Thurston 2006, Qin et al. in press). For their analysis, Lall and Thurston (2005, 2006) relied on semicontinuous carbon and $PM_{2.5}$ data collected in 2001 from two sites (Manhattan and Sterling Forest). Using PMF, six factors were identified, including traffic (strong OC and EC association) and residual oil (weak EC association). The analysis implies that nearly half of the $PM_{2.5}$ mass in New York City on an annual basis is derived from transport. Transported sulfates range from 80% to 90%, while not more than 15% of EC was attributed to transported emissions. The local OC component varied widely, from 74% to 33% for winter and summer respectively.

These results compare favorably to those obtained by Qin et al. (in press), who analyzed nearly four years (2000-2003) of STN data collected from four New York City metropolitan sites and one upwind site (Chester, New Jersey). The analysis predicted a larger transported fraction for $PM_{2.5}$ of up to 82%, a similar transported sulfate fraction (93%), and a significant transported nitrate fraction (54-65%). PMF factors for motor vehicles were derived, representing 8-22% of the total mass for spark-ignition and 3-15% for diesel vehicles across the five sites; most of the mass was carbonaceous. New analyses of motor vehicular emissions conducted by Shah et al. (2004) were cited as cautionary for apportionment purposes, as PMF factor assignment relied in part on relative OC and EC contributions. Their new evidence demonstrates at low speeds the OC/EC emission ratio for diesel vehicles resembles that of spark-ignition vehicles. The study makes several interesting points regarding discrepancies between existing PM inventories and derived PMF source factors. First, the inventories indicate substantial emissions from fugitive dust and wood combustion. Although a crustal component is derived, it does not represent an appreciable mass fraction. No wood factor was predicted by PMF. The study reveals that source apportionment results may be useful for emission inventory assessment.

Other Studies

Several other studies have investigated carbonaceous aerosols in NYS (NESCAUM 1999; Rattigan et al. 2005; Maciejczyk et al. 2004; Pun et al. 2002; Seigneur et al. 2003). The earliest study measured OC and EC at two sites for the 1995 calendar year (the sites were located in Rochester and Brockport) (NESCAUM 1999). The data confirm that OC and EC concentrations are higher at the urban site as compared to the background site, which indicates the importance of local sources in urban areas. In addition, the study revealed that OC and sulfate levels were comparable. The two Bronx studies monitored EC (using an aethalometer, which reports BC), generally finding weekday-weekend differences with early morning peaks (Rattigan et al. 2005; Maciejczyk et al. 2004). Considerable spatial variation was noted, with the urban core having the greatest EC levels relative to a background site in New Jersey and a site in Rochester, New York, a smaller urbanized area. Comparisons between aethalometer measurements and 24-hour filter EC showed similar seasonal trends, although absolute mass determinations varied by as much as a factor of two in the summer. The differences may arise from variations in the absorptivity of BC due to seasonal aerosol compositional changes (Rattigan et al. 2005).

Two regional modeling studies were conducted in the Northeast using CMAQ at 4-km grid spacing. The first investigated the fate and transport of diesel particles, whose emissions were dominated by on-road diesel and construction equipment as reported in the National Emission Trends inventory (Seigneur et al. 2003). Model gradients for a summertime episode showed urban concentrations of diesel PM as much as an order of magnitude higher than rural regions. Comparing measured EC and a fraction of diesel PM that is assumed to be EC revealed that much of the predicted urban EC could be accounted for by diesel PM. EC measurements in rural areas were substantially greater than predicted diesel EC, implying that other sources of EC impact rural areas. The second study estimated the contribution of biogenic emissions to $PM_{2.5}$ in the Northeast (Pun et al. 2002b). The simulation predicted that 12% of $PM_{2.5}$ originated from biogenic precursors. A model run without emissions of anthropogenic precursor estimated very little production of biogenic aerosols in the absence of anthropogenic pollution.

Trajectory Analysis

Trajectory analyses are often used to trace the path of polluted air masses prior to their arrival at a given receptor site. Such analyses, by linking downwind measurements of ambient air quality with specific geographic areas upwind, can assist in understanding the contribution of transported emissions from potential source regions on high- and low-pollution days. Understandably, inherent uncertainties exist with trajectory analysis. The inability to pinpoint the exact emission location(s) along the trajectory path represents one important limitation. In addition, the accuracy of any individual back trajectory calculation for a single observation or episode may be compromised by inherent limitations in the underlying Lagrangian trajectory models, which tend to become less accurate as the calculation progresses further back in time. Fortunately, a variety of techniques are available to mitigate these uncertainties and enhance confidence in the results obtained using trajectory analysis. These include techniques for triangulating results across multiple sites, ensemble techniques that combine the results of large numbers of back trajectories, clustering algorithms that group similar trajectories based on their spatial characteristics, and techniques for combining trajectory analyses with source apportionment models. All of these strategies can be useful in improving and refining traditional trajectory analyses.

Two approaches were employed to combine air mass transport information with monitored carbon aerosol levels in NYC to provide insights into the origin of carbonaceous pollution in the metropolitan area. The first technique explores a means for improving the accuracy of individual trajectories by grouping meteorologically similar back-trajectories into trajectory “clusters” and examining the relationship between the transport pathways defined by these clusters and downwind air quality observations. The second method, termed incremental probability (Poirot et al. 2001), stratifies pollution and compares the upwind residence-time probability of back trajectories associated with high pollution levels to the upwind residence-time probability of all trajectories. These and other techniques were employed by the Mid-Atlantic/Northeast Visibility Union (MANE-VU) in their regional haze contribution assessment report that documents likely source regions contributing to visibility degradation in Class 1 areas (NESCAUM 2006b).

All back trajectories used in MANE-VU's work were computed with the hybrid single particle Lagrangian integrated trajectory (HY-SPLIT) model (Draxler 1997). Back trajectories were calculated eight times per day at a starting height of 500 m above ground level using meteorological wind fields for the five-year period from 2000 through 2004. Meteorological data from the National Oceanic and Atmospheric Administration (NOAA) Air Resources Laboratory (ARL) archives were analyzed for use with the HY-SPLIT model. Wind fields from the Eta Data Assimilation System (EDAS), which cover North America with an 80 km spatial resolution and are based on three-hourly variational analyses, were used in model calculations.

Table 1-7. Queens College Cluster Characteristics: Pollution Level and Seasonal Composition.

	Median Pollution (µg/m^3)				Meteorological Season			
Cluster	PM$_{2.5}$	Sulfate	Organic Carbon	Elemental Carbon	winter	spring	summer	fall
1	**17.90**	**6.02**	**3.01**	**0.77**	7%	25%	46%	22%
2	16.20	5.07	2.23	0.66	13%	22%	38%	26%
3	*10.80*	***2.59***	2.49	0.67	11%	15%	41%	33%
4	16.75	4.45	2.95	0.75	18%	24%	36%	22%
5	***9.10***	*2.76*	***1.76***	***0.57***	8%	36%	21%	36%
6	16.20	4.48	2.74	0.85	28%	22%	30%	21%
7	12.25	*2.94*	2.17	*0.57*	23%	27%	21%	29%
8	12.30	*2.93*	2.42	0.64	42%	13%	21%	24%
9	**18.00**	**5.46**	**3.01**	**0.78**	19%	39%	22%	20%
10	***8.20***	***2.09***	***1.72***	***0.50***	12%	26%	28%	34%

Underlined (red) pollution values indicate "high" levels, whereas italicized (blue) levels show relatively "low" levels. Bold text shows the two highest (lowest) values for each pollutant.

The clustering algorithm employed, patterns in atmospheric transport history (PATH) (Moody et al. 1998; Dorling 1992), compares the three-dimensional position in space and time of large numbers of back trajectories. The method assigns each trajectory to a group with similar spatial characteristics. As calculated, these grouped trajectories or clusters represent the predominant pathways by which air masses arrive at the receptor site. Likewise, each cluster has a "central trajectory" that characterizes the average three-dimensional transport. The calculated central trajectories define the major transport patterns into New York City; they were compared to the five-year trajectory database to develop the associated upwind probabilities for air mass history as described in NESCAUM's contribution report (2006).

Ten predominant transport patterns to Queens College were defined. Median pollution levels corresponding to the members of each cluster were calculated to distinguish the pathways and regions linked to poor air quality from those with good air quality; results are shown in Table 1-7. The associated central trajectory and cluster residence times are plotted in Figure 1-15. From the table and figure, it can be seen that poor air quality is associated with air masses that spend substantial time in the highly populated regions of the East Coast (New York City, Philadelphia, Baltimore) and originate in the central part of the country (clusters 1 and 9). Better air quality is observed from air masses that originate over the Atlantic Ocean or arrive at Queens College from the northwest (clusters 5 and 10). Table 1-7 also shows a tendency exists for clusters populated by summertime trajectories to have higher pollution levels.

The cluster method identifies upwind geographic regions based primarily on meteorological characteristics and then determines the associated pollution levels. An alternative scheme, incremental probability (IP), reverses the order by grouping trajectories with similar pollution levels first, and then investigating the air mass origins. Two probability fields are developed: one derived from the pollution-selected subset of trajectories and the other from all trajectories. Their difference represents a measure of the likelihood that a given source region contributes more than "average" to high concentrations of a particular pollutant at a downwind receptor site (Poirot et al. 2001). To determine locations that are less likely to contribute to poor air quality at a given receptor site, trajectories with low pollution levels are selected.

For this work, two IP calculations were performed using data from the Queens College STN site. Figure 1-16a and b display the IP fields for organic and elemental carbon, respectively. Areas shaded in red show the region from which air associated with higher pollution levels originates. Blue regions illustrate areas less likely to be the source of high pollution levels. For both OC and EC, the most probable upwind source regions for pollution are concentrated in the East's urbanized region of New York, Philadelphia, and Baltimore. Air masses originate from the Midwest or Southeast, arriving at the region via the Midwest. High OC pollution levels originate from a slightly broader area than do high EC levels. The regions to the north and east of New York appear less likely as source regions of high levels of carbonaceous aerosol. This work agrees with other recent research that traces EC monitored in rural New York State to air masses transported from the Midwest (Khan et al. 2006).

Both of these qualitative trajectory methods associate meteorological pathways with downwind air quality. Whether sorted by common meteorology or common pollution levels, these techniques indicate a strong association between high carbonaceous aerosol levels and atmospheric transport from the west. The most important drawback of these methods is their inability to pinpoint where along the path the emission source(s) may lie.

Figure 1-15. 2000-04 Queens College Trajectory Clusters. Upper Panel Shows Central Trajectory; Lower Panel Cluster Residence Time.

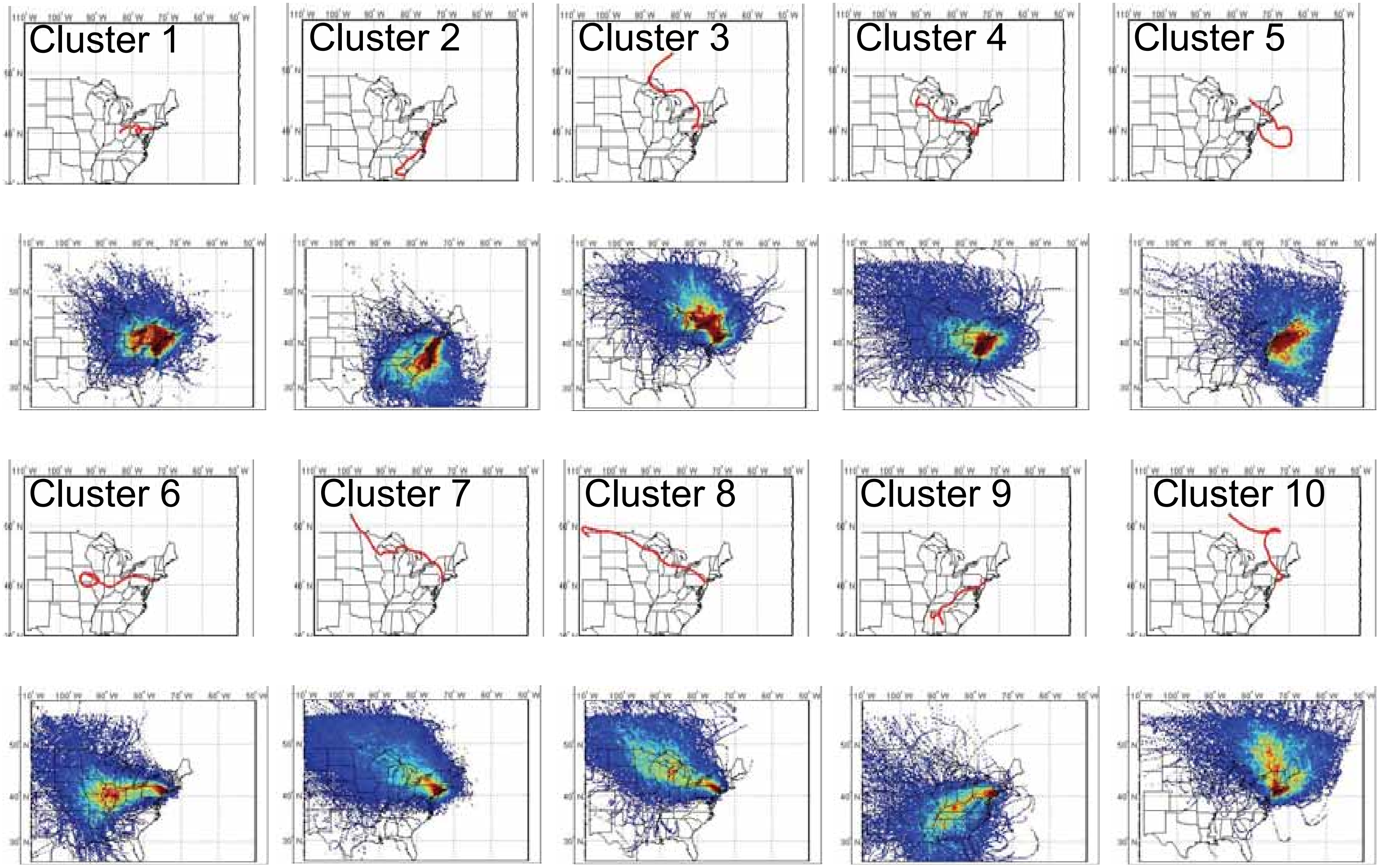
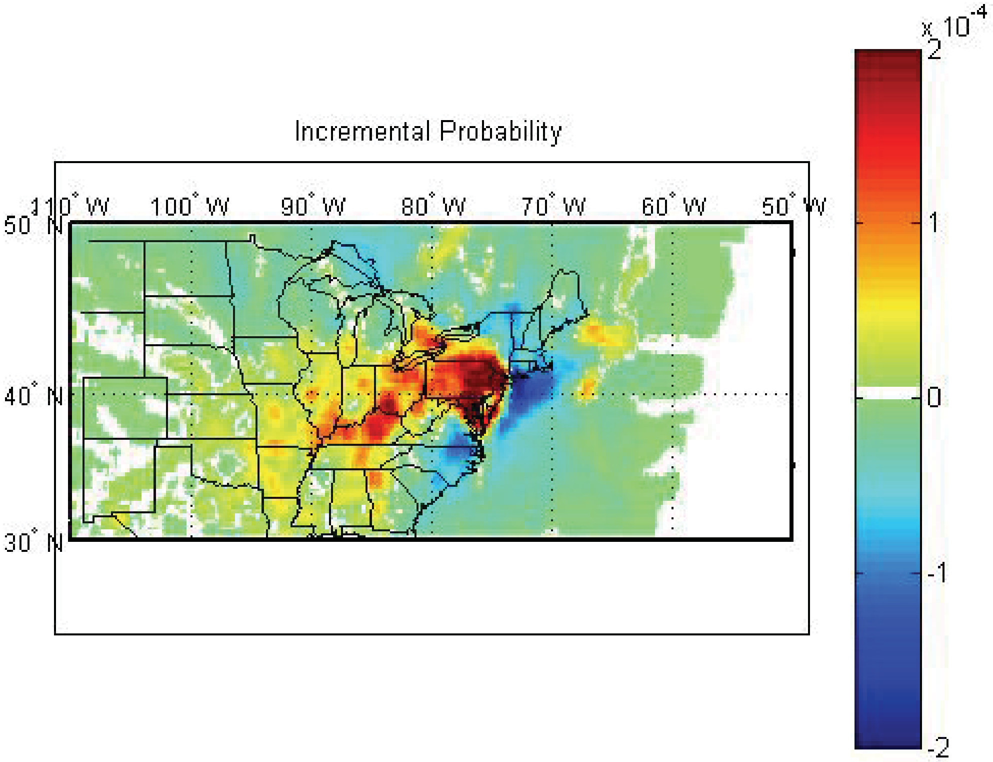
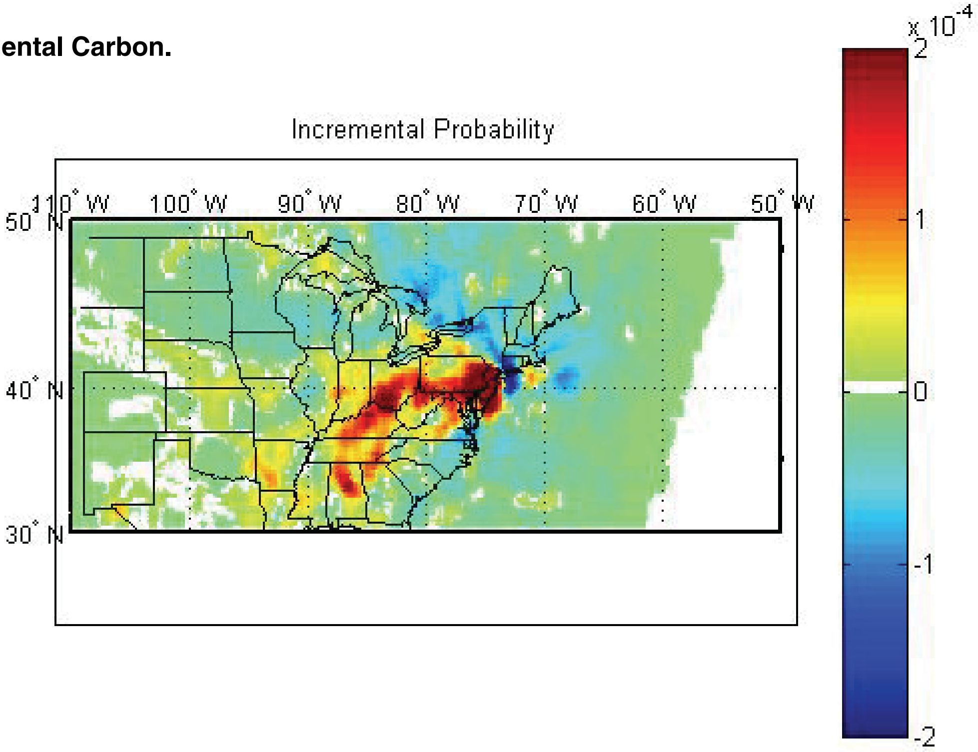


Figure 1-16a-b. Incremental Probabilities.
a. Organic Carbon.



b. Elemental Carbon.



MANE-VU Rural Aerosol Intensive Network

The MANE-VU Regional Planning Organization established the Rural Aerosol Intensive Network (RAIN), a network of three enhanced monitoring stations (Piney Run, Maryland, Mohawk Mountain., Connecticut, and McFarland Hill, Maine) designed to provide continuous data on the concentration, composition, and visibility impacts of $PM_{2.5}$. Data gathered complement the IMPROVE monitoring network and will further the understanding of the nature of visibility degradation in the Northeast. NESCAUM, working with state air quality agencies, has coordinated the deployment of the network. An initial data set has been analyzed with results reported in a technical memorandum (NESCAUM 2006b).

The RAIN sites collect highly time-resolved (1-2 hour) information on aerosol composition (both carbonaceous aerosol and sulfate), as well as meteorological and optical property measurements. These data help to characterize the sources and formation of regional aerosols. MANE-VU chose the RAIN sites to represent a broad region, removed from direct local source impacts. Each site is elevated relative to its surroundings in an attempt to capture transported aerosols. The sites are distributed along a southwest-to-northeast line that traces the predominant summertime wind patterns. The sites were chosen in part to permit an assessment of the progression of air masses as they traverse the central U.S. through to the Atlantic. The short-term nature of the measurements expands the ability to monitor atmospheric changes that cannot be captured by the 24-hour integrated IMPROVE measurements relied upon by the Regional Haze Program.

A number of important findings have emerged from preliminary data analyses of fine carbonaceous aerosol data from routine monitoring networks. Review of the decades-long available IMPROVE organic carbon data in the eastern U.S. shows a wide range of daily concentrations that peak in mid to late summer. STN data obtained and analyzed for this NYSERDA report display similar seasonal behavior. The long-term data set from the Acadia IMPROVE site revealed daily peak organic carbon values correlating moderately well with mean ambient temperature but correlating poorly with the wood smoke tracer “non-soil potassium,” which is measured potassium not associated with soil particles.

During ozone episodes in Maine and Connecticut, two-hour averaged ozone and OC data moved in tandem, with late afternoon maxima. This suggests similarities between the conditions that favor ozone and organic aerosol formation—likely related to atmospheric conditions (sunny and warm) and precursor availability. Interestingly, collocated monitored isoprene levels measured by auto-GC for Photochemical Assesment Monitoring Stationgs (PAMS) were also linked to ozone and OC. Research has shown biogenic emissions play a substantial role in ozone formation in rural areas, and they may indicate the existence of a strong biogenic component to organic aerosol chemistry in the summer. Measured BTEX compounds (benzene, toluene, and xylenes) did not follow the same patterns as isoprene and OC.

Over the next several years, the hourly data generated from these sites should vastly expand the understanding of bulk carbonaceous aerosol behavior in rural atmospheres. Source apportionment tools applied to the real-time OC/EC mass, sulfate mass, and meteorological parameters may better define the source contributions to regional aerosols. In addition, this time-resolved data should prove useful for validation of modeled aerosols. In total, information derived from the RAIN network will be invaluable to air quality planners as they work toward achieving fine particle reductions to benefit health and environmental welfare.

Fine Particulate Matter Supersite Results

In 1997, the U.S. EPA issued a new $PM_{2.5}$ NAAQS based on the best available information at the time. The agency recognized, however, that considerable gaps existed in the understanding of the sources, atmospheric processes, and health effects of fine particles. To address these knowledge deficiencies, the US EPA initiated a two phase ambient monitoring research effort known as the Particulate Matter Supersites program. The initial phase was conducted at two research sites (Atlanta and Fresno). The second phase was conducted at seven urban areas with relatively high fine particle levels (Baltimore, Fresno, Houston, Los Angeles, New York City, Pittsburgh, and St. Louis). The research program focused on three primary goals: (1) to measure ambient particulate matter at airsheds representative of the United States to better characterize source-receptor behavior and governing atmospheric phenomena; (2) to support health effects and exposure research through improved measurement techniques; and (3) to develop new and/or improved existing fine particle monitoring methods (Solomon and Allen 2004). The results of the supersite program were expected to bolster state efforts to prepare State Implementation Plans (SIPs) to address fine particle pollution in the U.S.

As a mass-based standard, planners first must understand the $PM_{2.5}$ composition to address the proper particle sources for attainment. Supersite results consistently show carbonaceous aerosols to be a major fraction of $PM_{2.5}$, accounting for 20-40% of the annual $PM_{2.5}$ mass (Solomon 2004). Research identified the importance of regional transport for carbon aerosols in eastern urban areas, while recognizing the significance of locally generated primary carbonaceous pollution. The contribution of secondary aerosol production was noted to be greatest during the summertime, peaking along with atmosphere oxidants such as ozone. Specific findings on carbon-based particulates from each research center follow.

Baltimore was chosen as a supersite in part due to its location along the eastern U.S. transportation corridor that begins in Washington, D.C., and extends through Boston. Both local mobile sources and transported pollution from the Midwest affect Baltimore’s air quality. To assess these pollution influences, measurements in Baltimore were focused on the short-term variability of physical and chemical properties of the ambient aerosol.

Three major measurement techniques were employed to obtain information on carbon aerosol: 3-hourly filter-PUF samples of speciated organics, hourly bulk carbon measurements from a Sunset Labs analyzer, and single-particle MS (Ondov et al. 2005). The detailed speciation measurements permit the separation and identification of local versus regional signatures. The measurements also highlight short-term source influences by using the temporal variation of unique source tracers (e.g., hopanes and n-alkanes C_{19-25} for mobile sources, odd C_{27-33} n-alkanes for plant waxes). Bulk carbon measurements were compared to 24-hour STN filter measurements and showed good agreement, although with consistently lower values (10%/20% for EC / OC). SPMS results indicated that 30% of the number of particles was carbonaceous, with spectra resembling those of combustion sources such as gasoline or diesel vehicles. During traffic-dominated PM episodes, OC and EC accounted for ~40% and ~10% of the measured mass.

Upwind of Baltimore and the eastern urban corridor, Pittsburgh and the surrounding area provides valuable insights about the transport of PM and its precursors from the Midwest to the eastern US. Pittsburgh is influenced by reasonably well-defined local, remote, and biogenic sources; this situation helps to establish and quantify source-receptor relationship. As at other supersites, a number of different measurements of carbonaceous aerosols were conducted to help characterize ambient aerosols impacting the region.

Researchers concluded that nearly 90% of the fine particle mass measured was derived from sources outside of Pittsburgh (Pandis et al. 2005). Half of the smallest particles, however, were deemed to be locally generated from transportation sources. Ultrafines were dominated by carbonaceous particles, representing half of the mass in summer and 70% in wintertime. As predicted by EC tracer-type methods, secondary carbon aerosols contribute roughly 30% of the total organic PM on an annual basis, ranging from 10% in winter to 50% in summer.

Measurements were conducted for four years at the St. Louis supersite in the Midwest, with two intensive studies conducted at nearby rural areas for comparison. In East St. Louis, IL, little seasonal variation was observed for carbonaceous aerosol (Turner 2005). On an annual basis, organic and elemental carbon species were 38% and 4% of the mass, respectively. Source apportionment showed an 18% contribution from mobile sources. Other notable contributors included meat cooking and wood smoke. Investigation of the mass factor associated with organic carbon yielded an annual average ratio of 1.8 with little seasonal variability.

Whereas results from the Baltimore and Pittsburgh may be most directly relevant to New York City, given their eastern locale, researchers based in Houston have also contributed to the developing understanding of carbon aerosol formation. The Houston area receives emissions from a wide range of pollution sources, including the typical urban anthropogenic mixture (e.g., mobile sources), biogenic sources, and a significant contribution from the petrochemical industry. Carbon aerosols made up nearly 30% of the fine particle mass on an annual basis, with peak organic carbon levels in the late summer and fall period (Allen 2005).

Source apportionment analysis using molecular markers identified primary emissions from mobile sources and meat cooking as substantial contributors to fine particle mass (representing 1/3 to 1/2 of the total). Using OC/EC ratios and isotopic composition, secondary organic aerosols were determined to be significant, especially in the summertime through September. In heavily forested areas especially, SOA formation was dominated by biogenic precursors. Analysis showed biogenic SOA formed primarily through reaction with ozone, with spatial variation directly related to terpene emission distribution. This observation leads in part to the conclusion that emission reductions in ozone precursors may indirectly lead to reduction in secondary particle formation.

Temporal variations were also displayed at the Fresno supersite, with higher carbon levels in the colder months. Diurnal patterns differed with the season, showing nighttime peaks in the winter from wood combustion and rush-hour peaks in the morning and evening during warmer weather (Watson et al. 2005b). The wintertime contribution from wood smoke was confirmed through use of the tracer, levoglucosan.

Much of the carbon aerosol work at the Fresno supersite was focused on filter sampling and comparison of results from different sampling configurations and techniques. Total carbon concentrations were shown to be equivalent within measurement uncertainty for a variety of denuded and nondenuded/backup filter-corrected sample streams. Results revealed the positive artifact of filter adsorption to be twice that of particle volatilization from a denuder-filter combination, although confirmation testing was recommended. TOT (transmittance) versus TOR (reflectance) comparisons showed transmittance predictions for EC were 30-75% lower than EC from reflectance predictions, with the lower value at common temperature evolution protocol and the higher value for a high temperature/fast analysis operation. Aethalometer BC values were found to be lower than EC by TOR and higher than EC by TOT.

Unlike Fresno, where carbon measurements concentrated on bulk aerosol properties, the Southern California supersite in the Los Angeles Basin investigated the chemical characteristics of particle number and size distribution (Sioutas and Froines 2005). Studies showed ultrafine particles were dominated by carbon, with higher wintertime levels attributed to increased condensation from motor vehicle emissions. Secondary organic aerosol formation in downwind areas was deemed primarily responsible for maximum ultrafine values found there in summer, although increased advection was also a factor. Correlations between EC, particle number, and diesel emissions were established. The dominance of motor vehicular emissions in the ultrafine mode was corroborated through concentrations of hopanes, which are molecular markers for such sources. Levoglucosan measurements were higher in the accumulation mode, along with tracers of SOA. From their in-depth research of motor vehicle emissions and their impact on ambient fine particle levels, the researchers concluded that filter technology *alone* would be insufficient to address particle emissions. Because much of the resulting PM derives from condensing vapors, removal of primary PM may exacerbate the levels of ultrafine semivolatile organic particles. Recent research on catalytic traps, however, showed effective removal of these semivolatile organics (Grose et al. 2006), implying the appropriate combination of filters, catalysts, and low-sulfur fuel could minimize the levels of ultrafine particles in the exhaust.

CONCLUSIONS

This chapter presented a state-of-the-science assessment of the major formation processes for carbonaceous aerosols and their measurement, with a focus on studies conducted in the State of New York. Laboratory experiments illustrated the complexity of the chemistry, which is governed by numerous factors. Major determinants of aerosol production include such things as the reactants (e.g., parent hydrocarbon, the oxidant(s) present), temperature, radical levels, seed aerosol characteristics and the partitioning behavior of the reaction products. Despite attempts to quantify and parameterize the various factors, current atmospheric models fail to capture the intricacies involved in SOA production. The existing tools, including source apportionment, along with rapidly developing measurement technologies such as aerosol mass spectrometry, provide an expanding source of information that will close the knowledge gaps in the coming years.

The second half of this chapter discussed data analyses specific to the State and relevant research findings from recent monitoring campaigns outside New York. The results show carbonaceous aerosols represent a substantial fraction of the $PM_{2.5}$ mass, with a large contribution from sources located within the State. Temporal and spatial differences were observed both within the New York City metropolitan area and across the State. Anticipated results from the SOAP study and supersite-related activities will continue to fill in gaps to identify the major sources of carbonaceous aerosols in the region. Based on the state of the science and current knowledge of the role of $PM_{2.5}$ carbon in New York State, the report recommends avenues for further study as detailed below.

Information Gaps and Research Needs:

Refine Emissions Inventories. Emissions inventories (EI) represent the basis for much air quality work and should be improved. They can be used directly to assess primary emissions, to provide a crucial input for chemical transport modeling, or to yield insights to help interpret derived profiles based on source apportionment of ambient data. Although current inventories may have been useful for past air quality management tasks, substantial refinements to existing emissions inventories are needed to continue the significant progress in air quality improvement (NARSTO 2005, Miller et al. 2006). Specific recommendations for the emissions inventory of carbon, especially OC and OC precursors, include:

- Update speciation profiles for mass of OC and EC with appropriate "definition" of analytical method for various source categories, including area, stationary and mobile.
- Develop accurate primary emission factors for fine PM mass (and its speciated components) for various combustion sources using methods based on "dilution tunnel technology," or other advanced techniques.
- Include tracer organic species required for receptor model analysis in source characterizations.
- Measure precursor emissions relevant for secondary aerosol production and develop methodologies to allow assessment of their secondary aerosol production potential, similar to the Incremental Aerosol Reactivity concept (Griffin et al. 1999).
- Identify and develop explicit definitions for other relevant properties to include in the inventory database for such things as source variability or quantification of analytical and other uncertainties.
- Conduct source apportionment analysis for the Speciation of Organics for Apportionment of $PM_{2.5}$ (SOAP). This will require a complete suite of major source profiles for sources of carbonaceous aerosols in New York State. This work will be conducted in collaboration with US EPA.

Enhance Chemical Transport Models. Chemical transport models represent a fundamental tool for evaluating air quality and control strategy effectiveness. These tools currently fail to capture the complexity of organic aerosol processes, and they require improvements. Potential avenues for model development follow.

- Expand models to incorporate other precursor VOC species or at least improve the mapping of species from species-specific emission inventory profiles to model species.
- Update models to address recent research results that indicate the existence of heterogeneous processes and their resultant reaction products.
- Develop new modeling approaches as needed, since complete gas-phase chemistry coupled with partitioning treatment may be unfeasible given its complexity, the computational demands (current computer technology may be inadequate), and unresolved details of the atmospheric chemistry of organic aerosols.

Expand Ambient Measurements. In the last decade, new methods have been developed for aerosol detection and quantification. Continued efforts should focus on real-time approaches with high time resolution over integrated sampling. The artifacts are likely to be less, and the data generated should be much more useful in understanding the atmospheric behavior of carbonaceous aerosols, with likely improvements in source apportionment techniques. Specific suggestions for improvement include:

- Emphasize techniques that rely on direct analysis, such as liquid chromatography-mass spectrometry or single-particle/aerosol mass spectrometry, over those that require extensive sample preparation.
- In regions with high levels of OC, conduct gas-phase measurements focused on semivolatile precursor species in addition to aerosol measurements to improve the understanding of gas-particle partitioning.
- Pursue refinements in understanding the mass associated with measured OC. Semicontinuous and real-time methods should provide greater detail in temporal variation in the associated mass factor. Sub-daily, seasonal, and spatial variations exist but are not accounted for in the analyses conducted in this report.
- Develop well-documented calibration standards. A current lack of such standards, from bulk carbon analysis down to that of individual species, prevents successful sample quantification. Calibration standards and analytical methodologies should be well-documented to allow the cross-comparison of ambient measurements from different research initiatives.
- Continue application and method refinement of semicontinuous carbonaceous aerosol monitoring. These results become more critical in light of the newly revised 24-hour NAAQS for $PM_{2.5}$. Such highly time-resolved data, especially when coupled with other similar data sets (e.g. trace gases, AMS data, and elemental particulate composition through Davis Rotating Drum Unit for Monitoring (DRUM) sampling or semi-continuous elements in aerosol system (SEAS) sampling) will provide a rich data set that will improve the scientific understanding of source variability and impact.
- Conduct expanded sampling to better define both spatial and temporal gradients in carbonaceous aerosols. The current filter-based regulatory sampling networks suggest substantial intra-urban and regional gradients exist. More measurements are required, especially in remote areas and smaller urban centers. Given limited resources, a combination of a reference site(s) and short-term satellite sites should provide more complete characterization of the temporal and spatial differences across the State.

Improve Chamber Studies. Chamber studies represent a primary mechanism for improving the scientific understanding of the atmospheric chemistry of organic aerosols. Organic aerosol composition is a complex function of many variables, including the mix of biogenic and anthropogenic hydrocarbons, humidity, oxidant/reactant levels, and temperature. Studies investigating these dependencies often occur under conditions that do not reflect ambient conditions. Areas for improvement include:

- Design studies that are relevant to ambient atmospheric conditions.
- Expand efforts to examine organic carbon precursors, both individually and collectively, to improve the understanding of real-world interactions. Many studies attempt to isolate individual

reactants or very specific reaction conditions. In real-world interactions, however, hundreds of different parent VOCs may contribute to observed particle phase organic carbon.

- Encourage the development of improved real-time analytical tools to minimize uncertainty introduced from sampling artifacts or derivatization procedures. These new instruments should be used to track short-term changes in initial reactant, intermediates, and final product levels.

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Section 2

HEALTH

INTRODUCTION

Over the past few decades, a growing body of experimental and observational evidence has implicated ambient exposure to mass-based fine particulate matter (≤2.5 µm, $PM_{2.5}$) in impairment of human health. Toxicological, clinical, and epidemiological research has centered mainly on respiratory and cardiac effects, ranging from minor irritation to exacerbation of chronic disease and even premature death. Although the primary PM National Ambient Air Quality Standards (NAAQS) are mass-based, $PM_{2.5}$ aerosols are actually a complex solid, liquid, and gaseous mixture. $PM_{2.5}$ emanates from a variety of natural and anthropogenic sources whose emissions profiles are influenced by atmospheric forces and chemical reactions. These factors affect population exposures over varying spatial and temporal scales.

Despite the diversity of $PM_{2.5}$ chemical composition across geographic areas, adverse health effects associated with $PM_{2.5}$ exposure consistently have been found in urban areas worldwide. It may be that no single characteristic of $PM_{2.5}$ is responsible for the entire spectrum of adverse health effects observed. Different agents, individually and in combination, could contribute to adverse health outcomes. In recent years, researchers have begun to focus on the carbonaceous fraction of $PM_{2.5}$. This interest stems from the unique properties of carbon particles and their presence in combustion-related source emissions found in ambient $PM_{2.5}$ in many populated areas.

The purpose of this chapter is to characterize and assess the emerging work of health-science researchers in order to describe what is known and unknown about the health risks of carbonaceous $PM_{2.5}$. The first part reviews relevant physical and chemical attributes of carbonaceous $PM_{2.5}$ and describes recent health studies. The objective of this analysis is to provide insight into the potential toxicity of varying carbonaceous $PM_{2.5}$ components. These health-related findings are assessed in the larger context of $PM_{2.5}$ health associations. Limitations of current science are also discussed. The second part of this chapter addresses potential exposure implications of carbonaceous $PM_{2.5}$ for populations in New York State. A case study focusing on mobile source emissions is presented. The chapter closes with an overview of current mass-based $PM_{2.5}$ NAAQS metrics and a discussion of the possible influence that $PM_{2.5}$ constituents could have on future standard setting. Gaps in knowledge are considered along with current thinking on research needs related to health and exposure.

HEALTH RELEVANCE OF PM COMPONENTS AND CARBONACEOUS $PM_{2.5}$

In support of the current $PM_{2.5}$ NAAQS, numerous epidemiological studies using mass-based $PM_{2.5}$ air pollution data have found associations between short- and long-term exposure to $PM_{2.5}$ and adverse health outcomes. These include lung function decrements, exacerbation of lung disease, respiratory and cardiac mortality, cancer, and developmental and immunological effects (U.S. EPA 2005). Studies relying upon mass-based $PM_{2.5}$ monitoring data, however, provide little causal understanding of the potential components that lead to specific adverse health effects. Recognizing this limitation in understanding, the National Research Council (NRC) Committee on Research Priorities for Airborne Particulate Matter identified 10 highest-priority research recommendations, including an assessment of the role of physiochemical characteristics of PM in eliciting adverse health effects (NRC 1998). This knowledge could facilitate the development of more efficient measures to control source-related constituents of $PM_{2.5}$ and determine appropriate regulatory control strategies, such as state implementation plans and federal ambient air quality standards.

Table 2-1. $PM_{2.5}$ Physical and Chemical Properties of Interest for Health Effects.

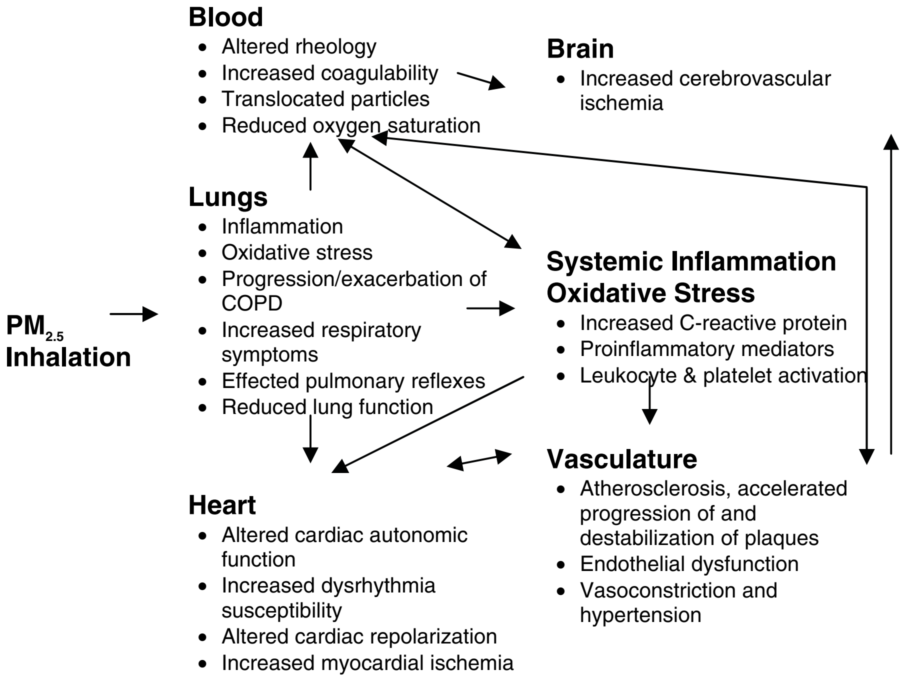
Health-relevant $PM_{2.5}$ physical characteristics and chemical components
Particle surface area
Particle number
Particle mass
Particle size distribution
Surface chemistry
Surface charge
Biological components (e.g., pollen, microbes)
Ions (sulfates, nitrates, ammonium)
Strong acidity (H^+)
Transition metals (water soluble, bioavailable, oxidant generation)
Elemental carbon
Organic carbon (total, nonvolatile, and semivolatile; functional groups and individual species)

A central difficulty in trying to sort out possible mechanisms by which $PM_{2.5}$ influences health is the complex nature of particles themselves. As shown in Table 2-1, ambient $PM_{2.5}$ has diverse physicochemical properties ranging from the physical characteristics of PM to the chemical components in or on the surfaces of PM (U.S. EPA 2004). Chemical components of ambient $PM_{2.5}$ that might contribute to adverse health effects include acid particles, a variety of trace metals, reactive organic species, and biological constituents. Other characteristics, including particle size, shape, and surface area, might also be responsible for health effects. It is possible that different properties of $PM_{2.5}$ may stimulate different mechanistic pathways. Exposure to $PM_{2.5}$ may result in one or more pathways being activated, depending on the chemical and physical makeup of the particles. Although numerous hypotheses examine the relationship between certain physiochemical characteristics of ambient $PM_{2.5}$ and biological responses, a complete understanding of the pathways by which very small concentrations of inhaled ambient $PM_{2.5}$ can produce pathophysiological changes requires further research (Lippmann and Ito 2000; Schlesinger 2000; Schlesinger et al. 2006; Utell et al. 2002).

Toxicology and clinical studies have begun to provide insight into the possible mechanisms by which $PM_{2.5}$ exposure may lead to adverse cardiopulmonary health effects, as summarized in Figure 2-1 (Pope and Dockery 2006). While the evidence is still fragmentary, taken as a whole it starts to sketch a framework of biological plausibility which, in turn, could help support causality of epidemiology associations. Basic models of $PM_{2.5}$-related health impacts begin with deposition of $PM_{2.5}$ in the lung airways and the deep lung gas-exchange zone, or alveoli. Through complex and interdependent pathways, $PM_{2.5}$ interactions with respiratory epithelium likely mediate respiratory, systemic, and cardiovascular effects. Such effects may be especially deleterious to individuals compromised by serious heart or lung disease states such as ischemic heart disease, cardiac arrhythmias, and chronic obstructive pulmonary disease (COPD). Localized effects include inflammation, recruitment of phagocytic cell types, and cellular remodeling, resulting in fibrotic changes. The presence of inflammation may induce systemic effects, including increased blood viscosity and coagulability, and increased risk for myocardial infarction among persons with heart disease. Constituents of $PM_{2.5}$, such as ultrafine particles (<0.1 μm, UFPs), can be detected in the systemic circulation or associated with circulating cell types soon after inhalation. $PM_{2.5}$ or its reaction products may also stimulate airway sensory nerves, leading to changes in lung function and in autonomic tone, thereby influencing cardiac function (Donaldson et al. 2001; U.S. EPA 2004; Frampton 2001; Lippmann et al. 2003).

Understanding the health significance of carbonaceous $PM_{2.5}$ is a sensible research objective because of key physical, chemical, and spatiotemporal attributes of the particles. These attributes, summarized in Table 2-2, provide a framework for assessing health and exposure relevance Physical attributes of carbonaceous $PM_{2.5}$ include the size range, number, and surface area of the particles. In addition to encompassing a significant fraction of $PM_{2.5}$ mass, carbonaceous material comprises most UFP number concentration.

Figure 2-1. Potential Pathophysiological Mechanisms Linking PM Exposure to Cardiopulmonary Health Effects.



Some researchers hypothesize that UFPs are especially dangerous because of their physical structure. Ultrafine particles have a very large surface area that provides a base for adsorbed or condensed exogenous chemical materials. The extremely small size of UFPs enables the transport of these materials into the gas-exchange alveolar portion of the lung, bypassing primary defense mechanisms in the upper conductive airways. Chemical attributes of the carbonaceous fraction of $PM_{2.5}$ include particle-bound organic gases and compounds that vary by source and transformation processes. Pro-oxidative organic hydrocarbons represent a major constituent of $PM_{2.5}$. Polycyclic aromatic hydrocarbons (PAHs), for example, adsorb onto PM and may play a toxicological role in generating reactive oxygen species (ROS), oxidative stress, and inflammation once inhaled. In this way, organic particles can adversely affect airway epithelial cells and macrophages—the primary defense of the deep lung—and damage cellular proteins, lipids, membranes, and DNA (Li et al. 2003; Nel 2005; Sioutas et al. 2005).

Table 2-2. Key Attributes of Carbonaceous $PM_{2.5}$ Relating to Health and Exposure Assessment.

Attribute	Health and exposure relevance
Physical	Sizeable fraction of UFPs and $PM_{2.5}$
	Large surface area provides suitable base for potentially reactive material (particle core or adsorbed chemical materials)
	Small size conducive to pulmonary deposition into deep lung and translocation into systemic circulation and organs
Chemical	Comprised of heterogeneous organic constituents, some of which are known toxicants, e.g., nitrated PAHs
Spatiotemporal	Dominant fraction of combustion-generated particles that are prevalent in populated areas, e.g., diesel exhaust, biomass smoke
	Potential for high exposure in proximity to local sources; e.g., mobile sources
	Diurnal and seasonal variability of combustion-related emissions, e.g., higher risk in morning rush hour, winter heating season
	Terrain and meteorological parameters, e.g., low-lying areas prone to inversions.

In many populated areas combustion-related emissions sources dominate the carbonaceous fraction of ambient $PM_{2.5}$, with possible exposure and health consequences. These combustion particles are largely comprised of adsorbed organic constituents. Toxicological findings connect combustion particles with a variety of responses in the airways of laboratory animals and humans, including inflammation, cellular injury, and increased permeability. Epidemiological studies investigating surrogates of motor vehicle exhaust, proximity to traffic sources, and intracity gradients indirectly implicate elemental and organic carbon, as well as other mobile source emissions, in adverse health outcomes (Gauderman et al. 2005; Hoek et al. 2002a; Ito et al. 2004; Kinney et al. 2000; Schlesinger et al. 2006). The location of populations and activity patterns they exhibit influence personal exposure to these emissions. For example, residential and commuting populations can be exposed to high levels of freshly generated particles from nearby traffic sources. Meteorological patterns and geography can also affect emissions loading, such as diurnal variations in atmospheric stability or low-lying areas prone to inversions (Fruin et al. 2004; Jerrett and Finkelstein 2005; Larson et al. 1996; Norris et al. 2000; Zhu et al. 2004).

TOXICOLOGY FINDINGS

This section investigates recent findings of toxicity mechanisms related to carbonaceous $PM_{2.5}$. The diverse properties of these particles present a challenge to toxicological research because of the many physical and chemical parameters of interest. This constrains the ability to present these findings coherently with respect to carbonaceous $PM_{2.5}$ attributes. For example, comparability is impeded across the collective body of toxicological research involving carbonaceous $PM_{2.5}$ because many studies use diverse endpoints, different exposure concentrations, different methods to collect and generate PM, and different biological models and exposure systems (Schlesinger 2006). To facilitate this review, analysis is therefore structured by particle characteristics, specifically carbonaceous components and size, as a means to assess generally what researchers now understand about the carbon fraction. This framework necessarily does not group results with respect to study design and toxicity endpoints.

It is difficult to identify definitively which material, group of materials, or other particle characteristics are the toxic agent for health outcomes related to $PM_{2.5}$. It is likely that a combination of different variables is responsible. Constituents investigated include organic compounds known to adsorb onto PM, laboratory-generated particles stripped of chemicals, chemical mixtures extracted from PM, and real-world PM. Studies have also attempted to determine how different particle physical characteristics such as size and surface area affect toxicity. Carbonaceous particles exist in varying sizes that serve as cores for adsorbed and condensed chemical species. Toxicological evidence indicates that carbonaceous particle chemistry and size are to some degree important factors that lend plausibility to epidemiological findings of health effect associations. Such evidence is comprised of a variety of pulmonary and cardiac measures observed in humans and animals exposed to carbonaceous particles. Most of these studies have investigated short-term biological responses. Research is needed to investigate subchronic and chronic effects. These findings—summarized in Table 2-3 and discussed in detail in the following text—help to support current hypotheses explaining how $PM_{2.5}$ exposure could lead to adverse health outcomes such as morbidity and premature mortality in humans.

Table 2-3. Synopsis of Recent Toxicological Findings Relating to Carbonaceous $PM_{2.5}$.

Particle characteristic	Animal or human biological indicators	Short-term response	Research uncertainty
Carbonaceous component			
CAPs: EC/BC, OC constituents; DEP mixture with high carbon content: extracts of organic material, e.g., PAHs	Pulmonary vasculature; heart rate parameters; alveolar macrophage and neutrophils	Oxidative stress; inflammatory response in pulmonary areas and circulation; induced ROS in macrophages and bronchial epithelial cells	Core "inert" material or specific chemical causing effect; possible combination of transition metals and other PM components; isolated or synergistic effects among components
Size			
Particle core: UFPs and fine EC/BC	Epithelial cell lines	Macrophage effects; C reactive protein expression; proinflammatory effects; cardiovascular HR and HRV effects	Teasing apart particle size properties; EC UFPs may have intrinsic redox properties; particle size and chemistry unclear
Organic constituents: coarse, fine, UFPs; EC/BC, OC; CAPs; urban background, traffic, industrial; DEP	Macrophage-epithelial cell lines	Oxidative stress; cytokine production; ROS	

BC = black carbon, EC = elemental carbon, OC = organic carbon, CAPs = concentrated ambient particles, DEP = diesel exhaust particulate, PAHs = polycyclic aromatic hydrocarbons; ROS = reactive oxygen species, HR = heart rate, HRV = heart rate variability

Carbonaceous Components

Some studies have investigated the toxicity of elemental and organic constituents of carbonaceous particulates directly. Characterizations of potential health impacts associated with speciated elemental or black carbon (EC or BC) and organic carbon (OC) fractions of ambient $PM_{2.5}$ suggest possible mechanisms relating both to the particle core and to adsorbed chemicals. More definitive research is needed, however, to elucidate the role of specific chemicals as well as core materials in varying biological outcomes relating to respiratory and cardiac stress. Other studies have investigated surrogate groups of carbonaceous mixtures such as diesel exhaust particulates (DEPs) and PAHs. Findings suggest that these organic components of $PM_{2.5}$ are likely to be toxicologically active.

Particle Constituents. Investigations of specific carbonaceous constituents often rely on the use of concentrated ambient particles (CAPs) to simulate ambient exposures. Ambient PM concentrators allow toxicological studies to be conducted at higher-than-ambient concentrations. This approach enables researchers to characterize subtle mechanisms by which $PM_{2.5}$ exerts biological effects, allowing the testing of hypotheses as to how $PM_{2.5}$ induces the adverse health outcomes demonstrated by epidemiological studies. For example, Batalha et al. (2002) found that short-term concentrated urban $PM_{2.5}$ exposures induced vasoconstriction of small pulmonary arteries in normal rats and rats with chronic bronchitis. These changes suggest that pulmonary vasculature might be an important target for ambient air particle toxicity. The balance of vasoconstrictive and vasodilative mediators is an important regulator of pulmonary vascular tone and blood flow. Vasoconstrictive effects have been hypothesized to relate to plausible pathogenic pathways to adverse cardiac outcomes. The median concentration to which rats were exposed was about 183 $\mu g/m^3$ over three days. Biological effects were correlated with specific $PM_{2.5}$ components, including silicon, sulfates, Pb, and EC and OC concentrations. The authors, however, could not identify whether the observed changes were due to isolated or synergistic effects among the $PM_{2.5}$ components.

A clinical study of human adult exposures to urban air conducted by Urch et al. (2004) found possible adverse vascular associations with organic carbonaceous constituents. The investigators examined the relationship between total and constituent $PM_{2.5}$ mass concentrations and the acute vascular response in young healthy adults at rest in a crossover design. Each recipient received a two-hour exposure to filtered air and CAPs plus ozone on separate occasions. The CAPs plus ozone consisted of air sampled from a busy downtown Toronto street, concentrated to about 10 times the ambient $PM_{2.5}$ levels during 2000-2001. About 40% of the ambient composition was organic matter and 2% EC. Linear regression analyses were performed on estimated $PM_{2.5}$ constituent levels, demonstrating an exposure-response association between elevated levels of OC and EC and a decrease in brachial arterial diameter immediately after exposure. These responses were seen both for total carbon and for separate OC and EC fractions. Total $PM_{2.5}$ mass concentration was not significantly correlated with the immediate post-exposure difference. The authors concluded that one potential mechanism by which CAP inhalation may lead to arterial constriction is through particle-induced oxidative stress in the lungs or circulatory system. Simultaneous ozone exposure is a factor complicating the authors' conclusion, as well as an association approaching significance for certain trace elements (cadmium, potassium).

More recently, two studies found biological effects when animals were exposed to BC and OC. Tankersley et al. (2004) measured heart-rate regulation in healthy old mice and terminal old mice exposed to fine BC (3 hr/dy for three consecutive days). The average concentration was $160 \pm 22\ \mu g/m^3$. The BC exposure used in the study, however, may not have reflected the carbon composition in ambient $PM_{2.5}$. Study findings suggest that BC exposure acutely alters heart rate and heart rate variability parameters. The study demonstrated that the characteristics of acute PM-induced changes in heart rate and heart rate variability were dependent on the degree of physiological aging. The study supports the likelihood that both sympathetic and parasympathetic tone are altered during $PM_{2.5}$ exposure in healthy senescent mice. During acute $PM_{2.5}$ exposure, heart rate regulation was more varied and less predictable in terminally senescent mice. These results are consistent with epidemiological $PM_{2.5}$ studies in humans and laboratory-based exposures in animal models showing that changes in heart rate variability forecast susceptibility to cardiac morbidity and mortality.

A second study with rats, by Kodavanti et al. (2005), found associations for pulmonary effects with CAP constituents, suggesting that biological outcomes were more closely related to chemical composition than to total mass. The study demonstrated a pattern of rat-strain-specific effects, including changes in alveolar macrophages and neutrophils, after seven acute 2-day exposure periods (4 hrs/dy), but not for 1-day (4 hour) periods. These effects could play a role in inducing adverse pulmonary responses to $PM_{2.5}$ exposure, such as by triggering inflammatory reactions. Levels of Zn, Cu, and Al were enriched several-fold and OC was increased to a small degree per given mass of CAPs during some of the exposure periods relative to other exposure periods. Sulfates, OC, and EC accounted for >50% of CAP mass, and OC concentrations were 10-20 times higher than EC. The authors hypothesized that CAP effects may be revealed only when a specific CAP composition is formed, such as one with enriched metal and organic constituents.

Although these and other studies have focused on the carbonaceous fraction of $PM_{2.5}$, they do not exclude the possible contribution of transition metals and other components in the biological effects of $PM_{2.5}$ (Nel et al. 2001). Silbajoris et al. (2000), for example, exposed rat trachea to residual oil fly ash (ROFA), a particulate pollution rich in transition metals, including V and Ni, produced during fuel oil combustion. At 4 hours of exposure, mild and variable inflammation in the lungs occurred; by 24 hours a pronounced inflammatory response was present. Using CAPs from Boston's atmosphere, Saldiva et al. (2002) found that short-term CAP exposure induced pulmonary inflammation in rats, with stronger associations in chronic bronchitic animals. Significant overall associations were found for the following three items: CAPs as a binary term, V, and Br. Pb and OC also exhibited a strong association. Maciejczyk and Chen (2005) investigated the effects of daily *in vitro* exposures to CAPs in mice, finding that cellular stress response was most highly correlated with Ni and V among individual components, and with a residual oil combustion source category.

Other studies, however, suggest that transition metals may not be responsible for the observed biological effects. Brown et al. (2000) investigated whether UFPs exerted their effects through transition metals or other soluble factors released from the surface of the particles. The authors tested the imflammogenicity of BC (320 nm average size) and BC UFPs (14 nm) using rat models. Treatment with BC UFPs led to significant *in vitro* effects, but particle leachates did not affect this outcome. Differences could be accounted for by increased surface area or particle number reflecting an intrinsic particle toxicity, including a possible role of protein-surface interaction. Donaldson et al. (2002) similarly found that transition metals did not play a role in the extra inflammagenicity of BC UFPs compared to BC. The authors hypothesized that large surface areas generated oxidative stress via mechanisms that did not depend on transitional metals. Schins et al. (2004) investigated the biological activity of coarse and fine PM collected from industrial and rural environments in Germany. Coarse-sized PM from the rural environment induced the most potent inflammatory reaction in rat lung following intratracheal instillation The researchers were unable to elicit significant inflammation using the collected fine PM, despite its having a higher metal content than the coarse PM.

Surrogate Mixtures. In addition to studies focusing on individual $PM_{2.5}$ constituents, toxicological research on mixtures of organic material with high carbon content, such as diesel particulate, has found that certain chemical groups can induce a range of biological responses. Ma and Ma's review (2002) of the cellular actions of the organic and particulate components of DEP in the development of lung diseases found that while the components exhibit different biological actions, both induce cellular oxidative stress. These effects can inhibit cell-mediated immunity toward infectious agents, exacerbate respiratory allergies, cause DNA damage, and under long-term exposure can induce the development of lung cancers.

Several studies have suggested that DEP organic extracts may induce formation of reactive oxygen species (ROS) in macrophages and bronchial epithelial cells, two key cell types targeted by PM in the lung. Redox agents activate the pathways for cytokines and chemokines involved in inflammation. These processes promote the progression of atherosclerosis and precipitate a range of acute cardiovascular responses from increased blood pressure to myocardial infarction (Nel et al. 1998, 2001). Hiura et al. (1999, 2000), for example, found that DEP organic extracts induced apoptosis in macrophage cell lines as well as pulmonary alveolar macrophages. The extracts also induced production of ROS. In contrast, the carbonaceous core that remained after extraction of the PM did not exert cytotoxic effects. The authors suggested that this toxic cellular response may explain the adverse effects of DEP in the respiratory tract. In another study, Li et al. (2002a) showed that epithelial cells exhibit a hierarchical oxidative stress response that differs from that of macrophages by more rapid transition from cytoprotective to cytotoxic responses. The authors used organic DEP extracts, including fractions enriched for PAH and quinones. McDonald et al. (2004) evaluated relationships between chemical composition of vehicle exhaust and toxicity of inhaled emissions using principal component analysis and partial least-squares regression techniques. Chemical constituents covarying most strongly with toxicity included nitrated PAHs, OC, and organic compounds present in lubricating oil. A recent review by Risom et al. (2005) concluded that DEP causes oxidative DNA damage.

Biomass combustion $PM_{2.5}$ is another emissions source rich in carbonaceous material. For example, McDonald et al. (2006) recently characterized hardwood smoke atmospheres, finding that the PM phase was >90% OC, with small amounts of ions, elements, and BC. The review by Zelikoff et al. (2002) of the toxicological effects associated with exposure to wood smoke found that laboratory animals experienced increased microvascular permeability and produced pulmonary edema. Wood smoke exposure caused necrotizing tracheobronchial epithelial cell injury; it possibly increased lung cancer incidence and compromised pulmonary immune mechanisms. One group of investigators coexposed mice to BC and acrolein, concluding that adverse pulmonary biologic effects were due to carbon particles acting as vehicles to carry acrolein into the deep lung (Jakab and Hemenway 1993). Across the studies, it is believed that respirable pollutants in wood smoke alter pulmonary immune defense mechanisms in the lung, leading to pathogenesis and compromised host resistance that in turn may lead to increased respiratory infections. More studies are needed, however, to determine the effects of long-term exposure and the particular constituents of wood smoke that may be responsible for observed toxicities (Zelikoff et al. 2002).

Toxicological studies have also compared the degree to which active organic and inorganic components of carbonaceous mixtures can produce biologically active particles. For example, Bonvallot et al. (2001) compared the cellular effects induced by different DEP components, using human bronchial epithelial cell lines. The investigators compared native DEP, organic extracts of DEP, and the carbonaceous core (washed DEP, or black carbon particles), finding that DEP-induced inflammatory response mainly involves organic compounds such as PAHs. The carbonaceous core also exhibited a slight effect. Yanagisawa et al. (2003), however, demonstrated that the toxicity of DEP can be due to the particles themselves as well as to the extractable components. The authors found that organic chemicals in DEP and the residual carbonaceous core or nuclei (washed DEP) were responsible for the aggravation of lipopolysaccharide (LPS) mediated lung injury in mice. LPS is a proinflammatory agent component associated with aggravation of lung diseases. Washed DEP combined with LPS synergistically exacerbated pulmonary edema and induced alveolar hemorrhage, whereas organic chemical DEP combined with LPS did not. This additional finding suggests that the residual carbonaceous nuclei of DEP rather than the extracted organic chemicals predominantly contribute to the aggravation of LPS-related lung injury. This may be mediated through the expression of proinflammatory cytokines, chemokines, and Toll-like receptors.

Particle Size

A number of studies have attempted to understand the role of physical size in determining toxicity of carbonaceous PM. UFPs, for example, have been the subject of several investigations because their surface area and number concentrations per unit mass are greater than other particle size modes. These characteristics allow for greater adsorption of toxic materials and increased deposition of materials to the deep lung. Research results on particle size modes have been mixed, providing no definitive answers as to which particle size properties drive effects and to what extent chemical composition is responsible. Complicating factors include the difficulty of separating the effects of particle size modes from those of chemical composition, as well as understanding the role of particle dosimetry.

Core Constituents. Some studies of particle size have focused primarily on synthetic carbon particles. These facilitate an understanding of whether specific particle sizes can drive health effects irrespective of chemical characteristics. For example, Beck-Speier et al. (2001) evaluated the responses of epithelial cells to synthetic agglomerates of UFPs and $PM_{2.5}$. Study findings suggested that surface area rather than the mass concentration of the particles determined the biological response of alveolar macrophages to the particles. Ramage et al. (2004) studied fine and ultrafine BC PM, showing increasing expression of C-reactive protein after exposing human lung epithelial cells to inflammatory stimuli for 18 hours. Kim et al. (2005) found that synthetic EC UFPs appeared to be a potent inducer of proinflammatory responses in normal human bronchial epithelial cells. Harder et al. (2005) examined systemic cardiovascular effects of exposure to synthetic carbon particle for 24 hours in young, healthy rats, following the exposure protocol of Karg et al. (1998). The study demonstrated significant effects on indices of cardiovascular risk, including increased heart rate and decreased overall heart rate variability. Low-grade pulmonary inflammation was also found, but not evidence of increased blood coagulability. Elder et al. (2004) exposed old rats to laboratory-generated carbonaceous UFPs for six hours at concentrations mimicking high episodic events. The study found systemic responses for extrapulmonary effects but no suggestion of pulmonary inflammatory response. Gilmour et al. (2004) exposed rats to fine and UFP black carbon for seven hours. Ultrafine BC caused an increase in total leukocytes found in bronchoalveolar lavage and increases in blood leukocytes, whereas fine BC did not.

In contrast to these findings, a clinical study conducted by Pietropaoli et al. (2004) demonstrated only minimal changes in pulmonary function in healthy and asthmatic human subjects after EC UFP inhalation. There were no differences in any measurements in normal or asthmatic subjects when exposed to UFPs at concentrations of 10 or 25 $\mu g/m^3$. Higher concentration exposures of 50 $\mu g/m^3$ caused a reduction in maximal mid-expiratory flow rate, as well as carbon monoxide diffusing capacity, suggesting mild small-airways dysfunction together with impaired alveolar gas exchange in normal subjects. No consistent differences in symptoms, induced sputum, or exhaled nitric oxide parameters were found.

Although many of the above studies found associations using carbonaceous particles characterized by minimal extraneous content (e.g., organic species, metals, oxides, or sulfates), such findings must be interpreted with caution. EC UFPs, by virtue of their large surface area, may produce more reactive oxygen species in comparison with larger particles on a mass basis (Pan et al. 2004; Venkatachari et al. 2005; Yanagisawa et al. 2003). For example, Pan et al. (2004) explored the inherent chemistry of DEP, finding that DEP contains a stable and prevalent paramagnetic species, not unlike those found by other researchers in cigarette tar and extracts and airborne particles. This finding supports the theory that DEP is a reactive entity that can catalyze the reduction of oxygen by a variety of reducing agents, including but not limited to biologically relevant agents. The redox properties of DEP appear to be intrinsic, as the majority of the catalytic properties remained even after multiple extractions with methylene chloride or aqueous acid to remove chemical components from the particle itself. The authors suggest DEP toxicity is the result of the reactivity of numerous chemical species participating in various reactions, ultimately leading to the disruption of cellular function. Thus, focusing solely on particle extracts as the toxic components may be insufficient in describing the toxicity associated with DEP exposure. This makes difficult any definitive conclusions about the biological effects that physical size and surface area may exhibit in inert particles.

Organic Constituents. Researchers have also attempted to elucidate the role of particle size with respect to particle chemistry, but with indeterminate findings. For example, Diociaiuti et al. (2001) compared *in vitro* toxicity of coarse and fine PM collected in an urban area of Rome in relation to chemical composition. Although the hemolytic potential in mouse macrophage cell lines was greater for $PM_{2.5}$ than coarse PM in equal mass concentration, when data were expressed as surface area per volume unit of suspension, no significant differences were found. The authors suggested that oxidative stress induced by PM on cell membranes could be due mainly to the interaction between the particle surface and the cell membranes. This could indicate that differences depended on the different chemical nature of the PM. The main chemical difference between the two fractions resided in a greater abundance of carbon-rich PM with sulfur traces in the fine fraction.

Huang et al. (2003) exposed human bronchial epithelial cells to 100 µg/mL CAP extracts in Taiwan for eight hours, finding that both size and composition of ambient PM were important factors in inducing cytokine production and lipid peroxidation. The PM samples were representative of background, urban, traffic, and industrial air pollution patterns. PM-induced cytokine production by epithelial cells was size dependent, being greatest for submicrometer PM. Unexpectedly, the study did not find associations for $PM_{1.0\text{-}2.5}$ in epithelial cells, although this size fraction did induce tumor necrosis production by macrophages. The study found positive correlations between lipid peroxidation induction and the EC and OC content of the samples. This suggests that some organic components might have biological activity, in particular with oxidative properties.

In the Los Angeles basin, Li et al. (2003) analyzed the chemical composition of ambient concentrated coarse, fine, and ultrafine PM in parallel with assays for generation of reactive oxygen species. The UFPs contained significantly higher OC and EC than $PM_{2.5}$ or coarse PM and were more potent in inducing oxidative stress. Ultrafines also had higher rates of subcellular penetration and mitochondrial damage. Although the authors suggested a role of organic agents in generating redox activity, whether the observed effects were due to particle size alone or to chemistry was unclear. Cho et al. (2005) studied the redox activity of coarse, fine, and ultrafine PM fractions collected in different sites in the Los Angeles basin. Activity was highest in the ultrafine fraction; comparison of the redox activity with chemical composition showed a reasonable correlation with elemental carbon ($r^2 = 0.79$), organic carbon ($r^2 = 0.53$), and with benzo[*ghi*]perylene ($r^2 = 0.82$), consistent with species typically found in mobile emission sources.

Additional studies have attempted to determine whether different size fractions are responsible for observed effects of PM, using samples generated by vehicle traffic in urban areas. Li et al. (2002b) examined the role of EC- and OC-group chemistry and particle size by exposing animal cell lines to CAPs and DEP extracts. The authors demonstrated that ambient CAPs mimicked the effects of organic DEP extracts. This method can be advantageous compared to the traditional use of particle extraction from substrate, as employed by studies discussed above. The investigators found that biological effects were positively correlated to the higher OC and PAH content of the fine particles from air in southern California, as well as the rise in PAH content that occurs in coarse PM during winter months there. OC accounted for about 45% of the total fine PM mass, whereas EC accounted for about 6%. The total average PAH content of the fine fraction was approximately four times higher than in the coarse PM.

A recent study conducted in Rome by Pozzi et al. (2003) evaluated the role of adsorbed contaminants on the inflammatory responses induced by urban air PM collected near a heavy traffic area. By exposing mouse monocyte/macrophage cell lines, the researchers compared the effects of coarse and fine urban air fractions with BC of comparable size to $PM_{2.5}$. All three types of PM induced the release of inflammatory mediators, but BC was consistently less effective than both fractions of the urban PM. Also, the coarse PM produced higher levels of the mediators than $PM_{2.5}$. BC, however, was more effective than both urban fractions in inducing cytotoxicity as measured by lipopolysaccharide indicators. The authors suggested that the contaminants absorbed on the PM were responsible for the outcome more than the presence of the PM itself. Because 70% of the fine urban fraction was composed of carbon-rich PM, which is the main component of BC, the differences in inflammatory reaction between BC and the fine urban fraction were likely due to the contaminants in the coating of the urban fine fraction. Overall, the extent to which the observed effects and interactions are determined by particle size and composition was not determined.

Kendall et al. (2004) investigated the surface chemistry of $PM_{2.5}$ from roadside and background samples in New York City in order to quantify the adsorbed and desorbed species after treatment with human bronchoalveolar lavage fluid. X-ray photoelectron spectroscopy analysis showed that the surface of $PM_{2.5}$ at both sampling sites consisted of 79 to 87% carbon and 10 to 16% oxygen, with smaller contributions of N, S, Si, and P. A different spectrometry technique detected additional organic and inorganic species, including metals and hydrocarbons. Surface characteristics of the roadside and background sites were similar, except for higher nitrate concentrations at the roadside locations. The authors also compared these findings to a previous analysis of London $PM_{2.5}$ and found similar species and quantities. In addition, the New York study found that oxygen content of $PM_{2.5}$ surfaces was the most significant determinant of lung lining adsorption. These oxygen concentrations increased with “aged” $PM_{2.5}$.

Translocation. There is experimental evidence suggesting particle translocation can result in biological action. Researchers have investigated the ability of inhaled carbonaceous UFPs to translocate to blood and extrapulmonary organs in order to gain insight on potential mechanisms. A selective look at studies includes using radioactive labeling (technetium-99m) UFPs to assess to what extent and how the human-inhaled UFPs passed into systemic circulation (Nemmar et al. 2002). Radioactivity was detected in blood at one minute, reached a maximum between 10 and 20 minutes, and remained at this level up to 60 minutes. Oberdörster et al. (2002) found that EC UFPs accumulated to a large degree in the liver of rats by 24 hours after exposure, indicating efficient translocation into the blood. Following this work, Oberdörster et al. (2004) found significant and continuous increases of EC UFPs in the central nervous system of rats throughout a seven-day post-exposure period following a six-hour inhalation exposure. Using ultrafine titanium dioxide particles in rats, Geiser et al. (2005) found that nearly 25% of inhaled particles were located within and beyond the epithelial barrier, in all major lung tissue compartments and cells, suggesting particles have direct access to intracellular proteins, organelles, and DNA. Penn et al. (2005) showed that PAHs transferred from the particle surface to the cell membranes, crossed the membrane, and appeared to accumulate in lipid vesicles.

EPIDEMIOLOGY FINDINGS

This section investigates recent epidemiology findings related to carbonaceous $PM_{2.5}$. The analysis is structured to assess the limited body of studies that have directly investigated a variety of carbonaceous components, size, and mixtures. Because these particles have diverse physical and chemical properties, comparative analysis of study designs and endpoints is restricted. The group of epidemiological studies that has used source apportionment techniques is evaluated separately, followed by a brief treatment of relevant studies researching occupational exposures to carbonaceous materials.

Epidemiological studies have provided only a limited understanding of how specific $PM_{2.5}$ components and characteristics affect human health, primarily because of a paucity of speciated data. A few studies of ambient and personal exposures have included the direct analysis of the carbonaceous fraction of $PM_{2.5}$, focusing on the role of EC and OC components. Investigators have reported associations for a range of health outcomes, including subtle cardiac responses, respiratory morbidity, and premature mortality, but with varying relative risks and significance when compared to other particle components or mass measures. Relevant but more indirect studies relating to carbonaceous $PM_{2.5}$ have examined the role of ultrafine particulates and black smoke. This body of work has found correlations with increases in cardiac morbidity and mortality. Epidemiology researchers have also evaluated source-oriented data, showing suggestive relative health risks related to exposure to traffic, diesel exhaust, and other indicators of general mobile source emissions. Occupational exposure studies have reported significant associations for lung cancer and respiratory effect outcomes and PAH and DEP exposure. Table 2-4 presents an overview of these findings, which are discussed in the following text.

Personal, Indoor, and Ambient Air Exposure

A brief overview of current exposure assessment methods in epidemiological studies is necessary to understand the inherent limitations of observational studies that give rise to measurement error. Where individuals spend time (such as outdoors, indoors, and in vehicles) and what activities they perform (e.g., smoking or cooking) influence their exposure to $PM_{2.5}$. Investigators commonly rely upon mass measurements of outdoor atmosphere $PM_{2.5}$ concentrations at monitors centrally located with respect to the area being studied. These measurements are used as surrogates for exposure to ambient $PM_{2.5}$. Exposure to ambient $PM_{2.5}$, however, can occur both outdoors and indoors, as outdoor ambient $PM_{2.5}$ can penetrate indoor environments. Total personal exposure may differ from a central site monitor measurement because of spatial differences in ambient $PM_{2.5}$, the fraction of ambient $PM_{2.5}$ present in indoor environments, and the variety of indoor (nonambient) sources to which an individual is exposed.

Exposure measurement error resulting from these factors reduces the statistical power of health effects analyses, making it more difficult to detect associations between exposure and health effects. Nonetheless, central site measurements are a useful surrogate for ambient $PM_{2.5}$ exposure. Because ambient and nonambient components of personal exposure are independent, the health effects due to nonambient $PM_{2.5}$ generally will not confound the risk estimated for ambient exposures (Ebelt et al. 2005; U.S. EPA 2005). These findings do not apply to ultrafine and coarse particles, which can have different chemical, physical, and source properties that affect spatial distribution and indoor penetration. Nor do they address the poorly understood contribution of organic compounds to indoor $PM_{2.5}$ (Polidori et al. 2006).

Possible confounding relating to covarying pollutants can also complicate epidemiological study findings. Studies employing single-pollutant models, for example, cannot examine the possible influence of other pollutants. Where high correlations between or among pollutants exist, including components of pollutants, interpretations must use caution. Generally, studies employing co-pollutant models can more definitely investigate the role of specific pollutants in health response associations. EPA's recent review of single- and multi-city studies of PM concluded that, for the most part, the addition of gaseous copollutants had little influence on PM associations. A number of research groups have found the effects of various indicators of PM and gases to be independent of one another (EPA 2005). However, the possible influence of covarying particle characteristics should be acknowledged in emerging studies of PM constituents.

Table 2-4. Synopsis of Recent Epidemiological Findings Relating to Carbonaceous PM.

Particle characteristic	Human biological indicators	Short-term or long-term response	Research uncertainty
Carbonaceous component, size, mixtures			
PM_{10}, $PM_{2.5}$, UFPs; EC/BC, OC, TC; COH, black smoke; ambient urban, ambient mobile source, ambient wood burning	Pulmonary and cardiac parameters, cancer	Oxidative stress in blood, blood pressure, heart arrhythmias, ischemia, airway inflammation; cardiac hospital admissions; cardiovascular disease; cardiovascular mortality, all-cause mortality; lung cancer	Possible combination of gases and other PM components; isolated or synergistic effects among components; fixed site monitoring exposure or personal exposure; surrogate source category or direct association
Source apportionment			
Oil burning, mobile sources, vegetative burning, sulfate-associated mass	Pulmonary and cardiac parameters	Hospital cardiovascular disease admissions; cardiovascular mortality; all-cause mortality	Inability to differentiate effects of source proximity and toxicity of PM sources; unique constituents for identification of sources
Occupational studies			
DEP, DE, PAHs; combustion sources	Pulmonary and cardiac parameters; cancer	Respiratory symptoms and disease; variety of cancers including lung and respiratory	Exposure classification; healthy worker effect

EC = elemental carbon, BC = black carbon, OC = organic carbon, TC = total carbon, COH = coefficient of haze, DEP = diesel exhaust particulate, DE = diesel exhaust, PAHs = polycyclic aromatic hydrocarbons

EC and OC Components

A small number of epidemiological studies have directly investigated whether the carbonaceous fraction of PM induces short-term effects in humans. One group of studies used time-series mortality and morbidity outcomes based on ambient air pollution data collected at central monitoring sites. Emerging work has focused on additional markers of exposure including plasma protein concentration, arrhythmias, blood pressure, and airway inflammation. The studies have used either ambient central site or personal exposure monitoring data.

Time-series studies have investigated health effects from ambient carbonaceous $PM_{2.5}$ in Phoenix and Atlanta. Using three years of daily data (1995-1997), Mar et al. (2000, 2003) found that cardiovascular mortality for residents over 65 years of age in Phoenix was significantly associated with the EC and total carbon (TC) components of $PM_{2.5}$ (one-day lag). Weaker associations were evident with OC and TC at one- and three-days lag. The Aerosol Research and Inhalation Epidemiological Study (ARIES) included classes of OC in city-scale epidemiological studies for Atlanta (Van Loy et al. 2000). Three recent studies have reported findings using the ARIES database. Klemm et al. (2004) suggested that carbon (EC + 1.4 OC) was an important contributor for daily associations of all-cause mortality and cancer among the elderly from 1998 to 2000. Using data from the same time period, Metzger et al. (2004) found a positive association between emergency department visits for congestive heart failure and $PM_{2.5}$, EC, and OC. Peel et al. (2005) examined three-day moving averages for single-pollutant models for emergency department data for specific respiratory effect outcomes. The authors found associations for several specific pollutants; for example, pneumonia was correlated with $PM_{2.5}$ and OC exposure.

A recent New York City study suggested that the criteria pollutants $PM_{2.5}$, SO_2, O_3 and NO_2 had a statistically detectable impact on acute asthma emergency department visits in the Bronx, a community with a relatively high baseline rate of acute asthma exacerbations (NY DOH 2006). Using two-pollutant and three-pollutant regression models, the study found that O_3 and SO_2, and to a lesser extent maximum one-hour $PM_{2.5}$, were the most robust pollutants. Daily maximum three-hour exposures for EC were also significant, although OC was not. The study noted that the high correlations between pollutants (including measured components of $PM_{2.5}$ – EC, OC, sulfate, hydrogen ion and metals) made it difficult to identify critical compounds with confidence.

Epidemiology researchers in Europe, Boston, and Seattle have begun to apply additional exposure metrics to $PM_{2.5}$ constituent data, finding associations of sensitive cardiac and respiratory measurements with personal and ambient monitoring data. A study of 50 healthy young adults in Copenhagen (Sørensen et al. 2003) positively associated personal BC exposure and plasma protein concentration (a biomarker of oxidative stress in blood). Personal exposure was measured in two-day periods followed by collection of blood samples four times in one year (1999-2000) at relatively low exposures (mean $PM_{2.5}$ 16.1 $\mu g/m^3$). No associations were found for ambient $PM_{2.5}$ background concentrations. This could indicate a strong influence on the oxidation of plasma proteins from the ultrafine fraction of $PM_{2.5}$ that largely comprises BC. It could also imply that measurements of personal exposure include particle fractions that are not included in the background $PM_{2.5}$ concentration. Oxidative stress may be involved in atherosclerosis, suggesting a possible mechanistic relationship with cardiovascular disease. Henneberger et al. (2005) conducted a prospective panel study in Erfurt, Germany, in 56 males with ischemic heart disease. The study showed that increased levels of particulate air pollution were associated with significant changes in electrocardiogram (ECG) parameters. Most of the changes started within hours. The analysis showed a significant increase in QT duration (a parameter measured by an ECG) in response to exposure to OC; other effects such as a decrease in T-wave amplitude occurred with increased exposure to UFPs and $PM_{2.5}$.

In a Boston study of arrhythmias and air pollution, Peters et al. (2000) hypothesized that patients with implanted cardioverter defibrillators would experience potentially life-threatening arrhythmias after air pollution episodes. Investigators compared defibrillator discharge interventions among 100 patients with such devices, according to variations in concentrations of $PM_{2.5}$, BC, and gaseous air pollutants that were measured daily for the years 1995 through 1997 in one Boston monitoring site. A stronger association was found for nitrogen dioxide (NO_2) and BC than for $PM_{2.5}$. NO_2 and BC might have been markers for local traffic-related pollution, whereas $PM_{2.5}$ was influenced by local and regional transported particulate matter. Zanobetti et al. (2004) assessed the effects of $PM_{2.5}$ on blood pressure in a vulnerable population of 62 outpatients with cardiac disease evaluated repeatedly at rest and during exercise in a cardiac rehabilitation program. This Boston study concluded that both elevated blood pressure and ambient $PM_{2.5}$ exposure are associated with increased risk of cardiovascular morbidity and mortality. In single-pollutant models, increases in BC were associated with increases in diastolic blood pressure. The study used stationary measures of ambient air pollution, which could have led to a modest underestimation of pollution effects as compared to personal exposure measurements. Also in Boston, O’Neill et al. (2005) examined whether endothelium-dependent and -independent vascular reactivity was associated with 24-hour fine particle exposure in individuals with and without diabetes, a disease that may place individuals at increased cardiovascular risk. BC, sulfates, and fine particle increases were associated with vascular reactivity among those with diabetes. The authors concluded that diabetics were vulnerable to sulfates from long-range coal transport and black carbon from local traffic.

Jansen et al. (2005) conducted a panel study of 16 older subjects with chronic obstructive pulmonary disease and/or asthma in Seattle during the winter of 2002-2003. The study showed that an increase in indoor and outdoor BC, as well as outdoor PM_{10} and $PM_{2.5}$, was positively associated with increases in fractional exhaled nitric oxide (a noninvasive method of estimating airway inflammation among sensitive groups). Data implicated combustion-derived particle-associated BC, primarily from wood burning and, to a lesser extent, mobile sources, as being associated with airway inflammation in adult subjects with asthma.

Studies that have examined coefficient of haze (COH), which is correlated with EC, have found positive and significant associations with mortality, although often with less significance than other particle measures. In California’s Santa Clara County, Fairley (1999, 2003) found statistically significant associations for COH and other pollutant indicators during the 1980-1986 period, with a COH relative risk of 1.06. During the 1989-1996 period, however, the relative risk decreased to 1.03. The authors suggest one possible reason that the risk rate might have changed was that COH levels diminished from the early 1980s to the 1990s. Goldberg et al. analyzed congestive heart failure medical records and daily measurements of

COH in Montreal from 1984-1993 (Goldberg et al. 2000; Goldberg and Burnett 2003; Goldberg et al. 2003). The investigators found positive associations with COH and other particle measures. The mean percent increase in daily mortality for an increase in COH across the interquartile range was 4.32% (95% CI, 0.95-7.8).

Related Particle Measures

Epidemiological studies have begun to suggest a role for carbonaceous particles in driving the relationship between $PM_{2.5}$ and health effects through related particle measures such as UFPs and air pollutant components including black smoke. Some studies have used source identification and other inferential approaches relating to PM chemical speciation to try to discern the relative toxicity of $PM_{2.5}$ from various sources. This was done to enable researchers to identify $PM_{2.5}$ sources that cause adverse health effects. A few studies have relied upon recently collected speciated monitoring data, usually from fixed site monitors, whereas others use surrogate components that indirectly implicate source categories. Most studies, however, do not provide direct evidence of associations, in part because of available air pollution monitoring data on UFPs and related toxic constituents (Delfino et al. 2005).

Wichmann et al. (2000) were the first to investigate associations of mortality with detailed size categories of UFPs and $PM_{2.5}$. Particle number concentrations ranging in size from 10 μm to 2.5 μm were measured at a central monitoring site in Erfurt, Germany from 1995 to 1998. The study showed that UFPs (represented by particle number) and $PM_{2.5}$ (represented by particle mass) were associated with increased human mortality. In two-pollutant models, associations of UFPs and $PM_{2.5}$ seemed to be largely independent of each other, and the risk was enhanced if both were considered at the same time (Stözel et al. 2003; Wichmann et al. 2000).

A few studies have directly evaluated the effects on cardiovascular health by UFPs or particle number concentration. In Helsinki, Finland, Pekkanen et al. (2002) found significant associations between indicators of myocardial ischemia (ST-segment depression during an exercise test) in adults with stable coronary heart disease (CHD) and ambient $PM_{2.5}$ mass, number concentrations of UFPs 10-100 nm, and number concentrations of accumulation-mode particles 100-1,000 nm measured at a central regional site within the city. Odds ratios were around 3.0 for all particle metrics for an increase around their interquartile distribution. A panel study in three European cities explored the relationship between UFPs and blood pressure in adults with coronary heart disease, using data from fixed background monitors in urban sites (Ibald-Mulli et al. 2004). The study found that daily increases in UFPs were associated with small decreases in systolic blood pressure and diastolic blood pressure. The results did not support findings from previous studies that had shown an increase in blood pressure and heart rate in healthy individuals in association with increased exposure to $PM_{2.5}$. Effects might have differed in cardiac patients because of medication intake and disease status. In a large European panel study, Timonen et al. (in press) studied elderly subjects with coronary artery disease in Amsterdam, Erfurt, and Helsinki. UFPs, NO_2, and CO were associated with increased cardiac vagal control, while $PM_{2.5}$ had mixed associations. The authors concluded that the cardiovascular effects of ambient UFPs and $PM_{2.5}$ can differ from each other, while effects may be modified by the characteristics of the exposed subjects and sources.

Two recent studies have tried to correlate UFP exposure to heart rate variability. Chan et al. (2004) assessed the relationship between heart rate variability and continuous personal exposure to particle number concentrations of submicrometer particles (size range 0.02–1 μm) during daytime periods in Taipei. Exposure was associated with decreases in both time-domain and frequency-domain heart rate variability indices in young healthy adults and elderly patients with lung function impairments. The panel study by Chuang et al. in Taipai (2005) monitored changes in $PM_{2.5}$ mass concentrations and heart rate variability to determine whether PM and heart rate variabililty associations differed by particle size. Exposures to $PM_{2.5}$ with a size range of 0.3 to 1 μm at one- to four-hour moving averages were associated with decreases in heart rate variability indices in both cardiac and hypertensive patients. Associations were not shown between heart rate variability and either ($PM_{1.0\text{-}2.5}$) or coarse particles ($PM_{2.5\text{-}10}$).

Frampton et al. (2004) exposed healthy and mildly asthmatic men and women via a mouthpiece over two hours to laboratory-generated carbon UFPs (average diameter approximately 0.025 μm) and on a different occasion to filtered air. The laboratory-generated carbon UFPs did not contain toxicologically important components such as metals and organic compounds. The study observed that inhalation of carbon UFP at concentrations up to 25 $\mu g/m^3$ caused no symptoms, changes in lung function, or evidence for airway inflammation in healthy subjects. Blood leukocyte subsets and adhesion molecule expression did show subtle changes that suggest there may be effects on endothelial function. The study also found evidence for effects on heart rate variability and on cardiac repolarization in healthy subjects. The study noted that the findings may differ for children, the elderly, or people with more severe asthma or other diseases. In an accompanying piece to the study, a review committee concluded that in this limited set of healthy and mildly asthmatic participants, the effects of exposure to 10 or 25 $\mu g/m^3$ carbon UFPs for two hours were small and did not lend support to the hypothesis that UFPs are more toxic than other components of the broader particle mix (Frampton et al. 2004).

Evidence of causal relationships related to carbonaceous particles comes from epidemiological studies in Europe measuring black smoke, which is roughly representative of EC. In many European cities, diesel cars approach 50% of all cars and comprise a large source of urban PM, and the concentration of black smoke mostly arises from EC emissions from diesel engines. Katsouyanni et al. (2001) found that both PM_{10} and black smoke were predictors of daily deaths across Europe, with similar effect estimate sizes. The investigators used daily measurements for PM_{10} and/or black smoke from 29 cities, finding a 0.6% increase in daily numbesr of deaths for all ages (95% confidence interval (CI) of 0.5-0.7%) for a 10 $\mu g/m^3$ increase in one-day black smoke levels when analysis was restricted to days with black smoke concentrations below 150 µg/m^3 (black smoke levels represent concentrations of black particles with an aerodynamic diameter <4.5 µm). Effect estimates were slightly higher for the elderly.

A large cohort study in the Netherlands assessed the relation between long-term exposure to traffic-related air pollution and mortality from 1986 to 1994. Hoek (2003) and Hoek et al. (2002b) estimated black smoke and nitrogen dioxide levels at home addresses based on measured regional and urban background concentrations and an indicator variable for living near major roads. Contemporary road traffic comprises 65% of black smoke concentrations in Dutch urban environments. Cardiopulmonary mortality was associated with living near a major road (relative risk 1.95, 95% CI of 1.09-3.52).

Le Tertre et al. (2002) examined the association between airborne particles (PM_{10} and black smoke) and hospital admissions for cardiac causes in eight European cities. Their results suggest that hospitalization increases are likely to be mainly attributable to traffic diesel exhaust. Increases associated with a 10 µg/m^3 increase in PM_{10} and black smoke were respectively 0.5% (95% CI, 0.2-0.8) and 1.1% (95% CI, 0.4-1.8) for cardiac admissions of all ages. For cardiac admissions over 65 years, the increases were slightly higher 0.7% (95% CI, 0.4-1.0) and 1.3% (95% CI, 0.4-2.2). Similar rates of 0.8% (95% CI, 0.3-1.2) and 1.1% (95% CI, 0.7-1.5) were seen for ischemic heart disease over 65 years. Control for ozone or SO_2 caused little change on the effect of PM_{10}, while control for the traffic related pollutants CO substantially reduced the PM10 effect, and NO_2 virtually eliminated it. In contrast, controlling for CO resulted in practically no change in the effect of black smoke and controlling for NO_2 only somewhat reduced it. These results suggest that hospitalization increases are likely to be mainly attributable to traffic diesel exhaust..

Source Apportionment

The emerging use of source apportionment provides an analytical method of relating specific $PM_{2.5}$ constituents, such as those found in sources rich in carbonaceous PM, to health effects. For example, Tsai et al. (2000) analyzed daily mortality in relation to specific PM_{10} source types for three New Jersey cities from 1981-1983. The exploratory study found that certain PM_{10} sources were significant predictors for mortality, including oil burning, sulfate aerosol (used as a chemical tracer for an unspecified source type), industry, and motor vehicles. The authors noted numerous uncertainties in their findings, such as the inability to differentiate the effects of source proximity to residents and adverse health impacts from the PM_{10} sources. To some extent, these uncertainties apply to most source apportionment studies.

Researchers have begun to apply time-series health data to source apportionment analysis, inferring indirect connections between carbonaceous PM sources and health effects. Factor analysis techniques used in the Phoenix studies by Mar et al. (2000, 2003) discussed above revealed that motor vehicle and vegetative burning pollutants were significantly associated with daily measures of cardiovascular mortality. Laden et al. (2000) used the elemental composition of $PM_{2.5}$ collected for the Harvard Six Cities Study (Dockery et al. 1993) to identify distinct source-related fractions of $PM_{2.5}$. Integrated 24-hour samples were collected at least every other day from 1979 until the late 1980s, with daily sampling during health survey periods. Elemental composition was determined by X-ray fluorescence; a factor analysis method was used to identify single tracer elements for unique source categories. Combustion particles in $PM_{2.5}$ from mobile and coal combustion sources were associated with increased daily mortality. A 10 $\mu g/m^3$ increase in $PM_{2.5}$ from mobile sources accounted for a 3.4% increase in daily mortality (95% CI, 1.7-5.2%), whereas an equivalent increase from coal combustion sources accounted for a 1.1% increase (95% CI, 0.3-2.0%).

Grahame and Hidy (2004) evaluated the use of factor analysis by Laden et al. to attribute health impacts to $PM_{2.5}$ sources, finding that major local sources need to be taken into account to minimize misclassification. Source tracer elements overlooked the complexity of source contributions in Boston, one of the six cities studied. The presence of major local sources of residual oil combustion confounded the authors' original interpretation of these sources as transported coal combustion emissions from distant upwind sources.

In a 14-city U.S. study, Janssen et al. (2002) evaluated whether the contribution of different sources to total PM_{10} emissions could partly explain the observed variability in exposure-effect relationships. The authors applied meta-regression techniques to monitoring data from 1985-1994 and county data for source emissions category and vehicle miles traveled. PM_{10} regression coefficients for daily counts of hospital admissions for coronary vascular disease increased significantly with increasing percentage of PM_{10} emissions from highway vehicles, highway diesels, oil combustion, and metal processing. Multivariate analysis, however, found that only percentages of PM_{10} from highway vehicles/diesel and oil combustion were significantly related to coefficients for CVD. The investigators concluded that exposure-related differences could potentially explain a substantial fraction of the variation in city-specific coefficients for PM_{10}.

Epidemiology studies have also investigated wood smoke exposure of residents living in populated areas where wood burning is a contributing or dominant source of $PM_{2.5}$ air pollution. A recent review of the health effects of wood smoke by Naeher et al. (2005) concluded that the concentrations and durations of wood smoke exposure associated with residential wood burning is likely to cause a variety of adverse respiratory health effects, especially in children. These include increases in respiratory symptoms, decreases in lung function, visits to emergency departments, and hospitalization. For example, a large epidemiological study conducted in Seattle, WA where wood smoke contributed approximately 85% of the wintertime PM in residential areas during the one-year study period found that the daily counts of emergency room visits for persons under age 65 were significantly associated with PM_{10} exposure on the previous day (Schwartz et al. 1993). An earlier literature review of daily 24-hour exposure to PM in study areas where residential wood combustion was considered a major ambient PM source concluded that adverse health impacts from acute wood smoke exposure were not less than those found in areas dominated by other PM sources (Boman et al. 2003).

Thurston et al. (2005) recently evaluated the consistency of various source apportionment methods in assessing source contributions to daily $PM_{2.5}$ mass-mortality associations. Seven research institutions independently analyzed the same samples collected in Washington, D.C., and Phoenix using varying apportionment methods. The groups generally identified the same major source types, each with similar elemental makeups. Across the two cities, the most consistently associated $PM_{2.5}$ source category was sulfate-associated mass. The weakest cross-analysis correlations were usually found for the sources with greatest uncertainty in their composition. Sources lacking unique constituents for identification included traffic and wood burning in Washington, and wood burning and metals in Phoenix. Variations in choice of research group or source apportionment method had only a small effect on the variations in the relative risk estimates for identified sources, relative to the variations in risk elements for different source components. Source-specific estimates for motor vehicles approached significance and yielded values for relative risk of 0.9% per 10 $\mu g/m^3$ in Phoenix and 4.2% in Washington.

Ito et al. (2004) investigated additional issues relevant to the interpretation of source apportionment data. The researchers assessed the spatial variation of source-oriented evaluations of $PM_{2.5}$ health effects from New York City's fine particle chemical speciation sites. Whether monitors represent regional, sub-regional, or local air pollution exposures of city or metropolitan area populations can affect interpretation of the relative causal role of sources purported to have health effects. The authors found the strongest temporal correlations across three monitors for individual $PM_{2.5}$ components related to secondary aerosols. Four major source/pollution types were identified, including secondary aerosols, soil, traffic-related, and residual oil/burning/incineration particulates. While secondary aerosols (largely regional) displayed the highest monitor-to-monitor correlation (r^2 range = 0.72-0.93), the more localized traffic-related factor was more variable (r^2 range = 0.26-0.95).

Occupational Studies

Epidemiological studies of occupational exposures to comparatively high levels of carbonaceous PM are mostly limited to identifying cancer-causing compounds found in work environments. Studies of PAH and DEP inhalation exposure in workers indicate the cancer-causing potential of these carbon-rich mixtures. Studies have also found evidence of noncancerous effects, including respiratory outcomes.

Pitch, asphalt, coke, and tars are chemically related dense semisolid carbonaceous materials used widely in construction and in the production of metals such as steel and aluminum. These materials are composed of high molecular weight, aliphatic and aromatic hydrocarbons with trace amounts of sulfur or metals. Environmental exposures of concern are generally related to the content of PAHs and typically occur in association with domestic heating, vehicle exhaust, waste incinerators, certain industrial activities, and cigarette smoking. In workplace industrial environments, PAHs are formed during combustion of organic material and high-temperature processing of crude oil, coal, coke, or other industrial carbon compounds. Higher levels typically occur at gas works and coke, aluminum, iron, and steel plants, where occupational exposure is primarily by dermal absorption or inhalation. Many of the compounds classified as PAHs are potent carcinogens, although the carcinogenic potential of individual chemicals varies by many orders of magnitude. Studies of workers at coke ovens and gas generation facilities, roofers, and asphalt workers have implicated PAH mixtures in a variety of cancers, including those of the lung and respiratory system as well as bladder and skin cancers. Studies of exposure in workers at black carbon production facilities, however, have found statistically insignificant increases in lung cancer mortality (Schwarz-Miller et al. 1998; Taiwo and Mobo 2005).

Researchers have also investigated occupational exposure to diesel exhaust (DE) (which includes gaseous components as well as particulate). Evidence exists for a causal relationship between risk for lung cancer and occupational exposure to diesel exhaust. However, experts do not agree on whether the effects of smoking and other potential confounders —including historical diesel exhaust exposure data—have been adequately accounted for in key epidemiological studies. In over 35 diesel exhaust-related lung cancer studies conducted over the past 40 years, overall a persistent association with cancer risk has been observed in occupations where diesel exhaust exposure was likely, including in studies where it was possible to adjust for cigarette smoking. Risk of lung cancer was consistently elevated by 20-50%. Increased lung cancer risk, although not always statistically significant, has been observed in many of the cohort and case-control studies within several industries, including investigations of railroad workers, truck drivers, heavy-equipment operators, and farm tractor operators (Bhatia et al. 1998; Garshick et al. 2003; HEI 1999, 2002; Lipsett and Campleman 1999). Garshick et al. (2004) recently assessed lung cancer mortality in U.S. railroad workers, using over 38 years of concentration data. The study found that workers in jobs associated with operating diesel locomotives had a relative risk of lung cancer mortality of 1.40 (95% CI, 1.30-1.51). No relationship between years of exposure and lung cancer risk was observed, possibly because of the healthy worker survivor effect, historical changes in exposure, and the potential contribution of coal combustion products before the transition to diesel.

Some recent studies have investigated occupational exposures to carbonaceous materials and noncancerous health effects, providing additional insight into high-exposure settings. For example, Randem et al. (2003) observed a cohort of male Norwegian asphalt workers, finding somewhat elevated associations of nonmalignant respiratory disease with years since first employment in the asphalt industry. Mortality from respiratory diseases was highest among workers first employed in the 1960s, especially pavers and mastic asphalt workers, compared with those employed in later decades. A weak dose response was found with cumulative exposure to PAHs and asphalt fumes and mortality in analyses using a job-exposure matrix.

The combined human and animal evidence indicates that short-term exposure to diesel exhaust has been associated with irritation of the eyes, nose, and throat; inflammatory responses in the airways and lungs; respiratory symptoms (cough and phlegm); and neurophysiological symptoms such as headache, lightheadedness, nausea, vomiting, and numbness or tingling of the extremities. Such symptoms have been described mainly in reports of individuals exposed to diesel exhaust in the workplace, or in clinical studies in humans exposed to high short-term concentrations. Human occupational studies of DE provide some corroborative evidence of possible respiratory symptoms and possible lung function changes, but they are generally deficient in exposure information. A few human studies in various occupational settings suggest that diesel exhaust exposure may impair pulmonary function, as evidenced by increases in respiratory symptoms and some reductions in baseline pulmonary function consistent with restrictive airway disease. Other studies found no particular effects. Long-term and cross-sectional human studies have provided evidence for an association between respiratory symptoms (cough and phlegm) and diesel exhaust exposure (Garshick et al. 2003; Kinney 2003; Lloyd and Cackette 2001; Ris 2003).

HEALTH EFFECTS EVIDENCE AND LIMITATIONS IN UNDERSTANDING

Overall, as outlined in Figure 2-2, the combined findings of toxicological, clinical, and epidemiological studies provide evidence of the toxicity of carbonaceous $PM_{2.5}$, but cannot conclusively point to specific physical evidence or chemical indicators to the exclusion of others. Each discipline has to varying degrees investigated the potential role of carbon mixtures and surrogates, emissions sources, and particle size modes that are rich in carbonaceous $PM_{2.5}$, and individual carbon constituents. Viewed together, much of this work provides evidence of health effects in the form of biological responses measured in animal and clinical studies, as well as adverse health outcomes demonstrated among human populations. Research has not determined which specific physical characteristics and chemical components of carbonaceous $PM_{2.5}$ are responsible for adverse health effects. The complex task of separating particle size from other particle characteristics such as chemical composition, number concentration, and surface area has limited the interpretation of study results (U.S. EPA 2004; Schlesinger et al. 2006).

Toxicological research suggests that carbonaceous particle constituents such as organic and EC, DEP and PAH mixtures; ultrafine, fine, and occasionally coarse particle size modes; and other source indicators elicit biological response. Findings have demonstrated the involvement of carbon-related constituents in a variety of responses such as particle translocation, cytokine expression by alveolar macrophages and airway epithelial cells, pulmonary vasculature effects, and subtle cardiac effects such as heart rate variability. These results are consistent with current mechanistic hypotheses for PM injury relating to cardiac and respiratory health effects. Postulated mechanisms point to the inhalation of PM leading to the production in lung tissue of reactive oxygen species, which may then induce subsequent oxidative injury and inflammatory responses in the lungs and/or systemically. For example, *in vitro* and *in vivo* mechanistic studies have shown that PM derived from carbonaceous sources mediate their adverse biological effects by inducing oxidative stress, which may alter intracellular signal transduction pathways that regulate a number of biological responses such as cell growth, stress response, and inflammation. These outcomes could have adverse health consequences by exacerbating the health effects of a variety of preexisting diseases (Delfino et al. 2005; Dreher 2000).

Figure 2-2. Summary of Toxicology and Epidemiology Evidence on Carbonaceous PM.

Research discipline	Surrogate or mixtures	Sources	Size mode	Constituents	Biological response
Toxicology, clinical	CAPs, DEP, PAHs	Mobile, industrial, biomass	UFPs, PM2.5	Inert core, EC/BC, OC extract	Animal models and humans: oxidative, inflammatory, epithelial, macrophage, vasculature stress; C reactive protein expression; HR and HRV effects
Epidemiology	COH, black smoke, DEP, DE, PAHs	Mobile, oil, sulfate-related, biomass	UFPs, PM2.5	EC/BC, TC, OC extract	Measured respiratory and cardiac indicators of stress and symptoms; hospital admissions for disease; mortality

CAPs = concentrated ambient particles, DEP = diesel exhaust particulate, PAHs = polycyclic aromatic hydrocarbons, EC = elemental carbon, BC = black carbon, OC = organic carbon, HR = heart rate, HRV = heart rate variability, COH = coefficient of haze, DE = diesel exhaust, TC = total carbon

The utility of toxicology studies of carbonaceous particles is tempered by the discipline's reliance on model or surrogate exposures as distinct from real-world environmental exposures, use of small sample size populations, and extrapolation from animal or *in vitro* models to human ambient exposure. As noted by Schlesinger (2000), dosimetric relationships between epidemiological and toxicological findings are tenuous. Observational studies show concentration-dependent associations of increased adverse health effects even at low ambient concentrations, while experimental studies show associations of biological responses only at much higher levels, with the relationship between exposure concentration and response not always obvious.

The limited use of epidemiological tools in understanding the specific role of carbonaceous $PM_{2.5}$ is largely attributable to insufficient speciated data and traditional limitations in study methodologies. Current data resources such as the U.S. EPA $PM_{2.5}$ Chemical Speciation Trends Network (STN), for example, might have sufficient temporal resolution to support time-series epidemiological studies, but they provide insufficient spatial resolution to support cohort epidemiological studies (Pun et al. 2004; Sheppard 2004). Methodological issues that have long challenged epidemiological investigations of fine particle mass metrics also constrain efforts to assess the health significance of PM constituents. These include the choice of susceptible populations, adequate spatial and temporal resolution of personal exposure assessment, and covariance of other pollutants (Delfino et al. 2005).

Epidemiological studies attempting to analyze specific carbonaceous species are inconclusive, providing possible but not incontrovertible evidence of health effects associated with carbonaceous species. Definitive understanding is lacking as to which factors related to the carbonaceous species drive health effects and whether these exert more toxicity than other PM chemical components and physical characteristics (Schlesinger et al. 2006). Environmental and occupational epidemiology studies of source-oriented combustion products or carbonaceous mixtures—such as mobile source-related emissions, DEP, and PAHs—provide some evidence of the potential role of carbon-rich emissions in driving adverse health effects. A growing number of epidemiological studies have found associations between mobile source emissions and health outcomes, indirectly suggesting the potential role of carbonaceous $PM_{2.5}$ and UFPs (Sioutas et al. 2005). Many of these studies have used surrogate measures of fossil fuel combustion such as black smoke, proximity of homes to traffic, and source apportionment data. Mobile source emissions are rich in carbonaceous material and UFPs, but these source emissions also contain co-varying pollutant gases such as CO and semivolatile organics. Whether traffic-related health associations are caused by physical size alone, or by the combined effects of chemical and biological components of particles, specific organic compounds, or gases is unknown.

It is difficult to know if researchers will someday determine whether a single source profile or group of sources drives the majority of health effects. For example, the recent study by Lippmann et al.(2005a,b) of heart rate measures during subchronic exposures of mice to regional anthropogenic CAPs in Tuxedo, NY, found that different source factors showed some indication of strong association over varying temporal periods. The study is unique for investigating subchronic toxicological effects. Investigators conducted their analysis in a large state park remote from large local PM sources, using ambient particles largely formed by long-range transported secondary ambient aerosols. The authors suggest that most, if not all, PM categories could have some, if various, effects on cardiac physiology, with various temporal periods. Moreover, some components might mitigate the effects of others.

Using source apportionment techniques to investigate urban air particles, Mar et al. (in press) analyzed inter-method variations in associations between estimated source contributions of $PM_{2.5}$ and daily mortality in Phoenix. Of the apportioned anthropogenic $PM_{2.5}$ source categories, secondary sulfates, traffic, and copper smelter-derived particles were most consistently associated with cardiovascular mortality. The sources with the largest effect size were secondary sulfates and traffic. The time lag of the time-series mortality associations varied across the various $PM_{2.5}$ source components, possibly suggesting that different mechanisms of effect are at work for the various types of $PM_{2.5}$. The New York and Phoenix studies point to the need for further research to disentangle the complex relation between fine particle components and adverse health effects.

EXPOSURE ASSESSMENT

The preceding sections reviewed a large collection of current health effect studies that have investigated the toxicity of carbonaceous $PM_{2.5}$. Many of these studies implicate carbonaceous $PM_{2.5}$ in adverse health outcomes, and some suggest possible mechanisms for these effects. They do not, however, point to carbonaceous aerosols as the sole or even dominant driver of $PM_{2.5}$-related health effects. In addition, uncertainty prevails as to which physicochemical aspects of this complex fraction of highly correlated $PM_{2.5}$ constituents exert toxicity. Nonetheless, carbonaceous constituents or sources rich in carbonaceous aerosols appear to represent an important determinant of $PM_{2.5}$ health-related responses. Therefore, it is useful to assess population exposure to carbonaceous $PM_{2.5}$ emissions in New York because the implications are of potential public health significance.

This subsection reviews key factors relating to exposure. It discusses elevated carbonaceous $PM_{2.5}$ levels in densely populated urban areas, large mobile source contributions to carbonaceous $PM_{2.5}$ in urban and rural areas, and dominant residential wood combustion contributions to carbonaceous $PM_{2.5}$ in rural areas of New York State. These findings indicate that mobile source and residential wood burning emissions present potentially hazardous exposures for New York populations, based on a review of health studies of possible relevance to New York. The subsection also presents an expanded mobile source case study that includes a NESCAUM analysis of population residence and traffic volume data in New York. The analysis found that large numbers and high densities of persons in the state live within 150 and 300 meters of major roads in selected rural, suburban, and urban counties.

Carbonaceous $PM_{2.5}$ Measurements and Source Inventories

Exposure considerations involve the interaction of many variables, including pollutant concentration, source characteristics, population attributes, and atmospheric processes (Brauer et al. 2002). These factors can vary between neighborhood blocks or statewide regions and from brief diurnal events to changing seasonal and yearly averages. Pollutant concentration depends upon where $PM_{2.5}$ and its constituents are measured and how accurately such measurements characterize ambient, nonambient, and personal exposures. Relevant emissions source characteristics include where and when sources operate, the quantity of these source emissions, and the physicochemical character of the emissions. Population attributes include where people reside and work, what activity patterns individuals have, and whether persons are susceptible to air pollution. Influencing each of these factors, atmospheric processes include the extent to which terrain and meteorological processes influence the transport and dispersion of pollutants, and how atmospheric forces govern aerosol chemistry.

Carbonaceous $PM_{2.5}$ levels in New York State result from a variety of local and distant emissions sources. Anthropogenic primary $PM_{2.5}$ commonly consists of unburned carbon particles directly emitted from high-energy processes such as combustion, and particles emitted as combustion-related vapors that condense within seconds of being exhausted to ambient air. Carbonaceous $PM_{2.5}$ is also comprised of secondary particles formed from precursor gases reacting in the atmosphere or through the addition of PM and condensable vapors onto preexisting particles. Across the state, urban area annual averages and 24-hour averages typically exceed $PM_{2.5}$ concentrations at nearby rural sites. Based on the analysis of STN data in this report, the annual carbonaceous contribution to overall $PM_{2.5}$ ranges from 28 to 40% in rural and urban areas, respectively.

The highest levels of carbonaceous $PM_{2.5}$ are often found in major urban centers or areas where local sources strongly influence ambient air quality beyond background levels influenced by interurban and long-range transport. Different sources influence concentration levels at rural monitoring sites. These include local sources, locally transported pollutants from within the region, and regionally transported pollutants from distant upwind sources. Nearly all $PM_{2.5}$ measurements are collected at centrally located monitors intended to characterize pollutant levels across large areas. As stated previously, traditional monitoring networks may not adequately represent total population exposure, and may not therefore be a good predictor of population risk. This is in part because the networks can fail to account for the variability of $PM_{2.5}$ and carbonaceous $PM_{2.5}$ across dynamic spatial and temporal scales, as well as population characteristics.

$PM_{2.5}$ source emission inventories represent another method of assessing potential population exposure. Based on available speciated inventory data (U.S. EPA NEI 1999, 2002), which represents 70% of the overall primary $PM_{2.5}$ emissions, carbonaceous $PM_{2.5}$ emissions comprise 55% of total $PM_{2.5}$ emissions in the state. The actual overall percent contribution may be somewhat different because the database lacks speciation profiles for 30% of total primary $PM_{2.5}$ emissions from a multitude of small sources. Table 2-5 presents calculated inventory numbers by major categories for all primary carbonaceous $PM_{2.5}$ sources in New York State (omitting the potential carbonaceous $PM_{2.5}$ fraction from the 30% unspeciated $PM_{2.5}$). The table is organized by New York metropolitan tri-state $PM_{2.5}$ nonattainment area (NAA) and three rural counties. The ten nonattainment counties in the New York City area are primarily urban, but they also include heavily populated suburban counties such as Westchester, Nassau, and Suffolk. The tri-state nonattainment area also includes ten New Jersey counties and two Connecticut counties. The three rural New York State counties are Greene, Lewis, and Cattaraugus. It is important to recognize that these data are generated by analytical methods subject to limitations and uncertainty. Chapters 1 and 3 discuss in more detail how speciated and inventory data are generated, along with the strengths and weaknesses of this form of analysis. While percentage values should not be considered precise, across major source categories they do indicate general proportions with some degree of confidence.

Table 2-5. Major Carbonaceous $PM_{2.5}$ Combustion Source Categories in New York City Metro Tri-State Nonattainment Area and Three New York State Rural Counties.

Source category	NY metro tri-state NAA (%)	3 rural NYS counties (%)
Mobile		
nonroad diesel	24	3
onroad HD diesel	13	2
onroad LD gas	7	1
Airport	1	0
Marine	3	0
Other	10	1
Residential energy		
wood combustion (RWC)	18	92
Other	1	<1
Charbroil meat cooking	18	1
Other stationary	5	0
Total	100	100

Percent contribution to total based on inventory estimates. Total does not include the carbonaceous $PM_{2.5}$ portion of an unspeciated 30% contribution to the primary $PM_{2.5}$ inventory from many small sources.

The table shows distinct inventory differences between the primarily urban nonattainment counties in the tri-state New York City area and the three rural counties. In the three rural counties, residential wood combustion generates 92% of all carbonaceous $PM_{2.5}$ emissions. These emissions also comprise a large fraction of total $PM_{2.5}$ emissions. An inventory assessment for three rural counties in the state found that residential wood combustion accounted for an average of 68% of total $PM_{2.5}$ emissions across the counties; the range was 46-78%. Mobile sources are the second largest category in the rural counties; nonroad diesel and onroad heavy-duty diesel account for 5% of carbonaceous $PM_{2.5}$. In the tri-state New York City counties, mobile sources generate 58% of all carbonaceous $PM_{2.5}$ emissions. These sources account for 29% of total $PM_{2.5}$ emissions in the area. Commercial charbroil meat cooking and residential wood combustion each comprise 18% of carbonaceous $PM_{2.5}$ in the New York City tri-state area.

It is useful to consider inventory tons per year in addition to percent contribution of various carbonaceous $PM_{2.5}$ categories in order to understand how nonurban and urban areas contribute to total statewide emissions. Table 2-6 provides these data for areas organized by five New York City counties and all other New York State counties (except for the four counties home to Albany, Buffalo, Rochester, and Syracuse). The table shows that nonurban counties in New York State contribute a sizeable number of mobile source nonroad and onroad emissions compared to the New York City urban area. These nonurban areas also contribute predominately to statewide totals for residential energy emissions.

Table 2-6. Total Carbonaceous $PM_{2.5}$ Combustion for Selected Source Categories in Five-County New York City Urban Area and Rural "Upstate" New York Counties.

Source category	5 New York City counties (tons per year)	New York State counties* (tons per year)
Mobile		
nonroad diesel	2,069	1,937
onroad HD diesel	568	1,638
onroad LD gas	465	672
Airport	180	26
Marine	469	0
Residential energy		
wood combustion	149	37,132
Other	66	49
Charbroil meat cooking	1,623	849

* These include all New York State counties except five New York City counties and four other counties that comprise the cities of Albany, Buffalo, Rochester, and Syracuse.

Demographic Features

In addition to $PM_{2.5}$ measurements and source inventories, population attributes represent a third important factor relating to exposure. Demographic features such as population density can help to characterize potential exposure implications, especially in combination with known measurement and source inventory data. For example, New York City experiences the state's highest $PM_{2.5}$ levels, and a substantial fraction of the city's carbonaceous $PM_{2.5}$ is generated by mobile sources. The city comprises over 40% of the state's total population and is also the most densely populated major city in North America. New York City's five-county 2000 U.S. Census data shows about 26,000 persons per square mile. The average density in New York County is nearly 67,000 persons per square mile; Bronx County and Kings County have about 32,000 and 35,000 persons per square mile, respectively. No other major U.S. city has a population density greater than 16,000 persons per square mile, and only four other cities exceed 10,000 persons per square mile.

As shown in Table 2-7 New York City and, to a lesser extent, other major urban areas across New York are home to a large portion of the state's population even though they comprise a comparatively small land area. This pattern was found by Johnson and Graham (2005) for the northeastern U.S., where 72% of the population lived in 6% of the region's landmass, exhibiting the highest population density levels in the nation. The percentage of child, adult, and elderly age subgroups living in urban areas was nearly identical, ranging from 71 to 73% across groups. Yet carbonaceous $PM_{2.5}$ sources and concentrations can have potential exposure implications for sizeable populations even in small towns situated in predominately rural areas that comprise much of New York State's nonurban population. Table 2-7 also provides examples of towns located in nonurban areas but which possess urban-scale density of greater than 1,000 persons per square mile.

Table 2-7. Demographic Features of Selected New York State Population Centers, 2000 U.S. Census.

Location	Population	Land area (sq miles)	Persons/sq mile*
New York State	18,976,500	47,214	400
New York City	8,008,300	303	26,400
Buffalo	292,700	41	7,200
Rochester	219,800	36	6,100
Syracuse	147,300	25	5,900
Albany	95,700	21	4,500
Ithaca	29,300	5	5,400
Saranac Lake	5,000	3	1,800
Dolgeville	2,200	2	1,200

*Rounded values may not divide precisely.

Residential Wood Burning

Although studies indicate that wood smoke inhalation presents a health risk to exposed populations (as discussed in the toxicology and epidemiology findings, above), wood combustion exposure assessment and measurement studies within New York State and in the Northeast region are limited. Studies of other areas that have large wood combustion source contributions, however, provide evidence to suggest indirectly that residential wood smoke is of potential exposure concern in areas within New York State where residential wood burning may be relatively high.

Emissions inventories for three rural counties in New York State show that residential wood combustion in these counties is a predominant source of carbonaceous $PM_{2.5}$. Other work has found high residential wood burning activity in areas within New York and the Northeast (e.g., MARAMA 2004). Visalli (1984) reported a sudden increase in wood use for residential heating during the early 1980s, finding that about one million wood heating systems were in use in New York State, with over 80% of the stoves less than five years old. During the past two decades, woodstove and fireplace insert ownership in the U.S. has been static, numbering about 14-17 million appliances (Broderick and Houck 2005). New England has the highest per capita woodstove ownership in the U.S., and people in the Northeast Census Region (New England, New Jersey, New York, and Pennsylvania) consume over twice the number of total cords of wood in woodstoves per year than those in the Midwest, South, or West (Houck et al. 2001). Nearly all U.S. woodstoves in current use today were manufactured without consideration of PM emissions, even though U.S. EPA standards limiting PM emissions from manufactured woodstoves went into effect in 1990 (U.S. EPA 1988). Upwards of 80% of the current woodstove units do not meet U.S. EPA certification requirements (Broderick and Houck 2005).

Investigations in urban and rural areas have found that wintertime residential wood combustion emissions can contribute significantly to ambient $PM_{2.5}$ and VOC levels (e.g., Fine et al. 2001; Maykut et al. 2003; McDonald et al. 2000; Schauer and Cass 2000). In areas subject to air stagnation, such as valleys, inversion conditions can give rise to significant buildup of $PM_{2.5}$ (e.g., Luhar et al. 2006; McGowan et al. 2002; Noullett et al. 2006). In Waterbury, VT, Sexton et al. (1984) estimated that wood burning was the major source of PM. Peak nighttime values exceeded afternoon levels by five- to tenfold, which the authors attributed to meteorology and emissions patterns. The analysis of atmospheric aerosol chemical composition over Vermont by Polissar et al. (2001) found strong local source influence from residential wood combustion in northern New England and southwestern Quebec during wintertime. Recently, near-source field monitoring of an outdoor wood boiler found elevated levels of $PM_{2.5}$ relative to background in a rural New York State county, suggesting a public health concern for populations living near these devices (Johnson 2006). Increasing use of outdoor wood boilers has seen a commensurate rise in public complaints to regulatory agencies (NESCAUM 2006; Schreiber et al. 2005).

PM from residential wood combustion is largely comprised of submicron particles with average mass diameters generally between 0.1 and 0.6 μm (Hueglin et al. 1997; Kleeman et al. 1999; Purvis et al. 2000; Rau 1989). Individuals can receive substantial exposure to outdoor-generated wood smoke particles while they are indoors because of particle infiltration. Studies in wood burning areas have found that indoor $PM_{2.5}$ levels (even in homes without woodstoves) can comprise from 20% to 80% of outdoor $PM_{2.5}$ levels (Anuszewski et al. 1998; Abt et al. 2000; Allen et al. 2003; Dockery and Spengler 1981; Meng et al. 2005; Larson et al. 2004. Recent studies have suggested that the ambient-generated component of $PM_{2.5}$ exposure is associated with greater measures of adverse health effects than indoor-generated $PM_{2.5}$ components (Ebelt et al. 2005; Koenig et al. 2005).

Mobile Sources

As discussed earlier, mobile source emissions comprise the largest fraction of carbonaceous $PM_{2.5}$ in the tri-state New York City area (58%) and the second largest fraction in three New York State rural counties (6%). Current studies suggest that exposure to mobile source emissions presents a potential health risk to populations who live and work in a variety of environments. A growing number of exposure assessment and measurement studies in Europe, North America, and the New York City area indicate that urban populations can be exposed to high levels of mobile source emissions over varying temporal and geographic scales. Few comparable exposure assessments of rural populations have been undertaken.

Based on these findings, the following case study presents a screening-level analysis of residential population proximity to and traffic volume on major roads in selected New York State counties. The analysis demonstrates that a large number of New York populations in rural, suburban, and urban counties live within 150 and 300 m of major roads. An analysis of suburban and urban counties found that major roads have high vehicle miles traveled and average annual daily traffic counts. These findings suggest that New York State populations are at potential risk to exposure to mobile source emissions. Population residence near major roads, however, represents only one of several possible mobile source exposure pathways to populations. Other pathways, such as pedestrian, in-vehicle, and in-station exposure require further research.

Selected Health and Exposure Findings. Exposure assessment studies employing a variety of analytical approaches report large geographic variations in mobile source emissions, typically finding elevated concentrations closer to high densities of traffic. Brauer et al. (2003) used a measurement and modeling procedure to estimate the long-term average exposure to traffic-related pollution in rural and urban traffic, and in urban background locations in Europe. Regression models using traffic related variables (e.g., population density and traffic intensity) explained a substantial fraction of the variability in annual average concentrations for all locations for $PM_{2.5}$ and filter absorbance (a marker for DEP). In Los Angeles, Zhu et al. (2002) found that concentrations of UFPs, black carbon, and CO decreased exponentially with distance downwind of a major freeway, with the most rapid decline occurring within 150 m and declining to levels indistinguishable from upwind background concentrations at 300 m. Particle number concentration near the freeway site was about 25 times greater than that at background locations. Hitchins et al. (2000) found that for wind blowing directly from a motorway in Australia, the concentrations of $PM_{2.5}$ and UFPs decayed to around half of the maximum at a distance of about 100-150 m from the road. The effect of vertical dispersion in densely populated street canyons presents another exposure consideration. Chan and Kwok (2000), for example, found that $PM_{2.5}$ can penetrate to the tenth floor and above in an urban area of Hong Kong. The ability of ambient $PM_{2.5}$ to infiltrate indoor environments such as roadside residential dwellings was reported in an Amsterdam study comparing outdoor and indoor air in homes located between major and small side streets (Abt et al. 2000; Wallace 1996). Black smoke concentration ratios between these two areas were 1.84 for outdoor air and 1.83 for indoor air (Fischer et al. 2000).

These and other studies have provided epidemiologists with indirect measures of exposure using population distance from roadway, proximity to high traffic density, and pollutant measurements. The recent review (in press) by White et al. found that pollutant concentrations can be substantially elevated near highways and roadways with large traffic volumes. They concluded that epidemiological data suggest increased risks for exacerbation of asthma and other respiratory diseases, premature mortality, and certain cancers and birth outcomes from air pollution exposures in susceptible populations (e.g., persons with respiratory or cardiac disease, children) residing in relatively close proximity to roadways. Based on exposure proxies, adverse effects have been found in populations close to major roads (e.g., up to 200 to 300 m distant) or near roads with high traffic intensity (e.g., as low as 10,000 vehicles per day at 50 m).

While carbonaceous $PM_{2.5}$ constituents such as black carbon, for example, may serve as an appropriate indicator of fresh mobile source emissions, they may not be causing observed health effects. In light of indirect exposure evidence, possible cofactors requiring further research include noise, socioeconomic status and related social stressors, road dust, and emissions related to brakes and tire wear. Additionally, many studies have been conducted in European cities where street layouts differ and diesel engines predominate. More recent exposure assessment and epidemiological work, notably in southern California, lends support to European traffic-associated exposure and health findings (e.g., Gauderman et al. 2005; Zhu et al. 2002).

In Europe, Künzli et al. (2000) estimated the impact of outdoor and traffic-related air pollution on public health in three countries in order to estimate attributable cases of morbidity and mortality. The researchers concluded that air pollution caused 6% of total mortality (>40,000 cases per year), attributing about half of all mortality to motorized traffic. In southern Germany, Peters et al. (2004) observed an association between subject exposure to traffic and the onset of a myocardial infarction within one hour. The time the subjects spent in cars, on public transportation, or on motorcycles or bicycles was consistently linked with an increase in the risk of myocardial infarction.

Gauderman et al. (2005) examined associations between traffic-related pollution (NO_2) and childhood asthma in 10 southern California communities. The study found increased asthma and wheezing associated with closer residential distance to a freeway and with model-based estimates of outdoor pollution from a freeway, concluding that respiratory health in children was adversely affected by local exposures to outdoor NO_2 or other freeway-related pollutants. In Los Angeles, Jerrett et al. (2005) reported general agreement with earlier Dutch and Canadian studies suggesting that intra-urban exposure gradients may be associated with even larger health effects than reported in inter-urban studies. Such studies implicated traffic particulates as a potential source of pollution associated with larger observed effects. Intra-urban exposure gradients found by Jerrett et al. resulted in reported chronic health effects nearly three times larger than in studies contrasting between-community exposure contrasts; these health effects could be partly the result of higher proportions of traffic particulates.

Several studies have investigated “hot-spot” exposures for populations near or inside train stations, bus stations, and port or marina areas; inside subways, buses, cars, and ferries; or active near transportation corridors (e.g., bicyclists, pedestrians). Recent studies have found that in-vehicle journey-time can lead to significant exposures within aircraft, cars and buses, ferries, and subways (Adams et al. 2001; Chan et al. 1999, 2002a,b; Lindgren and Norback 2002; Marshall and Behrentz 2005). Such exposures, although brief, can contribute a significant fraction of a person’s daily exposure, even though time spent in such microenvironments was only a fraction of total time (Rea et al. 2001). Factors influencing exposure include mode of transport, fuel use and emissions characteristics of transport device, proximity to undiluted emissions from vehicles, vehicle ventilation systems, driving circumstances, and local dispersion characteristics (Fruin et al. 2004; Marshall et al. 2003). These findings have potential exposure implications for populations that rely on mobile sources as part of their daily routine, that are active near mobile source environments, or that reside near mobile source infrastructure.

Mobile source-oriented studies in New York State have focused mainly on the New York City area. Maciejczyk et al. (2004) characterized ambient air quality in the South Bronx; several major interstate highways encircle the area, which is home to local industries and facilities that generate truck traffic. The investigators found that highways on all sides of the South Bronx, as well as other local sources, caused a measurable increase in residents' exposure to pollutants. Lena et al. (2002) measured $PM_{2.5}$ and EC on sidewalks and conducted simultaneous traffic counts in the South Bronx. Linear regression of site-specific mean EC concentrations on mean large truck counts predicted an increase of 1.69 $\mu g/m^3$ EC per 100 large trucks/hr ($r^2 = 0.84$). In contrast, mean concentrations of $PM_{2.5}$ exhibited only modest site-to-site variation. These findings reinforced earlier work by Kinney et al. (2000) along Harlem New York City sidewalks. The authors reported a fourfold variation of EC concentrations across sites associated with bus and truck densities on adjacent streets and a bus depot. The average concentrations of EC ranged from 1.5 to 6.2 $\mu g/m^3$. Less contrast was found for mean concentrations of $PM_{2.5}$, likely reflecting regional sources.

Kinney et al. (2002) also characterized levels of and factors influencing personal exposures to urban air toxics among high school students living in New York City. Personal, indoor, and outdoor concentrations of particle elements were generally similar, suggesting that ambient sources may have driven indoor and personal exposures for most elements. Researchers found temporal cycles reflecting synoptic weather patterns that influenced the larger region. They also observed a substantial "urban influence" of traffic-related VOCs and reported considerable inter-home spatial variability suggestive of differential impacts of local sources. These findings reinforce the need to consider how population activity and behavior patterns contribute to the magnitude, duration, and intensity of exposures in densely populated areas. Such needs are especially relevant for urban areas characterized by elevated carbonaceous $PM_{2.5}$ concentrations, proximity to large pollution sources, or terrain features conducive to poor dispersion of pollutants.

A limited number of studies have begun to evaluate the effect of mobile source emissions on health outcomes in New York State populations. Lin et al. (2002) investigated whether pediatric hospitalization for asthma was related to living near a road with heavy traffic in Erie County, excluding the city of Buffalo. They found that children hospitalized for asthma were more likely to live within 200 m of roads in the highest third of vehicle miles traveled (odds ratio 1.93, 95% CI 1.13-3.29). Subjects were also more likely to have trucks or trailers passing within 200 m of their residence compared to controls (odds ratio 1.43, 95% CI 1.03-1.99). The study did not include Buffalo residences because of the lack of complete local traffic data; air pollution from local urban traffic may be more important than that from state routes. In a

Buffalo neighborhood downwind of the Peace Bridge complex and freeways, Lwebuga-Mukasa et al. (2004) found an association between increased commercial traffic volume and increased health care utilization for asthma. Oyana et al. (2004) reported a clustering of asthma cases in close proximity to the complex and freeways. The investigators showed a decreased dose-response relationship indicating a decreased risk of asthma prevalence the farther an individual resided from the source of exposure. Following a 50% drop in total traffic at the Peace Bridge in September 2001, Lwebuga-Mukasa et al. (2003) found a decrease in health care utilization for respiratory diseases.

In New York City, Lipfert et al. (2006) used traffic density as a surrogate measure of environmental exposure for major U.S. cities in their study of long-term mortality in a large cohort of U.S. veterans. The researchers estimated traffic volume using U.S. EPA data for annual vehicle-km traveled by county, finding that Manhattan had the highest traffic density of all U.S. counties, followed by three other New York City boroughs. The analysis found that living in a high-traffic county was associated with reduced life expectancies, but it could not identify which aspect of the heavy traffic was to blame. Possible variables cited included air pollution, noise, psychological stress, socioeconomic status, or some combination of factors.

The New York State Department of Health performed a study of air pollutants and acute asthma exacerbations leading to emergency room visits in communities in the Bronx and Manhattan in New York City (NY DOH 2006). Preliminary measurements found $PM_{2.5}$ and EC concentrations peak during morning hours, with higher weekday daily averages than weekends (Luttinger and Wilson 2003). DeGaetano and Doherty (2004) reported similar $PM_{2.5}$ findings in their temporal and spatial analysis of hourly $PM_{2.5}$ concentration extremes in New York City. The authors found significant diurnal, seasonal, and day-of-week variations, with the highest concentrations typically between 7:00-9:00 a.m., during summer, and on weekdays. Lowest concentrations were generally between 4:00-6:00 a.m. in winter or weekends. Summertime meteorological conditions favoring Bermuda high-pressure systems and calm winds in wintertime led to high $PM_{2.5}$ concentrations.

Analysis of Selected Populations Residing Near Major New York Roadways. The review of mobile-source related studies suggests that mobile source emissions can present a potential health risk to populations in close proximity to elevated emissions. Most exposure studies have found that urban populations live in close proximity to mobile source emissions. Few studies, however, have assessed rural population exposure. In New York State, inventory analysis has found that mobile source emissions comprise a large fraction of carbonaceous $PM_{2.5}$ emissions in urban and rural areas. In addition, demographic analysis has found that high population densities occur in New York urban and nonurban areas. These findings suggest the need for more comprehensive mobile source exposure assessment of New York State populations. In response, a mobile source case study was conducted for New York State. As part of the case study, NESCAUM analyzed population residence and traffic volume data for selected areas of the state.

This analysis generated a screening-level estimate of the number of New York State residents in selected rural, suburban, and urban counties living in close proximity to major roads, and whether these roads experienced high traffic volume. U.S. Bureau of the Census (2000 Census), Mid-Atlantic/Northeast-Visibility Union (MANE-VU) and New York State Department of Transportation (DOT) data sets were analyzed using Arcview GIS version 3.1. Such an approach is insufficient to understand the larger exposure milieu, which includes pedestrian and in-vehicle exposure, as well as proximity to other mobile source infrastructure such as commuter transit terminals. The analysis also does not directly connect proximity and traffic volume estimates to measured pollutant concentrations or health effects.

To estimate population counts, 2000 Census Block Centroid Populations and Major Roads data sets were analyzed using ESRI Data & Maps 2002 (http://www.esri.com/). Spatial calculations were restricted to persons living within 150 m and 300 m of major roads, defined by Census Bureau Feature Class Codes (CFCC) as primary and secondary (including interstate, state, and county highways, access ramps, and arterials). The selected buffer distance was based on health studies finding statistically significant health effects within these distances alongside high traffic corridors in urban areas, as well as exposure studies finding high concentrations of mobile source indicator pollutants within these distances. Population estimates were generated by counting the number of people living in 2000 Census blocks whose centroid populations fell within 150 m and 300 m. Census blocks are the smallest geographic entities within a county for which the Census tabulates population.

As shown in Table 2-8a and Figure 2-3, a large number of persons live within 150 m and 300 m of major roads in urban, suburban, and even rural counties. Nearly all residents of three urban counties live within 300 m of major roads (88-100%). In New York County, 95% of residents live within 150 m (5% within 150-300m), whereas in Queens County, 28% live within 150 m (60% within 150-300 m). In Figure 2-3, the map displays block-level population numbers and 150 m and 300 m buffer distances from major roads. Ninety-one percent of the county's population lives within 300 m of major roads. In Table 1-8a, 72-74% of persons live within 300 m of major roads; in both counties 43% of persons live within 150 m. In three rural counties, 24-41% of persons live within 300 m of major roads; similar proportions live within 150 m and 150-300 m.

Table 2-8a-b. Number of New York State Residents by County and Within 150 m and 300 m of Major Roads.

a. Percent Total Population

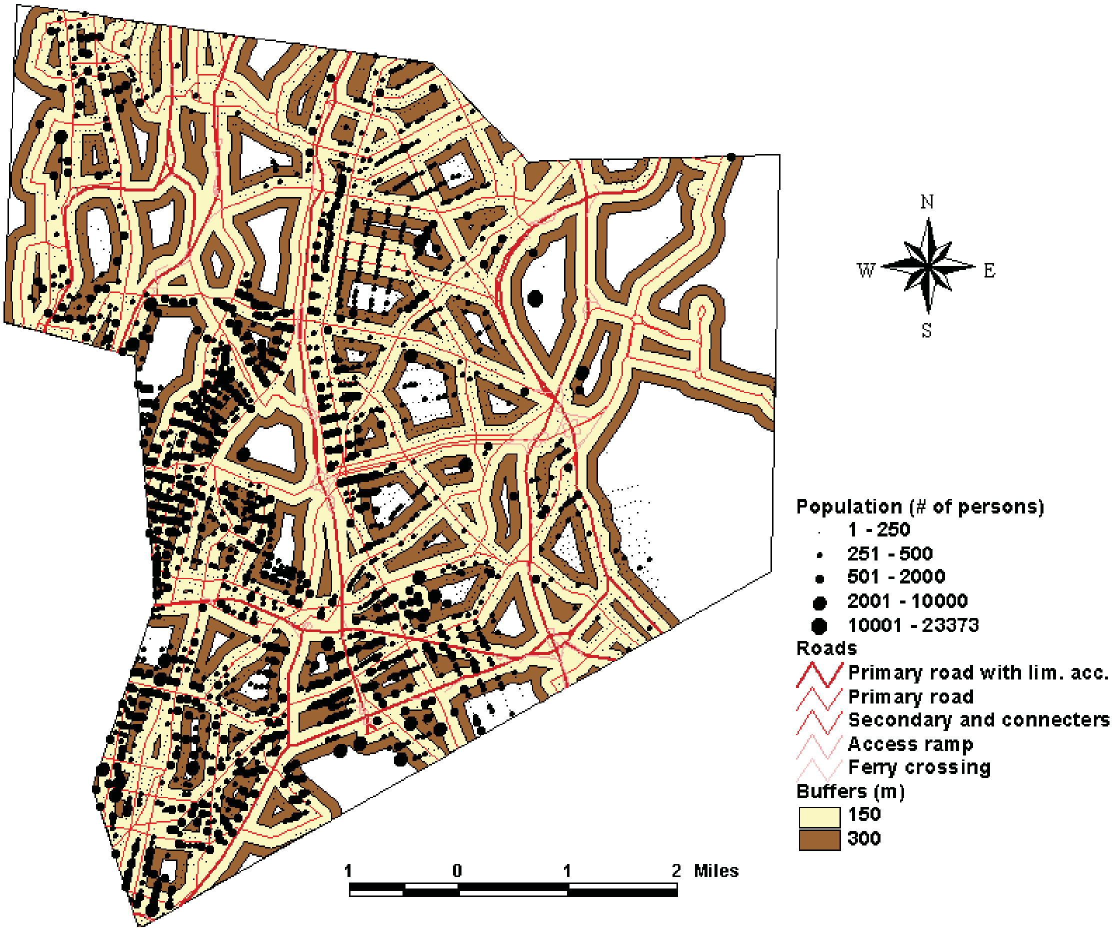
NYS counties	Population (%)			
	entire county (number)	residential distance to major road		
		<150 m	150-300 m	<300 m
Urban				
Bronx	1,074,102	66%	25%	91%
New York	1,571,892	95%	5%	100%
Queens	2,120,944	28%	60%	88%
Suburban				
Nassau	1,284,267	43%	30%	74%
Westchester	857,975	43%	29%	72%
Rural				
Cattaraugus	85,702	22%	19%	41%
Greene	47,851	17%	12%	28%
Lewis	26,926	13%	11%	24%

b. Population Density

NYS counties	Population density (persons/sq k)			
	entire county	residential distance to major road		
		<150 m	150-300 m	<300 m
Urban				
Bronx	11,044	12,855	11,388	12,411
New York	28,353	30,166	13,971	28,661
Queens	8,277	4,433	19,933	9,446
Suburban				
Nassau	1,933	2,029	2,429	2,177
Westchester	740	1,160	1,129	1,147
Rural				
Cattaraugus	26	116	106	111
Greene	31	83	62	73
Lewis	9	51	45	48

Table 2-8b provides the same data in the form of population density values, showing high densities in the 150 and 300 m buffer areas. In the three urban counties, population densities within 300 m were nearly identical to total county density. This finding was expected because a network of major roads comprised most of the area within the urban counties. Population densities within 150 m and 150-300 m varied considerably. In the urban counties, the buffer densities and countywide densities were considerably larger than the suburban and rural county densities, even though both the suburban Nassau and Westchester counties had high population numbers. In Westchester County, the population density within 300 m was nearly two-thirds greater than the county's overall density, while in Nassau County the buffer and countywide densities were similar. Densities within 150 m and 150-300 m were similar for the two counties. Densities in the three rural counties within 150 and 300 m were greater than for the entire counties; the population density within 300 m was two to five times greater than total county densities. Population densities within 150 m and 150-300 m were also similar.

Figure 2-3. Number of Persons at Census Block-Level Living Within 150 m and 300 m of Major Roads in Bronx County, New York.



Two analyses estimated traffic volumes on major roads. The first analysis used MANE-VU 2002 onroad Vehicle Miles of Travel (VMT) data for all major roads in the eight selected counties. The New York State Department of Environmental Conservation (DEC) originally prepared these data for submission to the U.S. EPA as a requirement of the Consolidated Emissions Reporting Rule (CERR) (MARAMA 2005). Vehicle class data were comprised of light-duty and heavy-duty gasoline, and light-duty and heavy-duty diesel. Road designation data were compiled from MOBILE-6 vehicle emission modeling software. Aerometric Information Retrieval System (AIRS) AMS road class codes were mapped to MOBILE-6. Analysis was confined to all CFCC major roads. These data do not provide pertinent information on driving behavior that could influence traffic emissions, such as idling, speed, and acceleration characteristics.

Table 2-9 displays VMT per year by vehicle class on major roads in selected urban, suburban, and rural counties. Across the counties, light-duty gasoline vehicles represent the most miles traveled (84-95%); light-duty diesel vehicles contribute the least number of miles (1%). The number of vehicle miles generated by each class was roughly proportional across the eight counties, although rural counties have proportionately less light-duty gasoline VMT (84-90%) and more heavy-duty gasoline and heavy-duty diesel VMT (4-7%). In suburban and urban counties, light-duty gasoline vehicles generated 92-95% of the VMT and heavy-duty vehicles 2-4%. Light-duty diesel VMT was identical across counties (1%). Within each grouping of counties, overall VMT were variable. For example, rural Cattaraugus and Greene Counties had about three to five times more VMT across vehicle classes than rural Lewis County. Urban Queens County had approximately double the number of VMT than urban Bronx and New York Counties.

Table 2-9. Annual Vehicle Miles Traveled on Major Roads in Selected New York State Counties.

NYS counties	Vehicle miles traveled (10^6 miles per year)			
	light duty gas	heavy duty gas	light duty diesel	heavy duty diesel
Urban				
Bronx	3,724	66	27	100
New York	3,849	74	29	111
Queens	7,074	127	51	197
Suburban				
Nassau	8,488	356	84	309
Westchester	7,277	330	67	274
Rural				
Cattaraugus	621	53	8	53
Greene	610	33	7	34
Lewis	194	8	3	10

A notable finding is that annual VMT for the four vehicle classes are greatest in the suburban Nassau and Westchester Counties. The urban Bronx and New York Counties (1.1 and 1.6 million persons) have about 50% of the gasoline and diesel traffic volume of suburban Westchester County (0.86 million persons) and 42-44% of the volume of suburban Nassau County (1.3 million persons). Urban Queens County (2.1 million persons) has about 81% less vehicle traffic than suburban Nassau County, whose population is about 40% less. Queens County has about 94% of the vehicle traffic of Westchester County, whose population is about 60% less than Queens. Rural area VMT are substantially lower than suburban and urban VMT. Average VMT per person was derived by dividing total county VMT by total county population. The values for the three urban counties range from about 2,500-3,000 VMT/person. The two suburban counties range from about 7,000-9,000 VMT/person. Two rural counties average about 8,000 VMT/person, while the third county (Greene) averages about 14,000 VMT/person.

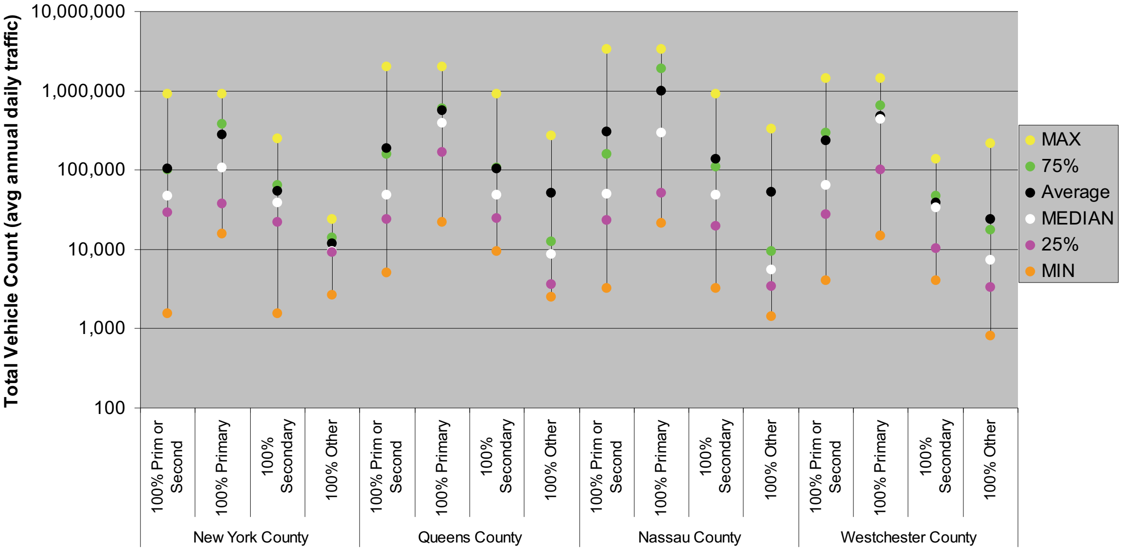
The second analysis integrated New York State DOT traffic volume data (www.nysdot.gov) for sampled roadways with New York State GIS street segment data (http://www.nysgis.state.ny.us/) for two urban counties (New York and Queens) and two suburban counties (Nassau and Westchester) from the eight-county analysis. DOT traffic data provided average annual daily traffic (AADT) volume reports for a subset of roads by county for various years ranging from 1992 to 2003. This metric estimates daily traffic volumes for specific roads over one year using short-term counts from individual sites. These data are adjusted for vehicle mix, day of week, and seasonality factors assembled from vehicle classification and continuous count stations.

An attempt to compare DOT road data characteristics for CFCC-defined major roads with the preceding population count analysis used identically named major roads as the common linkage between the DOT and ESRI data sets. This exercise was complicated because DOT can conduct multiple traffic counts at individual locations or segments along the same road. Each of these segments may fall within a different road classification (i.e., major or nonmajor) even though the road name does not change. To account for this, New York State Public Roads Digital GIS File Information was used, which facilitated a detailed comparison across road classifications in each of the counties analyzed.

For all major and nonmajor roads (i.e., the entire universe of roads) in each county, five classifications were developed. Major roads were classified as 100% primary, 100% secondary, or 100% primary or secondary (i.e., a combination of both). Roads not falling in any of these categories were classified as “100% other.” A road was defined as 100% if every segment along that road fit into the same category (primary, secondary, primary and secondary, or other). Roads consisting of multiple segment types (i.e., primary and/or secondary and other categories) were classified as “mixtures.” Next, all DOT sampled road segment traffic count data were sorted into these classes, excluding mixture roads to avoid including traffic count data on roads neither primary nor secondary along their total length. This facilitated a general comparison of counts for vehicles and population buffer areas (up to 300 m from roadways) within the same major road classifications across the DOT and ESRI data sets. However, the comparison applies only to aggregate traffic county data within each road classification, not to individual roads.

Figure 2-4 displays the results as summary statistics on a logarithmic scale. For the four counties analyzed, at least 75% of all primary and/or secondary roads had AADT vehicle counts greater than 10,000. Across the four counties, median primary road AADT vehicle counts are higher than secondary and other road category vehicle counts. The median 100% primary road AADT counts for the suburban Nassau and Westchester Counties (290,000 and 434,000, respectively) are similar in magnitude to New York and Queens Counties (108,000 and 384,000, respectively). With the exception of Nassau County (median value 503,000), roads classified as 100% primary or secondary in the four counties had substantially lower AADT road counts, similar in number to 100% secondary counts, ranging from 33,000 to 63,000 across both categories for all the counties. In all the counties, all median AADT values for roads classified as 100% other were less than 10,000.

Figure 2-4. Average Annual Daily Traffic Counts on Major Roads in Selected New York Counties.



The total length of roads for each class within the entire universe of roads was also calculated, and compared this to the total length of road segments sampled by DOT. Across the four counties, DOT sampling averaged 24% of all "100% primary or secondary" major roads for which population buffer areas were calculated. Sampling of nonmajor roads within the "100% other" category for all the counties was <4%. Roads classified as mixtures accounted for an average of 43% of all roads in the two urban counties, and 14% of all roads in the two rural counties sampled by DOT. It is possible that DOT included major road traffic counts in this class of roads that our analysis therefore did not incorporate. These findings indicate that traffic counts were conducted on at least one-quarter of the major roads averaged across the four counties.

NATIONAL AMBIENT AIR QUALITY STANDARDS FOR PM

The promulgation of National Ambient Air Quality Standards (NAAQS) represents a central regulatory effort to control ambient air pollution in the U.S. Of all the criteria pollutants, PM is the only one whose NAAQS is not chemically specific. Determining the most appropriate means to regulate PM as a group, as opposed to singling out components or classes of PM (e.g., sulfates, nitrates, transition metals, organic compounds) has long concerned scientists involved in the standard setting process (e.g., Friedlander and Lippmann 1994).

Monitoring programs and federal regulations implemented to limit the deleterious impacts of PM pollution currently measure PM on the basis of particle size range and mass concentration, using selected particle size indicators based on human health considerations and visibility impairment. From a health sciences standpoint, particle size remains an important determinant of health-related impacts. Particle composition is generally considered to be important as well, but it has been far more difficult to characterize and quantify its related health risks. As toxicological, clinical, and epidemiological studies start to make use of speciated data, it is possible that the next NAAQS review for particulates will consider specific components in addition to size range and mass concentration indicators. Such considerations may question whether controlling PM mass would sufficiently control carbonaceous PM. The next NAAQS could also recognize the possibility of combined effects of PM and gaseous co-pollutants; this would address realistic atmospheric mixture exposures reflecting the potential for overall mixture toxicity (Bachmann 2004; NRC 2004; Ross 2006).

Current Standards for PM

The NAAQS for PM is based upon the pollutant indicator, averaging time, numerical level of the indicator, and the statistical form of the standard. Over the past 30 years, as understanding of PM has advanced and new monitoring information has become available, the U.S. EPA has increasingly considered $PM_{2.5}$ as a separate air pollutant class from coarse thoracic particles (PM_{10}-$PM_{2.5}$). The agency developed two separate NAAQS for ambient PM based in part on epidemiological relationships between health effects and PM concentrations as measured with existing monitoring methods. The first NAAQS for PM was set in 1971 and targeted total suspended particulate (TSP) mass per unit volume of air. In 1987, the U.S. EPA changed the indicator from TSP to PM_{10}. The 1987 standards were set to protect against human health effects associated with 24-hour and annual exposure to particles that are more likely to reach the lower part of the human respiratory tract (i.e., the tracheobronchial region and the alveolar or gas-exchange region). The PM_{10} standard, like the TSP standard, was based on mass without regard to chemical composition.

On September 21, 2006, the U.S. EPA revised the $PM_{2.5}$ NAAQS established in 1997 and promulgated new standards for fine and coarse particles. The newly adopted primary $PM_{2.5}$ 24-hour and annual average standards are 35 $\mu g/m^3$ (98th percentile form) and 15 $\mu g/m^3$, respectively (U.S. EPA 2006a, b). The review process emphasized the importance of findings of health effects associated with acute and chronic exposure to $PM_{2.5}$ mass concentrations, including those characterized in time-series and cohort epidemiological studies (Pope and Dockery 2006). The U.S. EPA's justification was in large part attributable to finding stronger associations with adverse health effects for $PM_{2.5}$ than for PM_{10} (Vedal 1997). The effects of $PM_{2.5}$ on respiratory and cardiovascular systems are consistent with the concept that particles smaller than 2.5 µm in aerodynamic diameter penetrate more deeply into the gas exchange region of the respiratory tract, which provides direct access to systemic blood.

Potential Importance of Carbonaceous $PM_{2.5}$ in Future PM NAAQS Reviews

The NRC Committee on Research Priorities for Airborne Particulate Matter has concluded that information relating particle characteristics to their potential health risk remains largely incomplete and represents a critical research gap. Identifying the relative toxicity of PM components might inform the setting of future PM NAAQS, enabling the regulation of sources emitting the most toxic material. It is possible that additional research might draw attention not only to individual PM components, but also to combinations of pollutant components. Multiple pollutant combinations, including co-varying gaseous pollutants, may produce multiple effects by different biological mechanisms compared to individual pollutants or components. These considerations present substantial challenges that caution against any expectation of finding a single particle source category or component whose regulation would eliminate the adverse health effects associated with $PM_{2.5}$ pollution.

Rather than assuming the same health risk from all PM, findings of greater toxicity relating to specific physicochemical characteristics may lead to the formulation of standards based on different metrics, such as the choice of the indicator and averaging time. Were investigators to conclude that carbonaceous components are the key driver of ambient $PM_{2.5}$-related adverse health effects, the need to reduce sources such as combustion-derived mobile source emissions would become paramount. New NAAQS would require additional methods of exposure assessment, emissions characterization, and air quality models to formulate state implementation plans.

CONCLUSION

A large number of health effect studies indicate that carbonaceous $PM_{2.5}$ presents a public health concern. Toxicological, clinical, and epidemiological research has to varying degrees investigated the potential role of individual carbonaceous constituents, carbon mixtures and surrogates, emissions sources, and particle size modes that are rich in carbonaceous $PM_{2.5}$. Research provides evidence in the form of biological responses measured in animal and clinical inhalation studies, as well as adverse health outcomes demonstrated among human populations. Several epidemiological studies point to $PM_{2.5}$ or emissions sources rich in carbonaceous aerosol as important determinants of health-related responses. Toxicological findings confirm plausible mechanisms for ambient air $PM_{2.5}$-associated health effects found in epidemiological studies.

These studies do not provide conclusive evidence pointing to which of the many carbonaceous PM characteristics are most responsible for driving adverse heath effects. These characteristics include chemical composition, number concentration, particle size, and surface area. Nor do they rule out other $PM_{2.5}$ characteristics and components, whose assessment was not within the purview of this report. Despite differences in composition, health effects associated with $PM_{2.5}$ exposure have consistently been found in urban areas worldwide. This suggests that a single etiologic toxic agent—such as the general class of carbonaceous particles—may not be responsible for the entire spectrum of adverse outcomes. It is likely that different agents may contribute to health effects, and carbonaceous $PM_{2.5}$ emissions sources may be a significant contributor to some or all of these.

Many New York State populations are at potential risk from exposure to combustion sources rich in carbonaceous $PM_{2.5}$ emissions. Emission inventories find that certain source categories dominate rural and urban counties in New York. In three rural counties, residential wood combustion generates most of the carbonaceous $PM_{2.5}$ emissions and comprises over half of total $PM_{2.5}$ emissions. Mobile sources, especially diesel engines, account for only a small portion of carbonaceous $PM_{2.5}$ in those counties. In the tri-state New York City nonattainment area, mobile sources, likewise dominated by diesel engines, generate just over half of all carbonaceous $PM_{2.5}$ emissions and account for over a quarter of total $PM_{2.5}$ emissions. Commercial charbroil meat cooking and residential wood combustion each comprise a significant contribution of carbonaceous $PM_{2.5}$ in the tri-state New York City area. Source inventories show that a majority of New York State's total mobile source emissions (nonroad and onroad) come from nonurban counties outside the New York City five-county area, excluding counties that are home to Albany, Buffalo, Rochester, and Syracuse. The non-urban counties also contribute predominately to overall residential energy emissions for the State.

The large contribution of residential wood combustion to statewide carbonaceous $PM_{2.5}$ emissions raises potential public health concerns. Toxicological and epidemiological studies have linked pollutants in wood smoke to a variety of adverse respiratory health effects, especially in children. During wintertime, such emissions pose an even greater exposure risk to populations, especially in residential areas where terrain and meteorology contribute to poor dispersion of pollutants. In spite of these findings, exposure assessment, measurement, and health studies of residential wood combustion within nonurban areas of New York have been limited.

This section's mobile source case study reviewed current health research associating exposure to mobile source emissions containing carbonaceous aerosols with a variety of adverse health effects. Mobile source exposure studies have demonstrated that meteorological, geographic, and source-oriented factors influence carbonaceous component concentrations over time and over local and regional areas. In New York City neighborhoods, populations can be routinely exposed to high levels of mobile source emissions. In one nonurban county in western New York State, proximity to a state road was associated with pediatric asthma hospitalizations. Such studies indicate that mobile source emissions can give rise to exposure conditions of potential health concern among urban and nonurban New York populations. Research has not definitively shown that the carbonaceous $PM_{2.5}$ fraction, or other co-pollutants characterizing mobile source emissions, is uniquely responsible for observed health effects. Nor have researchers ruled out other possible causal explanations besides air pollution. Overall, more research is needed to characterize mobile source exposures in urban and rural areas of New York.

Demographic factors suggest health risks not only from substantial residential wood and vehicle emissions, but also from smaller carbonaceous $PM_{2.5}$ source categories. Population attributes—including population density, residence location, behavior, and activity patterns—could increase potential exposure at specific locations and during specific periods of the day, week, and season in rural, suburban, and urban areas. Additional sources of possible exposure include airport, ferry, harbor, and railroad corridors and terminals; in-station, in-cabin, or on-board environments; and portable internal combustion engines such as lawn mowers. This area requires additional exposure assessment research in New York State.

A screening-level analysis was conducted of residential population proximity to and traffic volume on major roads in selected New York State counties. The analysis demonstrated that a large number and high density of New York populations in rural, suburban, and urban counties live within 150 and 300 m of major roads. Traffic volume estimates found that major roads have high vehicle miles traveled and average annual daily traffic counts. These findings suggest that New York State populations are at potential risk to exposure to vehicle traffic emissions. Residence near major roads, however, represents only one of several possible exposure pathways for New York populations. Other exposure to mobile sources, including pedestrian, in-vehicle, and in-station environments, requires further research.

Finally, the current NAAQS for $PM_{2.5}$ is based upon the pollutant indicator, averaging time, numerical level of the indicator, and the statistical form of the standard. The relative toxicity of particle components needs to be identified before the NAAQS can be revised to regulate the sources emitting the most toxic components. More information is also needed on multiple pollutant combinations, including co-varying gaseous pollutants, and whether they produce multiple effects by different biological mechanisms compared to individual pollutants or components. These considerations present substantial challenges that caution against any expectation of finding a single particle source category or particle component whose regulation would eliminate the $PM_{2.5}$ pollution problem.

Information Needs and Research Gaps.

Health Effects Research Needs. Recent review papers, workgroups, and funding on the topic of health research relating generally to PM constituents or specifically to carbonaceous $PM_{2.5}$ provide comparable findings (e.g., HEI 2005; Lippmann and Ito 2000; Mauderly et al. 2004; NRC 2004; Schlesinger 2004; Schlesinger et al. 2006; Sheppard 2004). The suggestions for future research on health effects include:

- Future research efforts should draw upon a combination of toxicological and epidemiological approaches. Such multidisciplinary study designs should consider a range of health-related measures to improve understanding of the different effects of a variety of components. These suggestions apply not only to analysis of carbonaceous $PM_{2.5}$, but also to mixtures of air pollutants, including gaseous air pollutants.

- More specific pollutant monitoring data are needed. Current data resources such as the EPA's $PM_{2.5}$ Chemical Speciation Trends Network (STN), for example, might have sufficient temporal resolution to support time-series epidemiological studies, but they provide insufficient spatial resolution to support cohort epidemiological studies. When using speciated concentration data, health researchers should be aware of the possible uncertainties that characterize various data sources.
- Toxicology studies should endeavor to use comparable research protocols and biological endpoints to help determine which components of different pollutant mixtures are more or less harmful.
- Future PM toxicology research should provide better understanding of how environmentally relevant exposure conditions affect health.
- A systematic approach is needed to link and integrate epidemiological investigations across different study methods, metrics, and approaches to ensure better comparison and compilation of findings for the many pollutant constituents, mixtures, and exposure scenarios studied.
- Future epidemiological research should address how measurement errors might affect exposure-response findings. It should investigate whether there is significant spatiotemporal variation by species relative to total mass within and across monitored areas. It should identify whether current exposure assessment and statistical methods are of suitable design and estimation rigor, and how to determine the number and location of samplers necessary to investigate exposure effects.
- Methods to estimate health effects by using pollutant source study techniques hold promise but are still analytically in formative stages and should be improved.
- More assessment is needed to characterize population exposures to a variety of sources rich in carbonaceous material in rural, suburban, and urban areas. Greater understanding of the effect of complex terrain, meteorological conditions, and atmospheric processes on physicochemical PM composition and concentrations is needed. Research should include the specific investigation of temporal and spatial variability across populations, especially in specific locations where source dominance and population attributes could influence health.
- Concentration differences in carbonaceous $PM_{2.5}$ between monitoring sites in populated areas require more accurate resolution to guide future exposure assessments and health effects investigations. Investigators should continue epidemiological research using proximity measures and intra-urban exposure contrasts.

New York State Research Needs. A review of health and exposure findings leads to the following specific research needs for New York State:

- Because carbonaceous $PM_{2.5}$ concentrations can vary across short distances and over small time periods, methods to assess exposure variability should move beyond the use of central-monitor NAAQS compliance data alone. Daily (24-hour) and annual averaged ambient air quality data collected at central sites for current regulatory compliance in rural and urban areas may not reflect real and potentially meaningful population exposures to $PM_{2.5}$ sources rich in carbonaceous material.
- A greater understanding of population activity and behavior patterns is needed to determine how day-to-day actions of general and susceptible populations contribute to the magnitude, duration, and intensity of exposures to carbonaceous $PM_{2.5}$ sources, especially in densely populated areas (both nonurban and urban) in proximity to large local sources. Exposure should be assessed for:
    1. populations in proximity to residential wood combustion emissions (including outdoor wood boilers), especially in densely populated villages, towns, and small cities, and areas where meteorology and terrain are conducive to pollutant loading;
    2. populations in proximity to commercial meat-cooking establishments; and
    3. populations in proximity to mobile sources, including vehicles and airport, ferry, harbor, and locomotive corridors and terminals, as well as in or onboard aircraft, ferries, locomotives, and vehicles.
- The potential relevance of the above research to public health and preventative measures should be considered in the future determination of appropriate $PM_{2.5}$ standards and pollution control measures.

Future NAAQS Reviews and Public Health in New York. With respect to future NAAQS reviews for PM and possible effects of carbonaceous $PM_{2.5}$ on public health in New York, a review of health and exposure findings leads to the following general findings:

- Policy makers should reach agreement on a common research agenda. Calls for funding large-scale research of PM components, on the one hand, and multiple pollutant combinations on the other, should be reconciled toward more efficient standards setting. The National Research Council's Committee on Research Priorities for Airborne Particulate Matter Research Topic 5 calls for an assessment of hazardous PM components, envisioning a matrix approach to evaluate PM components and health responses. The committee's Research Topic 7 calls for research into the combined effects of PM and gaseous pollutants, envisioning a shift in focus from PM to a multipollutant research program. Researchers are therefore challenged to develop a systematic program to assess toxicity of PM mixture components while also considering PM's health effects within the broader context of other pollutants present in ambient air.
- Until a more definitive understanding of particle and multiple-pollutant toxicity is reached, an interim public health management approach should attempt to integrate current knowledge of toxicity and health effects with additional study of exposure distributions and magnitudes to

inform source control options. One possible management approach would proceed in three chronological steps:

1. determine pollutants with known toxicity,
2. assess exposure distribution and magnitude in relation to sources of these pollutants, and
3. use this information to determine the most efficient implementation of local and regional controls to maximize public health benefits.

The framework could be informed by exposure assessment methodology and policy uses for exposure estimates as outlined, for example, by Smith (1995) and Brauer et al. (2002). Such an approach would recognize that $PM_{2.5}$ reductions have led to measurable health gains, and therefore efforts should be made to reduce emissions based on available evidence of health effects and exposure. For example, a decrease in mass-based $PM_{2.5}$ concentrations regionally across the central and eastern U.S. was associated with a reduction in adult mortality rates (Laden et al. 2006). A decrease in traffic volumes in Buffalo, NY, was associated with a decrease in health care utilization for respiratory diseases (Lwebuga-Mukasa et al. 2003).

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Section 3

CONTROL STRATEGIES TO REDUCE PRIMARY $PM_{2.5}$ EMISSIONS IN NEW YORK STATE

INTRODUCTION

This section addresses opportunities to reduce primary carbonaceous $PM_{2.5}$ emissions (elemental and organic) from two stationary source sectors and five mobile source sectors in New York State. They include: 1) Commercial Cooking, 2) Residential Fuel Combustion, 3) Light-duty Vehicles, 4) Heavy-duty Trucks and Buses, 5) Nonroad Engines, 6) Airports, and 7) Marine Ports. The section is divided into seven subsections that correspond to the categories listed above. Each subsection includes a sector profile, sector emissions estimates, a description of federal and state authority to address $PM_{2.5}$ emissions from the sector,[11] and technical and policy options for reducing $PM_{2.5}$ emissions from that sector. The major focus is on the New York Metropolitan $PM_{2.5}$ Nonattainment Area[12] (NAA), but other major urban areas in New York as well as selected counties representative of a rural environment are also addressed. [13] Table 3-1 presents the mass of carbonaceous emissions for the seven sectors evaluated in this report, plus two other general categories:

According to the data from the U.S. Environmental Protection Agency (EPA), the seven source categories evaluated in this section account for approximately 90% of primary carbonaceous $PM_{2.5}$ emissions in New York, for which speciation factors for primary $PM_{2.5}$ emissions for elemental and organic carbon are available.[14] Control strategies for the multitude of individually small source categories that collectively emit about 30% of total primary $PM_{2.5}$ in New York were not evaluated. This is because of lack of reliable speciation profiles for these hundreds of different types of stationary sources. Where available, the EPA's sector- or category-specific speciation profiles for primary $PM_{2.5}$ emissions were used to estimate the carbonaceous fraction of $PM_{2.5}$ emissions from that sector or category (EPA 2002). For outdoor wood boilers, we relied on inventory data developed by Mid-Atlantic Regional Air Management Association (MARAMA)[15] because EPA data are not available for this sector.

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[11] Rules were reviewed for the cities of Buffalo and Rochester, for Erie and Monroe Counties (which contain these two cities), for the New York counties in the New York City $PM_{2.5}$ nonattainment area, for New York State, and for Cattaraugus, Greene, and Lewis Counties (three rural counties representative of a rural environment). Of these, Bronx, Kings, New York, Queens, and Richmond Counties are coterminous with the five boroughs of New York City; New York City issues all the rules and regulations for these counties. New York State law has a specific provision that addresses the interaction of the New York Air Pollution Control Act and federal Clean Air Act requirements. No permit issued under the New York Air Pollution Control Act can include standards that are more stringent than federal requirements unless authorized by state rules (N.Y. Environmental Conservation, sec. 19-0302).

[12] The New York (tri-state) metropolitan nonattainment area refers to a 22-county area that includes the five New York City boroughs and 17 other counties in the states of New York, New Jersey, and Connecticut.

[13] The New York City metropolitan $PM_{2.5}$ nonattainment area encompasses the five boroughs of New York City; Nassau, Suffolk, Westchester, Orange, and Rockland Counties in New York, ten counties in northern New Jersey, and two counties in southern Connecticut.

[14] Detailed carbonaceous $PM_{2.5}$ inventory data for New York can be found in Appendix A.

[15] There are no EPA $PM_{2.5}$ emission factors for outdoor wood boilers. The NEI does not include outdoor wood boilers.

Table 3-1 Total Carbonaceous $PM_{2.5}$ Emissions (Tons/Year) New York State & NYC Metro Tri-State Nonattainment Area.

	New York State			NYC Metro Tri-State NAA		
Source Category	Organic Carbon	Elemental Carbon	Total Carbon	Organic Carbon	Elemental Carbon	Total Carbon
On-Road LD Gasoline	1597	416	2013	1409	359	1768
On-road HD Diesel	1722	2384	4106	1338	1860	3198
Non-Road Diesel	1397	4615	6012	1338	4419	5757
Airport	66	184	250	66	188	254
Marine	158	392	550	199	468	667
Residential Fuel Combustion	34,453	6637	41,090	3945	836	4781
Charbroiling	3987	50	4037	4473	46	4519
Other Mobile	3105	487	3592	2126	315	2441
Other Stationary	999	586	1585	936	376	1312
Total	47,484	15,751	63,235	15,830	8867	24,697

Figure 3-1 shows the split between primary carbonaceous and non-carbonaceous $PM_{2.5}$ emissions in New York State. The pie chart on the left represents primary $PM_{2.5}$ emissions. Not included in this pie chart is approximately 30% of primary $PM_{2.5}$ emitted from the multitude of small sources mentioned above. Of the primary $PM_{2.5}$ mass emissions in the State (minus miscellaneous stationary sources not speciated), 55% are carbonaceous (elemental carbon plus organic carbon).[16] The pie chart on the right in Figure 3-1 shows the carbonaceous primary $PM_{2.5}$ emissions apportioned into nine sectors. Control technologies for eight of these sectors are described in this chapter ("other mobile" is included in the section on nonroad engines). One sector, "other stationary" is not described in this section since emissions in this category primarily consist of non-carbonaceousPM emissions from coal-fired power plants.

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[16] An analysis of primary carbonaceous $PM_{2.5}$ in the NAA was also conducted but is not presented. The analysis showed the seven source categories also accounted for 90% of primary carbonaceous $PM_{2.5}$ in the NAA in 2002 for sources that could be speciated.

Figure 3-1. Primary $PM_{2.5}$ Emissions and Carbonaceous Detail - New York State Average (2002).

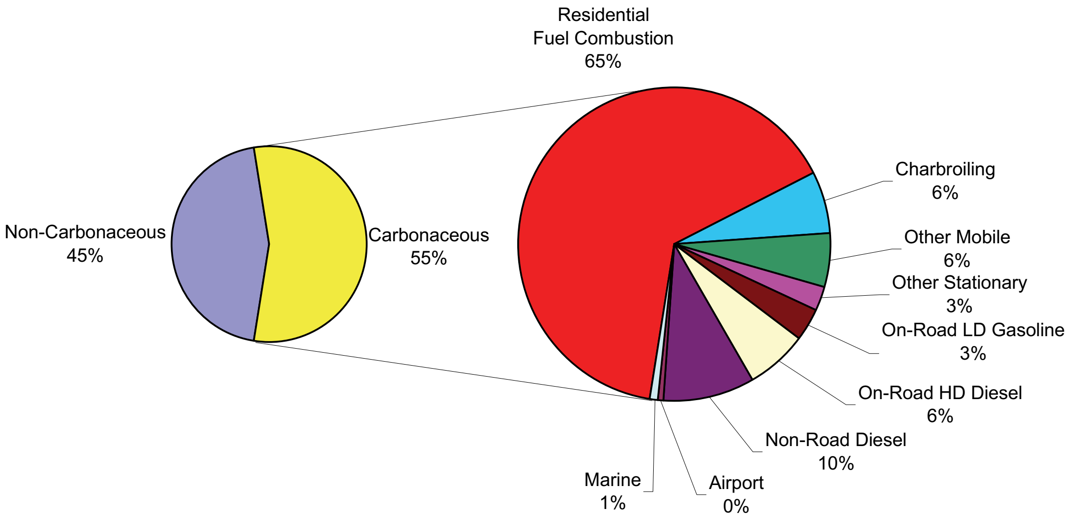
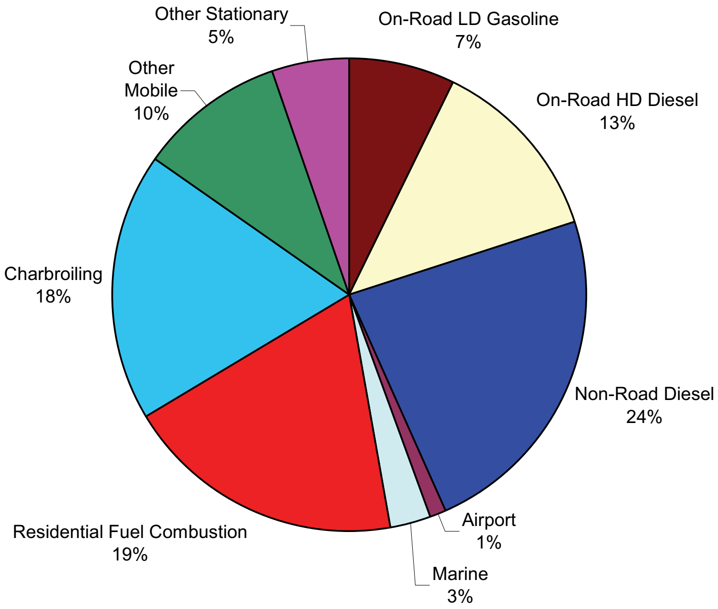


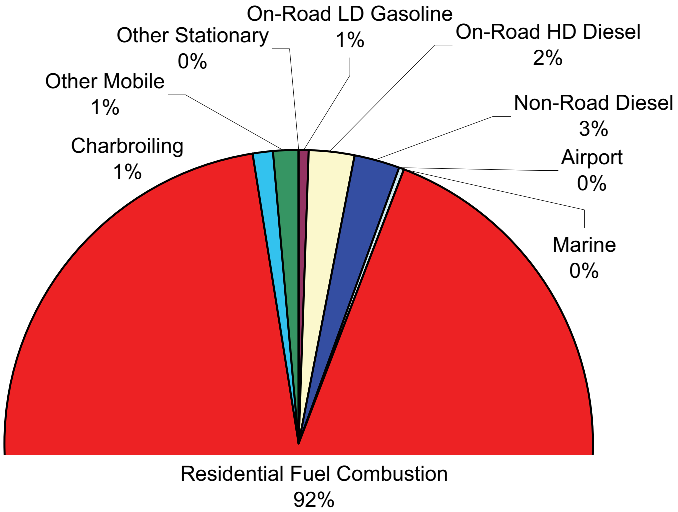
Figure 3-2. Total Carbonaceous $PM_{2.5}$ - NYC Metro Tri-State $PM_{2.5}$ Nonattainment Area.



The relative contribution to primary carbonaceous $PM_{2.5}$ from the various sectors is quite different in the nonattainment area (NAA) (Figure 3-2) as compared to rural areas (Figure 3-3). In the NAA, onroad and nonroad mobile source emission sectors (primarily emissions from gasoline- and diesel-powered engines) make up half of the carbonaceous $PM_{2.5}$ inventory.

Also noteworthy is the commercial charbroiling category, concentrated in population-dense urbanized areas and representing a significant fraction (almost 1/5 of the total) of overall carbonaceous emissions in the NAA. A large fraction of $PM_{2.5}$ from residential fuel combustion in the NAA comes from residential wood burning (over 90%). These emissions occur largely in the Long Island counties of Suffolk and Nassau. In contrast, the residential fuel combustion sector dominates carbonaceous $PM_{2.5}$ emissions in nonurbanized areas, accounting for more than 90% of the total, with the majority of the contribution coming from residential wood combustion.

Figure 3-3. Total Carbonaceous $PM_{2.5}$ - Three Rural Counties (Combined).



While carbonaceous emissions from diesel engines outside the NAA account for only about five percent of the upstate total from all source sectors, they account for almost 40% of total direct $PM_{2.5}$ emissions in the NAA.

Treatment of Secondary Organic Aerosols

In addition to primary organic carbonaceous $PM_{2.5}$, the measured ambient $PM_{2.5}$ is also comprised of secondary organic aerosols (SOAs) that are formed by atmospheric chemical reactions of volatile organic compounds (VOCs). SOAs are formed when atmospheric chemical reactions produce organic compounds that have low vapor pressures; as these low-vapor-pressure compounds saturate the gas phase, any further chemical production will result in aerosol production. Therefore, most SOAs are formed only from the oxidation of high-carbon-number VOCs, those comprised of six or more carbon atoms. This is because oxidation products of VOCs must have sufficiently low vapor pressures to enable the oxidation products to partition into the aerosol phase. Thus biogenic VOCs (for example, monoterpenes and sesquiterpenes) and aromatics (for example, toluene and xylenes) are important SOA precursors and lower-carbon-number VOCs such as propane, ethane, and propylene are not. The atmospheric science chapter of this report discusses secondary aerosol formation in detail.

Given the complexity of secondary formation processes, the emissions of VOCs from the seven categories evaluated in this report are not explicitly addressed. High-carbon-number VOC emissions are mostly attributed to mobile source categories (for example, aromatics from light-duty vehicles) evaluated in this study. Furthermore, the control options to reduce primary elemental and organic carbon emissions also address the reduction of VOC emissions.

The sections that follow describe the seven source categories noted above. They include sector emissions profiles and technical and policy options available to New York State for addressing primary carbonaceous $PM_{2.5}$ emissions from these categories.

COMMERCIAL MEAT COOKING OPERATIONS

This section discusses the technology and policy options available for reducing $PM_{2.5}$ emissions from commercial meat cooking establishments in the State of New York. The State has an estimated 58,000 eating-and-drinking establishments, 21,750 of which are located in New York City (National Restaurant Association 2006; NYC DOHMH 2006). This includes five-star restaurants, fast-food chain restaurants, bakeries, cafes, and the like. Many of these establishments generate particulate emissions when preparing food. According to the EPA, the majority of PM emissions from this sector come from the cooking of meat products (Roe et al. 2004), and PM emissions from meat cooking are primarily in the fine particulate ($PM_{2.5}$) size range (McDonald et al. 2003). In particular, meat charbroiling is estimated to generate over 80% of total $PM_{2.5}$ emissions from commercial meat cooking (Roe et al. 2004).

Commercial cooking operations produce a mixture of solid, liquid, and gaseous particles, including water and grease, noncondensable gases, and solid organic matter. Some portion of the water and grease vapor condenses on the ventilation equipment, while the remaining particles and grease droplets are released to the atmosphere. How much of the vapor is condensed rather than released to the atmosphere depends on a host of factors.

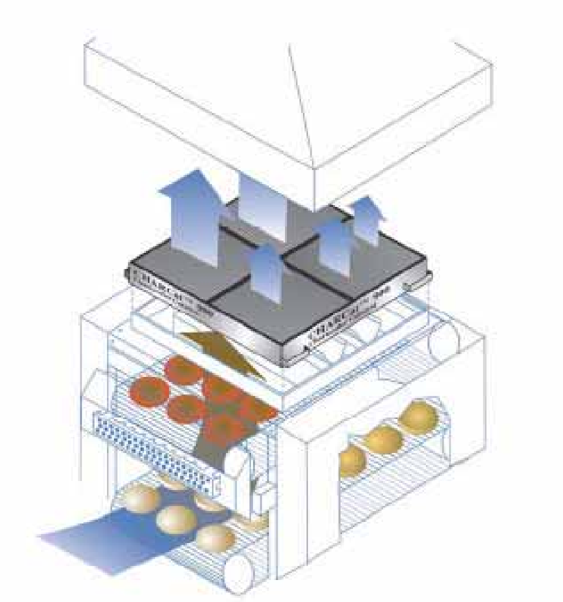
Sector Profile

A large share of New York's restaurants is located in the five boroughs of New York City. For example, for fast-food restaurants, the New York City Department of Health's online database of restaurant inspection reports lists 292 Subways, 251 McDonald's, 98 Burger Kings, 54 Wendy's, and 39 Pizza Huts (NYC DOHMH 2006).

The EPA's National Emissions Inventory (NEI) focuses on five basic categories of cooking equipment used in commercial meat cooking: (1) chain-driven (or conveyorized) charbroilers, (2) underfired charbroilers, (3) deep-fat fryers, 4) flat-griddle fryers, and (5) clamshell=griddle fryers (Roe 2004). Limited data are available on the number of these specific appliances currently in use in New York. Among the large chain restaurants, Subway, Quizno's, and Burger King all rely on chain-driven charbroilers (Engelhard 2006). McDonald's uses flat-griddle fryers to prepare its hamburgers.

Figure 3-4. Chain-Driven Charbroiler.

Courtesy Engelhard Corporation



Chain-driven Charbroilers. Chain-driven charbroilers (Figure 3-4), which fast-food restaurants often use, rely on a conveyer belt to carry the food product through the heat source. Often they cook both sides of the food in a single pass—making them faster than traditional, open grills. Most chain-driven charbroilers are fueled by natural gas, although electric models are also available (Perryman 1997).

Underfired Charbroilers. Underfired charbroilers resemble a barbecue and have a heat source below a heavy metal grid, which marks the food with a distinctive striping. Some restaurants make their charbroiler, which generates flames and smoke, a showpiece within their establishment (Fisher 2002). Underfired charbroilers generate particulate emissions when grease falls from the meat onto the radiant surface. Unburned drippings collect in a grease tray. Most underfired charbroilers burn natural gas, but some use charcoal or wood with ceramic stones to distribute the heat (Perryman 1997). Because of the smoke they generate, underfired charbroilers require significant ventilation (Fisher 2002).

Deep Fat Fryers. Deep fat fryers use an exposed hot metal surface to heat cooking oil, which is then used to cook the food. Fast-food establishments use deep fat fryers to prepare food in batches. Most raw food products contain a high percentage of water by weight (10–75%), and most of the water at the surface vaporizes during the cooking process, creating an oily mist (Perryman 1997).

Griddles and Griddle Hybrids. Flat griddles are appliances that consist of an exposed metal plate used to fry food. Most griddles are gas-fired, although some are electric. The Clamshell® griddle is a newer type that employs a two-sided cooking configuration to reduce cooking time, thereby also reducing emissions (Perryman 1997).

Sector Emissions

The $PM_{2.5}$ emission rates of commercial cooking appliances vary by type of meat, the fat content of the meat, and cooking conditions. High-fat meats cooked on an underfired charbroiler produce the highest $PM_{2.5}$ emissions and so dominate emissions from the sector. Regulatory efforts have focused on the control of emissions from chain-driven charbroilers, however, as they are more cost-effective to control (Perryman 1997). The $PM_{2.5}$ emissions rates for charbroiling hamburger, steak, and chicken range from 4.5 to 40.0 pounds (lb) per 1,000 lb of uncooked meat. High-fat hamburgers (25% fat) cooked with an underfired charbroiler produce the greatest amount of $PM_{2.5}$. In testing of meat cooking on a charbroiler (hamburger, steak, and chicken), the emissions were almost entirely composed of organic carbon (accounting for approximately 96% of the fine particle mass), with small amounts of metals and other elements and inorganic ions (McDonald 2003). Table 3-2 summarizes the $PM_{2.5}$ emissions rates of several meat/appliance combinations.

Table 3-2. PM2.5 Emissions Rates from Cooking Meat.

Source: McDonald 2003

	Fat Content	Appliance	$PM_{2.5}$ (per 1,000 lb of meat)
Hamburger	21%	Chain-driven Charbroiler	4.5–7.4
	25%	Underfired Charbroiler	15.0–40.0
	21%	Underfired Charbroiler	7.1
Steak	3%	Underfired Charbroiler	7.8
Chicken with skin	10-30%	Underfired Charbroiler	7.2–10.4

According to the EPA’s 2002 National Emissions Inventory, charbroiling (both chain-driven and underfired) generates over 4,000 tons of $PM_{2.5}$ emissions annually in New York State—accounting for 74% of $PM_{2.5}$ emissions from the commercial cooking sector.[17] This represents only 2.6% of total $PM_{2.5}$ emissions statewide. In the New York City metropolitan nonattainment area for $PM_{2.5}$ (including counties in Connecticut and New Jersey), charbroiling generated 4,598 tons of $PM_{2.5}$ or more than 9% of total annual $PM_{2.5}$ emissions within the area. (2,709 tons of these emissions are attributable to New York counties within the NAA.) The contribution of charbroiling to total $PM_{2.5}$ emissions in other New York urban areas is similar to the state average, and lower in rural areas (see Appendix A). By comparison, a California study found that meat-cooking operations produce 21% of fine *organic* particle emissions in the Los Angeles area (Rogge 1991). Particle emissions from meat cooking are primarily in the $PM_{2.5}$ size range (McDonald 2003) and, as noted earlier, consist almost entirely of organic carbon.

Among the sources within the commercial cooking sector, underfired charbroilers account for the largest share of emissions (74% statewide). Chain-driven charbroilers account for 10% of $PM_{2.5}$ emissions, and flat-griddle frying 15%. Table 3-3 summarizes the $PM_{2.5}$ emissions from the commercial cooking sector in New York State.

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[17] According to the EPA’s Speciate database, $PM_{2.5}$ from charbroiling is composed of approximately 97.3% organic carbon and 1.2% elemental carbon, and the total elemental carbon plus organic carbon is 98.5% of $PM_{2.5}$ emissions (EPA 2005g).

Table 3-3. New York State $PM_{2.5}$ Emissions from Commercial Meat Cooking (2002).

Source: U.S. EPA, National Emissions Inventory 2002

Source Category	New York State $PM_{2.5}$ Emissions	
	Tons	Percent
Chain-Driven Charbroiler	483	10%
Underfired Charbroiler	3,616	74%
Flat Griddle Frying	745	15%
Clamshell Griddle Frying	50	1%
Total	4,894	100%

Federal and State Authority

There are no federal regulatory requirements for controlling $PM_{2.5}$ emissions from restaurants; state, local, and municipal air agencies in the U.S. have generally not been active in regulating commercial cooking facilities. However, the South Coast Air Quality Management District (AQMD) in California regulates chain-driven charbroilers, as discussed below, and lists catalytic oxidizers in its best available control technology (BACT) guidelines for controlling PM_{10} emissions from these sources (Perryman 1997).

Neither New York State nor any of the counties or cities in the State whose requirements were reviewed for this study set numerical standards for air pollution emissions from commercial cooking facilities. However, many of these jurisdictions have requirements that are generally applicable to emissions from these facilities. For example, the New York State regulations that prohibit air emissions that are harmful or "unreasonably interfere with the comfortable enjoyment of life" are applicable to commercial cooking operations (NYS DEC sec. 211.2). The same is true of the New York State regulations that prohibit emissions above a certain opacity level (NYS DEC sec. 211.3). The New York City Administrative Code has similar prohibitions on "detrimental" emissions and on emissions that violate opacity limits (NYC AC secs. 24-141, 24-142). The Laws of Suffolk County (LSC sec. 76-1011, 76-1014) and the Code of the City of Buffalo (CCB secs. 66-2, 66-3, and 66-4) mirror these requirements. The Code of the City of Rochester also has applicable opacity limits (CCR secs. 100-1, 100-3).[18]

There is another category of requirements that, like these general nuisance-type and opacity rules, has caused some commercial cooking operations to install controls. These are the requirements that relate to ventilation, which arise out of human health and odor considerations, but also out of a concern for potential grease fires. For example, various provisions of New York City's Building Code and Fire Prevention Code, both contained within Title 27 of the New York City Administrative Code, require the ventilation of

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[18] There are differences, however, in specific opacity provisions. For example, NYS DEC sec. 211.3 prohibits emissions "having an opacity equal to or greater than 20% (six-minute average) except for one continuous six-minute period per hour of not more than 57% opacity." Buffalo prohibits smoke that is denser than Number 3 on the Ringelmann Chart (CCB secs. 66-2, 66-3), whereas Rochester prohibits smoke that is denser than Number 2 (CCR secs. 100-2, 100-3).

commercial kitchens (e.g., NYC AC secs. 27-758, 27-776, 27-4275). Reference Standard (RS) 13-1-3.1, an appendix to New York City's Building Code, requires that commercial cooking equipment that produces smoke or grease-laden vapors, such as broilers, be equipped with an independent exhaust system that includes a hood, a duct system, a grease removal device, and fire extinguishing equipment, all of which must meet the requirements specified in RS 13-1-3.1. Both Suffolk County and Buffalo have less detailed provisions that require ventilation to address cooking odors (LSC sec. 760-1364; CCB sec. 370-6).

Technical Options for Emissions Reduction

Most $PM_{2.5}$ emissions from commercial cooking establishments are generated when grease from the meat falls into the heating element or flame and combusts. The South Coast AQMD has regulated emissions from chain-driven charbroilers after finding that they were cost-effective to control. The South Coast AQMD chose not to regulate emissions from underfired charbroilers, which are the largest sources of emissions from commercial cooking, after conducting an extensive review and concluding that available controls are not cost-effective.

Chain-Driven Charbroilers. Chain-driven charbroilers, which generate 10% of the sector's $PM_{2.5}$ emissions in New York State, generate particulates when grease from the cooking meat falls onto the high-temperature radiant surface. The most common technology used to control these emissions is the flameless catalytic oxidizer (Perryman 1997). The system is mounted into the charbroiler ventilation duct and contains a bed of inert ceramic material coated with a metal catalyst. Smoke and gases from the cooking process are routed through the system, which oxidizes unburned PM and hydrocarbons to carbon dioxide and water. Independent test results suggest that properly maintained catalytic oxidizers can have PM removal efficiencies exceeding 90% (CE-CERT 1997). In experiments, emissions from a chain-driven charbroiler equipped with a catalytic oxidizer averaged 1.29 lb of PM emissions and 0.32 lb of VOC emissions per 1,000 lb of hamburger, reductions of 83% and 86%, respectively, from high-end estimates of uncontrolled emissions (Whynot 1999).

Leading up to and following promulgation of the 1997 South Coast AQMD rule, a large number of restaurants in southern California installed catalytic oxidizers. Their success at reducing emissions, grease buildup, smoke, and odors at a reasonable cost has increased the demand for catalytic oxidizers throughout the country. While only two other areas in the U.S. (Ventura County Air Pollution Control District and San Joaquin Valley United Air Pollution Control District) have adopted restrictions similar to those of the South Coast AQMD, a significant number of restaurants in other parts of the country (including restaurants in New York and New Jersey) have already installed catalytic oxidizers. In some cases, restaurants have installed them in response to nuisance complaints (Harned 2006).

In 1997, the cost-effectiveness of retrofitting catalytic oxidizers on commercial chain-driven charbroilers was estimated by South Coast AQMD staff to be $1,680–$2,800 per ton of PM and VOCs reduced. Restaurant operators, commenting on the South Coast AQMD's draft rule, reported costs of more than $7,000 per ton of emissions reduced (Perryman 1997). According to the South Coast AQMD, the higher cost estimate represented a unique outlier situation in which more extensive structural changes to the cooking facility were required (South Coast AQMD 1997b).

Recent capital cost estimates by manufacturers range from $1,200–$2,400, depending on the size of the catalyst (Engelhard 2006; Harned 2006). Site-specific retrofit issues can increase the cost of installation. Newer chain-driven charbroilers are specifically designed to accommodate catalytic oxidizers, allowing the system simply to be dropped into place (Harned 2006). For example, AyrKing Corporation manufactures a catalyst system designed specifically for the Burger King Nieco 9025 double broiler.

Catalyst systems require regular cleaning to maintain their effectiveness. AyrKing Corporation recommends that its catalyst be cleaned at least monthly by soaking it in warm water for a period of one hour, after allowing the system to cool (AyrKing 2006).

In a report by the Ventura County Air Pollution Control District (APCD), a restaurant owner in California reported a reduction in duct cleaning costs as a result of installing a catalyst system. According to the report, the catalyst saved the owner approximately $320 per year by reducing the need for duct cleaning from about four to about two times per year. In addition, because the catalyst reflects heat back into the charbroiler, natural gas use at the restaurant was reduced by about 7% (Ventura County APCD 2004a). These co-benefits, which will vary from location to location, effectively reduce the net cost of controlling emissions.

Underfired Charbroilers. Underfired charbroilers produce four times the emissions generated by chain-driven charbroilers per unit of product cooked. When cooking high-fat meats; they are estimated to generate 74% of the $PM_{2.5}$ emissions from the commercial cooking sector in New York State (Whynot 1999; Roe 2004; EPA 2006a). However, the South Coast AQMD, which has evaluated options for controlling PM_{10} emissions from underfired charbroilers since 1991, concluded in December 2004 that cost-effective control options are not available. According to that agency, catalytic oxidizers are not effective with underfired charbroilers because the exhaust temperatures are not sufficient to promote an efficient reaction (Perryman 1997).

In the South Coast AQMD, underfired charbroilers contribute 83 percent of the restaurant-sector PM_{10} inventory (South Coast AQMD 2004). The South Coast AQMD evaluated two control options: (1) the replacement of underfired charbroilers with Smokless^{TM} broilers, and (2) an add-on scrubber manufactured by CAST, Inc., but ultimately they did not adopt regulations (South Coast AQMD 2004).

In 1999, about 70 restaurants in the U.S. used the Smokless^{TM} broiler. This broiler is not a control device, but a complete alternative to the traditional underfired charbroiler. Unlike the underfired broiler, the Smokless^{TM} broiler is not a direct-flame cooker and may result in a product that differs in appearance and taste. This contributed to the decision by South Coast AQMD staff to recommend against a rule that would require underfired charbroilers to be replaced with this technology or with some other alternative cooking device. The Smokless^{TM} broiler's capital cost is approximately $5,550, compared to about $3,100 for a similarly sized underfired charbroiler. Replacing an underfired charbroiler with a Smokless^{TM} broiler is reported to reduce PM_{10} emissions at a cost of $3,550 per ton; however, it does mean relying on a different cooking device (South Coast AQMD 2000).

A second emissions control option for underfired charbroilers is the CAST, Inc. add-on scrubber. For a very large restaurant (cooking 2,800 lb of meat per week), the South Coast AQMD estimates a cost effectiveness of $1,100–$7,300 per ton of PM_{10} reduced, depending on the type of meat cooked and the volume of make-up air saved by the device (South Coast AQMD 2004). According to the manufacturer, there are roughly 12 CAST scrubbers installed throughout the U.S., and the system has been improved over the past several years to increase energy savings associated with heating, ventilation, and air conditioning (Hopkins 2005).

An electrostatic precipitator (ESP) can also be used to control PM emissions from commercial meat charbroilers. The ESP used for controlling charbroiler particulate emissions (called the "Penney-type" ESP) involves a two-stage process. The exhaust stream enters the first stage from the hood, where it is charged; the charged particles are then collected in the second stage. The South Coast AQMD notes that improper cleaning can interfere with particulate removal efficiencies (Perryman 1997). There are a few dozen ESPs operating at restaurants located in the South Coast AQMD area (Whynot 1999). The South Coast AQMD estimates a cost-effectiveness of $10,000 per ton of particulate matter reduced (Perryman 1997).

United Air Specialists, Inc. manufactures an ESP system, known as the Smog-Hog® Air Pollution Control System, designed specifically for commercial kitchen emissions control. The system is used by hotels, high-rise buildings, airports, casinos, food courts, and sports and entertainment complexes throughout the U.S., including McDonald's, Pizza Hut, Burger King (San Francisco Airport), Planet Hollywood, Four Seasons Hotel Boston, and Mohegan Sun Casino (United Air Specialists undated). According to a contractor in the New York City area, the Smog-Hog® system is estimated to cost between $20,000–$30,000 (installed) for a typical sized restaurant (i.e., a restaurant with a 10-foot ventilation hood). Table 3-4 provides estimates of the cost per ton of $PM_{2.5}$ removed for a restaurant cooking 200 and 400 hamburgers per day. For a large banquet hall or corporate cafeteria, the cost could increase to upwards of $100,000 (Hayes 2006).

Table 3-4. Illustrative Example of Cost Effectiveness Calculations for an ESP System.

	Case 1	Case 2
Hamburgers cooked per day	200	400
$PM_{2.5}$ emission rate (lb/1,000 lb of meat)	15	15
Annual $PM_{2.5}$ emissions (lb)*	195	390
ESP cost	$20,000	$20,000
Control effectiveness (%)	90%	90%
$PM_{2.5}$ reduction (lb, over six years)	1,053	2,106
Cost per ton of $PM_{2.5}$ reduced ($)	$37,987	$18,993

* Assumes ¼ lb hamburgers and restaurant operates five days per week and 52 weeks per year.

Additionally, wet scrubbers are capable of PM removal efficiencies of 90% or higher, depending on particle size, load, flow rate, and design pressure drop. Wet scrubbers rely on a finely atomized stream of liquid to capture particulate and gaseous pollutants from an exhaust stream, like that of a restaurant charbroiler.[19] There are several wet scrubbers currently permitted in restaurants in the South Coast AQMD area (Perryman 1997). The South Coast AQMD estimates a cost of $13,000 per ton of PM reduced for a wet scrubber (Perryman 1997).

Apart from these add-on controls, the design of a charbroiler's metal grid can influence the amount of smoke generated by the device. Some grids are designed with a series of gutters that channel grease away from the flames of the broiler and toward the grease pan (Fisher 2002). Some models have a slanted grid that also serves to divert grease away from the flames. No estimates are currently available regarding the effectiveness of these designs in reducing $PM_{2.5}$ emissions. Also, griddles are available that do not have an exposed flame but that can create the characteristic stripes of an underfired charbroiler with far less smoke and more efficient energy use (Fisher 2002).[20]

Policy Options

South Coast AQMD Rule 1138 requires operators of both new and existing chain-driven charbroilers to install a catalytic oxidizer (Roe 2004). Alternative control devices can be used if they are at least as effective as a catalytic oxidizer in reducing PM and VOC emissions. Chain-driven charbroilers cooking less than 875 lb of meat per week are exempt (South Coast AQMD 1997a).

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[19] For example, see Moli-tron's wet scrubber for restaurants: http://www.moli-tron.com/scrubber.asp.

[20] For a fact sheet on grooved surface griddles, see http://mn.centerpointenergy.com/for_your_business/foodservice/fact_sheets/9552Griddles.pdf

Other areas have adopted or have considered adopting the South Coast AQMD rule. The Ventura County APCD and the San Joaquin Valley Unified Air Pollution District (UAPD), both in California, have adopted the South Coast AQMD rule (Ventura County APCD 2004b; San Joaquin Valley UAPD 2002). New Jersey is considering adopting the South Coast AQMD standards for chain-driven charbroilers (New Jersey Homes and Restaurants Workgroup 2005).

The South Coast AQMD conducted extensive research on emissions controls for underfired charbroilers and concluded that none of the options available as of 2004 met its cost effectiveness criteria. Accordingly, the agency chose not to regulate these charbroilers. South Coast AQMD staff have recommended that the governing board look to other industry sectors for more cost effective particulate reductions (South Coast AQMD 2004).

In addition to the efforts by California air districts, Pitkin County in Colorado has required PM_{10} reductions from restaurant operations. (Aspen County, Colorado, was formerly classified as a nonattainment area for PM_{10}.) Chapter 6.12 of the Pitkin County Code requires commercial cooking operations to maintain a control device on charbroilers installed on or after January 1, 1993. The control device must reduce uncontrolled PM_{10} emissions by at least 90% (Pitkin County Code 2004). The Pitkin County Code also prohibits the cooking of high fat-content meat (greater than 15% precooked fat content by weight) on charbroilers installed after April 25, 1983, but before January 1, 1993, unless an emission control device has been installed that reduces uncontrolled PM_{10} emissions by at least 90% (Pitkin County Code 2004). Finally, any commercial cooking facility within specified zones in Pitkin County that proposes to extensively remodel its kitchen is required to install a grooved griddle or an equivalent low-emissions cooking device if high fat-content meats are regular menu items. Pitkin County required such controls to be installed on existing restaurants by December 1, 1984 (Pitkin County Code 2004).

The State of Colorado has published a document also available online, “Guide to Environmental Regulations for Restaurants and Food Service Businesses.” This document instructs facilities to check for smoke from outside hood stacks and to use good cooking practices; it also suggests that restaurants with chain-driven charbroilers consider installing a catalytic converter and that other restaurants with underfired charbroilers consider using a scrubber or ESP (CO DPH&E 1999).

Recommendations

Based on the above analysis, the following viable strategies are recommended for reducing carbonaceous $PM_{2.5}$ and therefore exposure from this category:

- The State could consider regulating $PM_{2.5}$ emissions statewide from new and existing chain-driven charbroilers, in a manner similar to the South Coast AQMD's Rule 1138. This measure can provide over 80% reductions in the carbon fraction of primary $PM_{2.5}$ emissions in a cost-effective manner that could result in an approximate 250 ton reduction of $PM_{2.5}$ annually.
- The State could further investigate the feasibility of cost-effective technologies for controlling $PM_{2.5}$ emissions from the underfired charbroilers in the NYC Metropolitan NAA. Though not as cost-effective as controls for chain-driven charbroilers, these measures may be justifiable from public exposure perspective. Assuming a 100% compliance rate, more than 3,000 tons of carbonaceous $PM_{2.5}$ could be reduced each year.

RESIDENTIAL FUEL COMBUSTION

This section discusses the control technology and policy options available for reducing $PM_{2.5}$ emissions from residential sources in New York State, focusing primarily on the options available for households to reduce direct on-site $PM_{2.5}$ emissions. The major sources of PM emissions from the residential source category include (1) the combustion of wood in fireplaces, woodstoves, and outdoor boilers; (2) the combustion of fossil fuels (natural gas and oil) in boilers, furnaces, and other equipment; and (3) the outdoor burning of yard waste and other waste materials. As shown in Table 3-5, these sources account for an estimated 49% of direct $PM_{2.5}$ emissions in New York State. (Note that the $PM_{2.5}$ emissions from outdoor wood-fired boilers are not available in EPA's inventory and thus are not included in Table 3-5.) Wood combustion is the single largest source of $PM_{2.5}$ emissions from the residential source category, accounting for 39% of $PM_{2.5}$ emissions statewide (U.S. EPA 2006a). In rural counties, wood combustion can account for upwards of 70% of emissions (see Appendix A). On a mass basis, primary $PM_{2.5}$ emissions from residential wood combustion are composed of approximately 11% elemental carbon and 57% organic carbon (U.S. EPA 2005g). Thus, more than two-thirds of the primary $PM_{2.5}$ emissions from wood combustion consist of carbonaceous compounds.

Sector Profile

New York State has more than 7,754,500 housing units, including both single-family homes and multi-family apartment buildings (U.S. Census Bureau 2006). In 2001, the residential sector in New York State consumed 1.2 quadrillion British thermal units (Btus) or "quads" of energy—the third highest total of all the states, and the equivalent of more than 1% of total U.S. energy consumption in all economic sectors that year (EIA 2006). This high energy use is attributable to the state's large population and cold climate. New York is the third most populous state, with most of the population living in urban areas (EIA 2006).

Table 3-5. Major Sources of $PM_{2.5}$ Emissions within the Residential Source Category.

Source: EPA 2006a

Source Category	Source Subcategory	$PM_{2.5}$ Emissions (tons) U.S. EPA 2002 National Emissions Inventory	Percent of Source Category Emissions	Percent of Total New York State $PM_{2.5}$ Emissions
Residential wood combustion	Fireplaces Woodstoves Outdoor wood burning equipment	60,584	78%	39%
Residential Furnaces	Natural Gas	81	<1%	<1%
Residential Fuel Combustion	Distillate Oil	1,285	2%	<1%
Open Burning	Yard Waste	2,811	4%	2%
	Household Waste	12,526	16%	8%
Source Category Total		77,287	100%	49%

Every four years, the U.S. Department of Energy's Energy Information Administration (EIA) conducts a national survey of household energy use known as the Residential Energy Consumption Survey. According to the most recent survey, in 2001 more than half (58%) of total household energy consumption in New York State was devoted to home heating, while water heating was reported to account for an additional 18%. Lighting and other appliances (excluding refrigerators and air conditioners) also accounted for 18% of household energy consumption. Refrigerators and air conditioners are reported to have accounted for 1% and 4% of total energy consumption, respectively (EIA 2001).

Natural gas, used by 52% of households in New York State, is the most commonly used heating fuel. Oil, used by 37% of households, is the second most common fuel; electricity, used by only 8% of households, is the third most common (U.S. Census Bureau 2004). New York resembles several of the other states in the Northeast in this regard. Nationwide, less than 9% of households rely on fuel oil for home heating, while 31% of households rely on electricity for home heating (U.S. Census Bureau 2004a).

In terms of electricity use, in 2001 a New York household consumed, on average, roughly half the amount of electricity consumed by an average U.S. household: 5,974 kilowatt-hours (kWh), compared with 10,656 kWh. As previously indicated, relatively few households in New York rely on electricity for space heating. Most New York households also use natural gas or fuel oil for water heating. Only 13% of New York households used electric water heaters in 2001. In New York, kitchen appliances accounted for the largest share of household electricity consumption in 2001 at 32% of the total (EIA 2006).

Wood is generally not the primary source of energy for home heating in U.S. households. Widespread use of woodstoves and fireplaces in some areas, however, is an important contributor to ambient levels of particulate matter. In certain winter pollution episodes, more than half of the atmospheric $PM_{2.5}$ concentration in a local area can be attributed to wood smoke (Schauer and Cass 2000). According to the EPA, there are 40–45 million wood burning appliances in the U.S., 15 million of which are woodstoves, either free standing or fireplace inserts (Brockman 2005). Many of the woodstoves that are in regular use in the U.S. today were installed prior to adoption of the federal New Source Performance Standards (NSPS) in 1988. In the Northeast, households consume more than twice the amount of wood per year than households in the Midwest, South, or West (Houck et al. 2001).

Information on the use of wood burning equipment in New York State, including the types of appliances used and activity rates, is limited. For several years, the Mid-Atlantic Regional Air Management Association (MARAMA) has been working to improve the residential wood combustion $PM_{2.5}$ emissions inventory for New York and other states in the region, and work is ongoing to improve the existing inventory (MARAMA 2006). Preparing an accurate inventory of $PM_{2.5}$ emissions from residential wood combustion is a challenging task because of variability in the use of wood burning equipment for home heating, the different types of equipment used, the different types of wood and manufactured fuels used, and the variability in the properties of the fuel, such as moisture content (MARAMA 2006). Information on MARAMA's Residential Wood Combustion Survey and Emissions Inventory Projects is available from its Web site at http://www.marama.org.

Sector Emissions

As noted in Table 3-5, according to the EPA's 2002 National Emissions Inventory, residential wood combustion produced 60,584 tons of $PM_{2.5}$ emissions in New York State. On a mass basis, this $PM_{2.5}$ from residential wood combustion is composed of approximately 11% elemental carbon and 57% organic carbon (U.S. EPA 2005g). It is the single largest source of $PM_{2.5}$ emissions from the residential source category, accounting for 39% of all $PM_{2.5}$ emissions statewide. In rural counties, residential wood combustion accounts for a much higher share of total $PM_{2.5}$ emissions—as high as 78% in Lewis County and 71% in Cattaraugus County (see Appendix A).

In the metropolitan NAA, residential wood combustion is estimated to generate 6,610 tons of total $PM_{2.5}$, or more than 13% of total annual $PM_{2.5}$ emissions within the area. Approximately 4,500 tons of this $PM_{2.5}$ is carbonaceous. The bulk of these emissions (75%) occurs in the New Jersey and Connecticut counties within the nonattainment area (see Appendix A).

Open burning of household waste (e.g., paper and plastic) is the second largest source category within the residential sector, accounting for 8% of statewide $PM_{2.5}$ emissions. Open burning produces $PM_{2.5}$ emissions, as well as a host of other pollutants, depending on the materials burned. Burning household trash presents a special concern because of the air toxics generated (EPA 2003). Open burning of yard waste (e.g., leaves and brush) and residential oil use account for less than 2% and less than 1% of statewide $PM_{2.5}$ emissions, respectively.

MARAMA has also developed an inventory of $PM_{2.5}$ emissions from the residential source category for New York and other states in the region. The MARAMA and U.S. EPA inventory data are summarized in Table 3-6. The large differences in emissions estimates from two sources indicate the need for continuing efforts of the states and the EPA in improving emissions inventory for some of the hard-to-quantify source categories.

Federal and State Authority

Woodstoves. In 1988, the EPA established New Source Performance Standards (NSPS) for woodstoves, requiring all new woodstoves sold in the U.S. to be certified to meet certain particulate emissions limits by 1992 (EPA 1988).[21] EPA's mandatory smoke emissions limit for woodstoves is 7.5 grams of smoke per hour (g/hr) for noncatalytic stoves and 4.1 g/hr for catalytic stoves.[22] Some newer stoves have EPA-certified emissions in the range of 1–4 g/hr (compared to an average emissions rate of approximately 42 g/hr for earlier models).

The Energy Policy Act of 2005 provides a rebate to consumers who purchase a renewable energy appliance with a thermal efficiency rating of 75%, including wood and wood-pellet stoves. The bill provides a 25% rebate, capped at $3,000.

Although there are no New York State laws or regulations that specifically address woodstoves, the Code of the City of Buffalo has a provision addressing private residences that appears to be applicable:

> *Fuel-burning equipment or devices installed to heat buildings used exclusively for private residences containing fewer than three dwelling units or flats shall be equipped with efficient smoke-eliminating apparatus, unless such equipment or device is fueled with anthracite coal, coke, oil, gas, or other smokeless fuel (CCB sec. 66-15).*

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[21] EPA's Web site provides a listing of EPA-certified wood stoves: www.epa.gov/compliance/resources/publications/monitoring/.

[22] Wood stoves offered for sale in the State of Washington must meet a particulate emissions limit of 4.5 g/hr for noncatalytic wood stoves and 2.1 g/hr for catalytic wood stoves.

Additionally, the general state and local nuisance-type air pollution requirements and opacity limits discussed in the section on commercial cooking are applicable to woodstoves, as well.

Table 3-6. $PM_{2.5}$ Emissions Inventory Data For NY State: U.S. EPA NEI and MARAMA (Version 3).

Source: U.S. EPA 2006a; MARAMA 2006a

Source Category	Source Subcategory	$PM_{2.5}$ Emissions (tons) MARAMA 2002 MANE-VU Version 3	$PM_{2.5}$ Emissions (tons) U.S. EPA 2002 National Emissions Inventory
Residential wood combustion	Fireplaces	25,471	60,584
	Woodstoves	8,997	
	Outdoor wood burning equipment	2,234	
Residential Furnaces	Natural Gas	1,429	81
Residential Fuel Combustion	Distillate Oil	1,222	1,285
Open Burning	Yard Waste	430	2,811
	Household Waste	1,023	12,526

Note: In preparing the 2002 National Emissions Inventory, EPA concluded that the emissions factors used in prior inventories to estimate $PM_{2.5}$ emissions from natural gas-fired equipment were too high (U.S. EPA 2005d). EPA's estimates have since been reduced by 95%. This may account for the large discrepancy between the two inventories.

Outdoor Wood-Fired Boilers. Neither the NSPS applicable to woodstoves nor any other federal air pollution regulations cover this new and emerging source category. As with woodstoves, there are no New York State or local provisions that address outdoor wood-fired boilers, with the exception of Sec. 66-15 of the Code of the City of Buffalo, cited in full above (requiring fuel-burning devices used to provide heat in private residences to be equipped with "efficient smoke-eliminating apparatus"). The general nuisance and opacity requirements also apply to these sources.

Open Burning of Garbage and Yard Waste. Many jurisdictions have requirements that specifically address open burning of garbage and yard waste. Of the jurisdictions whose requirements were reviewed, New York State, Buffalo, Monroe County, and New York City have such requirements. New York State regulations prohibit open burning of rubbish (including paper, trees, lawn and garden debris, plastics, wood) in certain locations; and of garbage (includes food waste), subject to certain exceptions. The exceptions apply so long as they do not conflict with other laws, and they generally require a permit or require the state to designate an area as appropriate for open burning (NYS DEC secs. 215.1, 215.2, 215.3). The Code of the City of Buffalo prohibits "open, unconfined fire[s]" without a permit (CCB sec. 66-19).

Monroe County outlaws the burning of refuse in open fires; exceptions include "outdoor grills and fireplaces; campfires; and burning of tree trimmings, animal and vegetable waste, and brush, so long as this does not create a nuisance" (CMC sec. 569-40). New York City likewise forbids open fires, with exceptions for certain barbecue equipment (NYC AC sec. 24-149). The nuisance-type air pollution and opacity requirements are also applicable to open burning.

Fuel Sulfur Limits. With certain limited exceptions, New York State regulations prohibit the sale or use of fuel with sulfur content that exceeds the limits specified in Table 3-7 (NYS DEC sec. 225-1.2).

Table 3-7. NY DEC Sulfur-in-Fuel Limitations.

Area	Percent Sulfur by Weight	
	Residual Oil	Distillate Oil
New York City	0.30 (approx 3,000 ppm)	0.20 (approx 2,000 ppm)
Nassau, Rockland and Westchester Counties	0.37 (approx 3,700 ppm)	0.37 (approx 3,700 ppm)
Suffolk County: Towns of Babylon, Brookhaven, Huntington, Islip and Smith Town	1.00 (approx 10,000 ppm)	1.00 (approx 10,000 ppm)
Erie County: City of Lackawanna and South Buffalo	1.10 (approx 11,000 ppm)	1.10 (approx 11,000 ppm)
Niagara County and remainder and Erie County	1.50 (approx 15,000 ppm)	1.50 (approx 15,000 ppm)
Remainder of State	1.50 (approx 15,000 ppm)	1.50 (approx 15,000 ppm)

New York City also has limitations on the sulfur content of fuel (NYC AC sec. 24-169), which closely mirror the state requirements. They are as follows:

- Fuel oil grade No. 2: 0.2% sulfur by weight
- Residual fuel oil and fuel oil grade No. 4: 0.3% by weight

Technical Options for Emissions Reduction

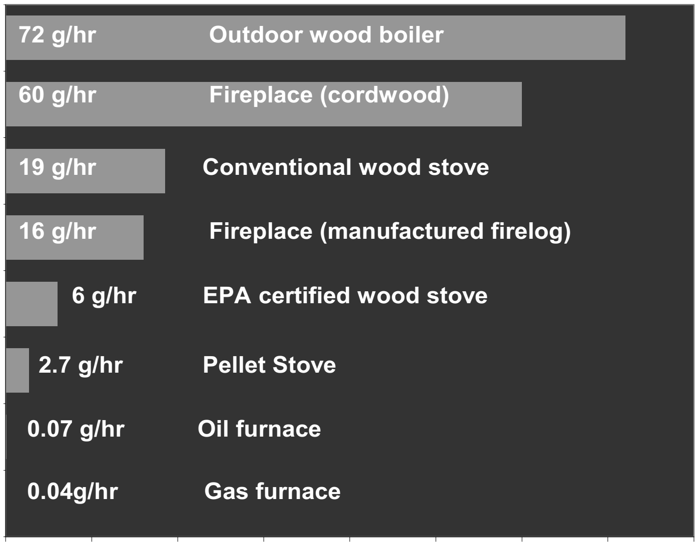
Wood Stoves and Fireplaces. There are three basic strategies for reducing $PM_{2.5}$ emissions from woodstoves and fireplaces. These strategies address the OC and EC fractions that add up to about 70% of the primary $PM_{2.5}$ emissions from wood combustion: (1) reducing or eliminating the burning of wood where less polluting alternatives (e.g., natural gas) are available; (2) relying on higher-efficiency, lower-polluting woodstove technology; and (3) adopting good burning techniques (e.g., using properly sized and properly seasoned wood).

As a result of the EPA's 1988 emissions limits, all new woodstoves and fireplace inserts must be certified to a maximum limit of 7.5 grams of smoke per hour. Fireplaces and outdoor wood boilers are exempt from this regulation. The EPA's NSPS limits have not only improved the emissions performance of new woodstoves but have also improved their fuel efficiency. Woodstove manufacturers use two basic technologies to meet the EPA standards: catalytic and noncatalytic combustion (Wood Heat Organization 2005). With catalytic combustion, the exhaust from the stove passes through a specially coated ceramic honeycomb inside the stove, where the smoke gases and particles ignite and burn. The ongoing stove performance depends on the maintenance and replacement of the catalytic honeycomb, which can degrade in as little as two years without regular cleaning and maintenance (Wood Heat Organization 2005).

Noncatalytic stoves rely on the design characteristics of the unit to accomplish complete combustion of the wood. These design features include firebox insulation, a large baffle to produce a longer, hotter gas flow path, and preheated combustion air introduced through small holes above the fuel in the firebox. Noncatalytic stoves have no catalyst to replace; however, some of the internal components of a noncatalytic stove need replacement as they degrade because of high temperatures. Most of the stoves currently available are of the noncatalytic type. Wood pellet stoves, which burn sawdust and other wood products pressed into small pellets, have grown in popularity and produce lower emissions than a conventional stove. Some pellet stoves can also burn dried corn kernels.

Figure 3-5. PM Emissions Rates (g/hr).

Source: Schreiber et al. 2005; Houck et al. 2000, Houck and Tiegs 1998



Some of the alternative fuel options for woodstoves include manufactured firelogs and natural gas fireplaces, as well as alternative heating devices such as natural gas furnaces. (See Figure 3-5 for a comparison of average PM g/hr emissions rates for various home-heating devices.) Gas fireplaces are becoming increasingly popular, with newer models that mimic the look of a real wood fire. These self-contained units can be installed into an existing masonry fireplace. Manufactured firelogs, made from wax and sawdust for open-hearth fireplaces, can reduce

$PM_{2.5}$ emissions. Studies suggest that manufactured firelogs can reduce $PM_{2.5}$ emissions by an average of 69%, relative to burning cordwood in a fireplace (Houck 2000).

As indicated in Table 3-8, woodstoves and fireplaces are the leading sources of $PM_{2.5}$ emissions within the residential buildings sector. Table 3-8 presents $PM_{2.5}$ emissions factors (in lb of $PM_{2.5}$ per ton of wood burned) for a conventional wood stove, catalytic EPA-certified stove, noncatalytic EPA-certified stove, and pellet stove. Data come from two sources and show reasonable agreement: (1) EPA's AP-42 emissions factor handbook and (2) a recent study prepared for the Hearth, Patio, and Barbecue Association (Houck et al. 2005a).[23]

Table 3-8. Wood Stove $PM_{2.5}$ Emissions Factors.

Source: U.S. EPA 1996; Houck et al. 2005

Stove Type	AP-42 $PM_{2.5}$ Emissions Factors (lb/ton)	Houck $PM_{2.5}$ Emissions Factors (lb/ton)
Conventional wood stove	30.6	66.8
Catalytic EPA-certified wood stove	16.2	15.1
Noncatalytic EPA-certified wood stove	14.6	11.7
Pellet stove	8.8	2.5

The benefits of replacing a conventional wood stove with an EPA-certified unit can be estimated based on the emissions factors in Table 3-8, but they are higher when one takes into account the higher efficiency of the newer technology. A modern stove provides higher heating value for the same quantity of wood burned; an EPA-certified catalytic or noncatalytic stove is about 10% more efficient than a conventional wood stove, and a pellet stove is 20% more efficient (Houck 2005). Higher-efficiency, cleaner-burning stoves also reduce creosote buildup and may reduce the number of chimney cleanings required.

The average cost of a new stove, including the cost associated with a chimney upgrade, is about $2,500 (Houck and Broderick 2005). Assuming that a conventional wood stove with a $PM_{2.5}$ emissions factor of

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[23] In the Hearth, Patio, and Barbecue Association study, the authors adjust the average emissions factors for conventional stoves because they assume lower burn rates than EPA assumed in its AP-42 Emission Factor Handbook. This results in higher emissions. Also, the study suggests lower average emissions factors for EPA-certified stoves, based on research demonstrating that the emissions performance of wood stoves has improved since the earliest models were produced in the early 1990s. Earlier models were used as the basis for the AP-42 emissions factors for EPA-certified stoves. EPA is currently in the midst of re-evaluating the AP-42 emissions factors for wood stoves, which may shed some light on the differences in estimated emissions rates between these two sources (Wood 2005).

66.8 pounds per ton (lb/ton) is replaced by a noncatalytic wood stove with a $PM_{2.5}$ emissions factor of 11.7 lb/ton, $PM_{2.5}$ emissions per unit would be reduced by roughly 140 lb per heating season.[24]

According to the EPA, preliminary estimates suggest that changing out existing wood stoves can be a cost-effective option for reducing $PM_{2.5}$ emissions. The EPA estimates that this approach can cost less than \$2,000 per ton of $PM_{2.5}$ reduced (Houck et al. 2005a). The economic feasibility of purchasing and installing a new wood stove is a separate issue, which depends on the income of the household. As discussed later in this chapter, money grants from local health departments and from the EPA have been used to fund the replacement of older stoves with new low-emission stoves for low-income families.

Outdoor Wood Boilers (OWBs). Over the past several years, outdoor residential wood boilers have become a focus of heightened concern and state regulatory attention because of their high air pollution emissions rates and growing popularity (e.g., Colburn 2004; CTDEP 2004; Spitzer 2005; WIDPH 2005). A typical OWB can emit six times more $PM_{2.5}$ emissions than an EPA-certified wood stove (see Figure 3-5). Vermont has proposed a rule to regulate new wood-fired boilers, and the New York Attorney General has petitioned the EPA to promulgate NSPS standards for the boilers, or to revise the existing standards for residential wood stoves to include outdoor wood boilers (Spitzer 2005).

An outdoor wood boiler is a wood-fired furnace housed within a small shed located near the home. Inside the boiler is an oversized firebox that can accommodate large loads of wood. (Some states report the burning of yard waste, construction debris, and domestic wastes as well.) A reservoir of water surrounds the firebox. The unit cycles water through the jacket to provide hot water to the building. Water pipes run underground to deliver hot water for space heating and other uses year-round. The unit can be filled with wood and left to burn for long periods, even days at a time.

The total cost to purchase and install the smallest outdoor wood boilers can range from \$8,000–\$10,000, with costs increasing with the size of the unit. The boiler itself costs approximately \$5,000, excluding installation. Installation typically involves laying a concrete foundation, installing a power source, and installing underground piping from the unit to the house and other additional piping. NESCAUM estimates that over 155,000 OWBs have been sold nationwide since 1990, with 13,182 sold in New York (NESCAUM, 2006). Some of the other states with large number of these OWBs include Michigan (29,568), Wisconsin (27,452), and Minnesota (13,936).

Outdoor wood-fired boilers can generate large amounts of smoke because of the damper system that regulates the output of the unit. When heat is no longer required, an air damper cuts off the air supply to the

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[24] This example calculation also assumes an annual average cord usage of 1.75 cords per stove (adjusting the amount of wood used by the new stove to reflect its increased efficiency), and a cord-to-mass conversion factor of 1.4 tons per cord, which is reported to be typical of wood used in the eastern U.S. (Houck and Broderick 2005).

furnace, producing a smoldering fire. The smoke and creosote that accumulate on the internal surfaces of the unit are released to the air in a belch of smoke when the air damper opens and the natural draft forces air into the firebox. Without the catalytic and noncatalytic control systems used by conventional wood stoves, the equipment emits about 12 times as much PM as EPA-certified stoves, as indicated in Figure 3-5 (NESCAUM 2006).

Fuel Oil Combustion. Fuel oil is the second most common fuel used for space heating in New York. As indicated in Table 3-5, it accounts for about 2% of $PM_{2.5}$ emissions in the residential source category. (Natural gas accounts for a larger share of residential fuel combustion use, but it generates minimal direct $PM_{2.5}$ emissions.) New York consumes more residential fuel oil than any other state (CCAP 2005). Heating oil, also known as No. 2 fuel oil or distillate oil, is much like diesel fuel, with at least one important distinction: heating oil contains between 2,000 and 2,500 parts per million (ppm) of sulfur (Bookhart 2003). By comparison, sulfur levels in onroad diesel fuel are currently limited by regulation to less than 500 ppm and will be reduced to 15 ppm in late 2006 (EPA 2001a). In addition, residual fuel oil is used as a back-up fuel for heating large apartment buildings in New York City when natural gas supplies are interrupted. According to a petroleum infrastructure study produced for NYSERDA in 2006, at least 471 apartment buildings in New York City are in this category. Sulfur content of this residual fuel ranges from 5000 ppm to over 40,000 ppm. Switching from high-sulfur fuel oil to low- (500 ppm) and ultra-low- (15 ppm) sulfur oil can reduce $PM_{2.5}$ as well as SO_2 emissions. The PM emissions from oil-fired heating systems include unburned carbon particles and any ash residue in the fuel (i.e., soot). Sulfates are also formed when SO_2 in the ambient air is oxidized to secondary sulfate particles. Switching to low-sulfur-content fuel is reported to eliminate 80% of the $PM_{2.5}$ emissions generated by residential oil heating systems, and 75–80% of the SO_2 emissions (McDonald 2003). On a mass basis, $PM_{2.5}$ from residential oil combustion is composed of approximately 19% elemental carbon and 24% organic carbon (EPA 2005g).

In testing, low-sulfur heating oil has been shown to reduce maintenance and service requirements. NYSERDA and Brookhaven National Laboratory conducted a three-year demonstration project using low-sulfur fuel for residential heating. One thousand homes participated in the study, which used diesel fuel with a sulfur content of 500 ppm. Over the course of the project, use of low-sulfur oil caused maintenance and service requirements to decline substantially, while at the same time dramatically reducing SO_2 emissions (EPSA 2005).

Policy Options

Several policy options are available for reducing $PM_{2.5}$ emissions from the residential source category in New York. Some of these options require legislative or regulatory action; others can be implemented based on existing authority.

Wood Stove Change Out Programs. A growing number of communities have been exploring wood stove change-out programs as an option for addressing particulate air pollution. In 2001, counties in western New York bordering the Great Lakes agreed to participate in *The Great Wood Stove & Fireplace Changeout* with areas in other Great Lakes states, but no wood stoves or fireplaces were sold in New York through the program (North Central Hearth Products Association 2001). In 2005, the EPA provided funding and technical support for the development of two model programs. The EPA and its project partners initiated one such program in June 2005 in Libby, Montana, a small town in western Montana. The hearth industry has provided over $1 million in wood stoves and other products to assist in this campaign (Wood 2005). The Libby experience is expected to be a test case for eastern states considering similar programs. On September 29, 2005, the Southwest Pennsylvania Air Quality Partnership announced the start of its wood stove change-out program with a $100,000 grant from the EPA and others, and an $80,000 grant from the Allegheny County Health Department. The program offers free change-outs for low-income families and replacements at a discount for other residents (Allegheny County 2005).

EPA's Web site, www.epa.gov/woodstoves/changeout.html, provides additional information on the EPA's wood stove change-out program, including technical resources for launching a local initiative. Other communities, such as Delta County, Colorado, have offered rebates or low-interest loans for stove change-outs. In the Delta County program, the amount of the rebate varied depending on the efficiency and type of the new model. For instance, higher rebates were awarded to households that installed pellet or gas stoves. In February 2006, the EPA issued guidance for quantifying the emissions reductions resulting from wood stove change-out campaigns for states seeking state implementation plan (SIP) credits for such programs (STAPPA/ALAPCO 2006).

Wood Stove Mandatory Removal Programs. Several communities in California and Nevada, including the San Joaquin Valley, Mammoth Lakes, and Reno, require all non-EPA-certified wood stoves to be removed or retired prior to the sale or transfer of a property. This strategy can be a powerful method for accelerating the removal of older stoves and is generally feasible because the cost of removing a stove usually constitutes an insignificant share of the property sale price. This strategy could also be used to address outdoor wood-fired boilers.

Outdoor Burning Restrictions. The primary strategy for addressing emissions from open burning is to ban the practice, while providing waste-disposal alternatives. Public education and consistent enforcement are also keys to program success. Increasing household waste pickups, establishing and promoting free or low-cost drop-off centers, and educating residents about disposal alternatives all serve to discourage open burning (EPA 2001, 2003).

Connecticut, Maine, Massachusetts, New Jersey, New Hampshire, and Vermont have statewide bans on backyard burning of household waste (EPA 2003). In New York, some counties ban the practice; however, the practice is not banned statewide. As mentioned above in the federal and state authority discussion, Monroe County outlaws the burning of refuse in open fires (CMC sec. 569-40). Legislation is currently pending in the New York State Assembly that would ban the open burning of household solid waste throughout the state (Bill No. A3073). The bill exempts the burning of yard waste, trees, and brush. The bill would institute penalties consisting of a warning for a first offense, a $75 fine for a second violation, and a fine of $100-250 for third and subsequent violations.

In response to high rates of residential garbage burning in northeastern Minnesota and northwestern Wisconsin, the Western Lake Superior Sanitary District has developed a toolkit of information for local officials to assist them in addressing the problem. The toolkit includes a model ordinance to regulate outdoor burning, public education materials to alert people to the hazards of burning household waste, case studies of successful programs, as well as other resources. The toolkit is available at www.wlssd.com/Open_Burning/Clearing_the_Air_downloadvs.pdf.

Low Sulfur Heating Oil. The sulfur levels in residential fuel oil can be reduced by amending existing law or through other legislative action. According to the Empire State Petroleum Association, heating oil companies are willing to market low-sulfur fuel oil to their customers (EPSA 2005), and legislation is currently pending in both the State Assembly (Bill No. A6453) as well as the State Senate (Bill No. S01390) that would direct the Department of Environmental Conservation (DEC) to enact regulations limiting allowable sulfur levels in home heating fuel. In the near term, the high price of oil is likely to discourage action on these bills. Because of the comparatively low price for residual fuel oil, its use as a back-up fuel in large apartment buildings in New York City is likely to continue, absent any regualory initiatives to promote alternatives.

Recommendations

The most important source of EC and OC emissions in New York State is residential wood combustion (wood stoves, fireplaces, and fireplace inserts) and the emerging source category of outdoor wood-fired boilers. The following viable strategies are recommended to reduce carbonaceous $PM_{2.5}$ levels and therefore exposure from this category:

- The State could evaluate the feasibility of wood stove change-out initiatives (replacing older high-polluting stoves with EPA-certified models) that are being implemented in other parts of the U.S.
- The State could promote the federal tax incentives (included in the Energy Policy Act of 2005) for high-efficiency wood stoves as well as supplement the program with funding of its own.
- The State could require cleaner burning wood for fireplaces.

- The State might seriously consider regulating emissions from outdoor wood-fired boilers by developing performance-based emissions standards.
- In the NAA approximately 4,500 tons of carbonaceous $PM_{2.5}$ result from wood burning. If the above measures were implemented between 2,000 and 3,000 tons of carbonaceous $PM_{2.5}$ emissions could be avoided annually.

LIGHT-DUTY MOTOR VEHICLES

Total direct emissions of fine particulates from light duty vehicles are the result of a combination of emissions from vehicle exhaust, brake and tire wear, road wear, and re-entrained road dust. In 2002, light-duty vehicle exhaust plus brake and tire wear-related emissions totaled 2,131 tons of $PM_{2.5}$ in New York State (approximately 1.4%of total $PM_{2.5}$ emissions in the State). Road wear and re-entrained road dust contributed another 3,968 tons of $PM_{2.5}$, accounting for another 2.6% of total statewide $PM_{2.5}$ emissions. Total primary $PM_{2.5}$ from all five subcategories of emissions attributed to light-duty vehicles is composed of 26% organic carbon and 7% elemental carbon. Thus, approximately 33% of light-duty vehicle primary $PM_{2.5}$ emissions is carbonaceous. According to the 2002 National Emissions Inventory (NEI) Summary for New York State, there were 12,538 tons of $PM_{2.5}$ emissions from paved roads, and heavy-duty trucks were responsible for the majority of this total. Only about 7.7% of this total (970 tons per year) was due to road surface wear. As the term implies, road surface wear is the result of vehicles traveling over roadways and eroding the paved surfaces. In contrast, most of the road dust emissions are a function of re-entrainment of silt and vegetative materials deposited onto the roadway by wind, vehicles, application of salt and/or sand for traction, and other processes. Road dust is largely inorganic "crustal" material, derived from the erosion of soil surfaces, but also includes elemental and organic components, the content of which varies by geographical region. According to the limited data available, on average, paved road dust is around 1% elemental carbon and 13% organic carbon.

In addition to primary $PM_{2.5}$, light-duty vehicles emit hydrocarbons from the tailpipe and as evaporative losses which form secondary organic aerosols (SOA). While the focus of this report is on primary $PM_{2.5}$ emissions, SOA emissions from light-duty vehicles and potential controls are discussed briefly in this section.

Total light-duty vehicle primary $PM_{2.5}$ emissions in New York State are small relative to other sectors, such as residential fuel combustion. However, the source is potentially very important because light-duty vehicles contribute significantly to public exposure and elevated levels of carbonaceous $PM_{2.5}$ in urban areas, in vehicle cabins, and at roadsides. Light-duty vehicle emissions are concentrated in highly populated urban areas. An analysis of inventory data for the five New York City counties (New York, Queens, Richmond, Brooklyn, and Bronx Counties) demonstrated that light-duty vehicle emissions comprise 12 percent of total carbonaceous $PM_{2.5}$ emissions. Thus, they impact large numbers of people—those who commute or who walk, bike, or live near heavily trafficked roads.

Sector Profile

In 2002, there were more than 9.1 million light-duty vehicles registered in New York State.[25] These vehicles traveled more than 129 billion miles (NEI 2002) and consumed 15.7 million gallons of gasoline each day (EIA 2004). The vast majority are powered by gasoline engines; diesel cars make up less than 0.5 percent of the vehicles in New York State's light-duty fleet. Recent changes in California's on-board diagnostic (OBD) requirements will allow greater numbers of diesel vehicles to be certified for sale in New York State.[26] This, combined with consumer demand for diesel light-duty vehicles, due to their power and fuel economy benefits, means that the number of diesels in New York State will likely be greater in the future.

The geographical focus of this section is on 15 of the 62 counties in New York State. The nine counties[27] that comprise the New York Metropolitan Area (NYMA) contain 63% of the state's population (U.S. Census 2000). Their citizens register 48% of the light-duty vehicles and are responsible for 47% of the statewide light-duty vehicle miles traveled (VMT). Outside of the NYMA, there are four additional metropolitan areas with significant vehicle populations. They are Albany (Albany County), Buffalo (Erie County), Rochester (Monroe County), and Syracuse (Onondaga County). Two additional counties (Orange and Putnam) are adjacent to though not officially part of the NYMA. They register high VMT counts, relative to their populations, and participate in the NYMA's reformulated gasoline program. These six non-NYMA counties account for 15% of the state's population and 23% of the statewide light-duty VMT.

Table 3-9. County Population and Annual Vehicle Miles Traveled (VMT).

County	Population (2000 Census)	Percent of Total NYS Population	County In NYMA?	Gasoline LDV VMT in CY 2002 (10^6 mi/yr)	Percent Total NYS Gasoline LDV VMT	Per Capita Gasoline LDV VMT (10^3 mi/ person/yr)
Albany	294,565	1.6	No	3,280.1	2.5	11.1
Bronx	1,332,650	7.0	Yes	4,327.8	3.3	3.2
Erie	950,265	5.0	No	8,132.8	6.3	8.6
Kings	2,465,326	13.0	Yes	4,487.6	3.5	1.8

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[25] Light-duty vehicles include three classes. These are conventional passenger vehicles, capable of holding 12 passengers or less, light-duty trucks with a gross vehicle weight rating (GVWR) of up to 3700 pounds, and the heavier class of light-duty trucks with a GVWR of up to 8500 pounds.

[26] Although diesel light-duty vehicle numbers are currently small, it will be important for policy makers to consider the impacts of increasing numbers of these because diesel light-duty vehicles emit close to the PM standard whereas gasoline light-duty vehicles typically emit well below the PM standard.

[27] The nine NYMA counties are Bronx, Kings, Nassau, New York, Queens, Richmond, Rockland, Suffolk, and Westchester.

Monroe	735,343	3.9	No	6,752.8	5.2	9.2
Nassau	1,334,544	7.0	Yes	10,610.0	8.2	8.0
New York	1,537,195	8.1	Yes	3,987.3	3.1	2.6
Onondaga	458,336	2.4	No	4,307.4	3.3	9.4
Orange	341,367	1.8	No	4,038.3	3.1	11.8
Putnam	95,745	0.5	No	2,749.5	2.1	28.7
Queens	2,229,379	11.7	Yes	7,160.7	5.5	3.2
Richmond	443,728	2.3	Yes	1,827.7	1.4	4.1
Rockland	286,753	1.5	Yes	2405.1	1.9	8.4
Suffolk	1,419,369	7.5	Yes	17,886.0	13.8	12.6
Westchester	923,459	4.9	Yes	8,012.9	6.2	8.7
Totals		78.2		89,966.0	69.4	

Table 3-9 summarizes the population and VMT for each of the 15 counties. In general, the more populous NYMA counties have lower per-capita VMT relative to their non-NYMA counterparts. Two exceptions are Suffolk and Nassau Counties.

Sector Emissions

Exhaust Emissions. Gasoline is a volatile hydrocarbon fuel that in a closed and perfect combustion system would be fully oxidized to carbon dioxide and water. In reality, gasoline engines achieve only partial oxidation of the fuel and emit from the tailpipe various products of incomplete combustion (e.g., EC, OC, unburned fuel). Newer engine technologies achieve a more complete oxidation of hydrocarbon fuels compared to their older counterparts, resulting in lower emissions of PM and hydrocarbons. In addition, the EC and OC fractions of total carbon emissions vary by model year grouping. These facts were borne out in a 1996-97 study conducted by the University of California, Riverside, College of Engineering-Center for Environmental Research and Technology (CE-CERT) in which emissions were measured on 129 vehicles, spanning model years from 1965 to 1997. The pertinent results are summarized in Table 3-10.

Table 3-10. Exhaust Emission Rates by Model Year Grouping (CE-CERT, 1998).

Vehicle Model Years	Av. PM Emission Rate (g/mile)	OC Emissions (% Total Carbon)	EC Emissions (%Total Carbon)	Av. THC Emission Rate (g/mile)
Pre – 1981	0.0338	79.3	20.7	5.06
1981 – 1985	0.049	72.6	27.4	2.07
1986 – 1990	0.0144	69.7	30.3	0.96
1991 – 1997	0.0025	37.1	62.9	0.34

Not surprisingly, emissions from individual engines have decreased as technology has improved for new engines. Although the relative contribution of EC emissions appears to have,increased, the overall PM emission rate (g/mi) has decreased. Overall emissions also have improved through attrition of the older fleet of vehicles. In addition to attrition, an older vehicle typically will travel fewer miles in a year as compared to its newer counterpart. Table 3-11 illustrates this decline in annual VMT by model year grouping.

Table 3-11. Gasoline Annual LDV VMT* By Model Year Grouping.

County	Pre-1981 (10^6 VMT/Yr)	1981 – 1985 (10^6 VMT/Yr)	1986 – 1990 (10^6 VMT/Yr)	1991 – 1997 (10^6 VMT/Yr)	1998 – 2004 (10^6 VMT/Yr)	Total (10^6 VMT/Yr)
Total VMT	299.2	449.7	3,751.6	22,871.9	62,593.6	89,966.0

* VMT is for Calendar Year 2002 and assumed to closely approximate VMT even for later-year vehicles.

While the decline in VMT as vehicles age partially offsets the relatively high per-vehicle emissions contribution, older vehicles (pre-1997) still contribute half of the EC, OC, and PM emissions from all light-duty vehicles, as shown in Table 3-11 and Table 3-12. This is the case even though newer vehicles account for more than two-thirds of total vehicle miles traveled. Because of this, programs aimed at lowering emissions from older vehicles could provide significant PM reductions from the light-duty sector.

Table 3-12. Primary Carbonaceous Exhaust Emissions (tons/year) by Model Year Grouping.

	Pre-1981 (Tons/Yr)	1981 – 1985 (Tons/Yr)	1986 – 1990 (Tons/Yr)	1991 – 1997 (Tons/Yr)	1998 – 2004 (Tons/Yr)	Total Tons (15 counties)
Elemental Carbon	1.68	4.74	12.16	23.55	64.49	106.62
Organic Carbon	6.50	12.53	27.93	13.86	37.93	98.75
Total	8.18	17.27	40.09	37.41	102.42	205.37

Tire, Brake, and Road Surface Wear. The wearing of tires, brakes, and road surfaces results in $PM_{2.5}$ emissions. The emission factors used are a function of VMT (EEA, 2003). Table 3-13 and Table 3-14 show the EC and OC emissions estimates for these categories. More than 80% of carbonaceous $PM_{2.5}$ related to tire, brake, and road surface wear is organic carbon as can be seen from Table 3-13.

Table 3-13. Primary Carbonaceous Emissions from Tire, Brake, and Road Surface Wear.

	Tire Wear (Tons/Yr)	Brake Wear (Tons/Yr)	Road Surface Wear (Tons/Yr)	Total Tons (15 counties)
Elemental Carbon	30.42	13.77	6.84	51.01
Organic Carbon	71.65	55.19	87.47	214.31
Total Tons	102.07	68.96	94.31	265.32

As mentioned previously, $PM_{2.5}$ emissions from road surface wear represent a small fraction of $PM_{2.5}$ from road dust. Road dust-related $PM_{2.5}$ is approximately eight times greater than $PM_{2.5}$ emissions from road wear. Table 3-14 compares EC and OC fractions of $PM_{2.5}$ emissions from road surface wear and re-entrained road dust. The table shows that over 90% of carbonaceous $PM_{2.5}$ from these two sources is OC.

Table 3-14. Comparison of Total Carbonaceous $PM_{2.5}$ Emissions from Road Wear and Road Dust for the 15 New York State Counties.

	Road Wear (Tons/Yr)	Road Dust (Tons/Yr)	Road Total (Tons/Yr)
Elemental Carbon (EC)	7	74	81
Organic Carbon (OC)	87	739	826
EC+OC Total	94	813	907

Secondary Organic Aerosols from Light-Duty Motor Vehicles. Various organic compounds in gasoline that are emitted through incomplete combustion or evaporation are precursors to the formation of SOA, which contribute to $PM_{2.5}$. The most significant of these are aromatics with seven or more carbon atoms (Grosjean and Seinfeld 1989).[28] Reformulated gasoline (RFG), in addition to its reduced ozone precursors, has the advantage of containing lower percentages of these aromatic precursor compounds when compared to conventional gasoline (CG). Table 3-15 illustrates this difference, based on information on baseline gasoline for PADD I[29] in 1999 (EPA420-R-00-023, December 2000).

Table 3-15. Baseline (1999) Gasoline in PADD I.

Parameter	Summer CG	Summer RFG	Winter CG	Winter RFG
Aromatics (vol%)	35.4	24.4	30.0	21.5

Only a fraction of the various aromatic species in gasoline contributes to formation of SOA. Typically the fractional aerosol coefficients range between 0.6% and 6.3% (Grosjean and Seinfeld 1989). Table 3-16 estimates mass transformation of gasoline aromatic emissions to SOA in the 15 metropolitan counties.

Table 3-16. Annual LDV SOA Mass Transformation by Model Year Grouping.

	Pre-1981 (Tons/Yr)	1981 – 1985 (Tons/Yr)	1986 – 1990 (Tons/Yr)	1991 – 1997 (Tons/Yr)	1998 – 2004 (Tons/Yr)	Total Emissions (15 counties)
Total Tons	11.00	6.64	25.70	56.07	93.18	192.59

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[28] The significant aromatic precursor compounds are toluene, xylene, ethylbenzene, trimethylbenzene, and ethyltoluene (Grosjean, 1989).

[29] Petroleum Administration for Defense Districts (PADD) are geographic aggregations of the 50 States and the District of Columbia. PADD I includes the States listed below:
PADD IA (New England): Connecticut, Maine, Massachusetts, New Hampshire, Rhode Island, and Vermont.
PADD IB (Central Atlantic): Delaware, District of Columbia, Maryland, New Jersey, New York, and Pennsylvania.
PADD IC (Lower Atlantic): Florida, Georgia, North Carolina, South Carolina, Virginia, and West Virginia.

A majority of the 15 counties addressed in this section already uses Reformulated Gasoline as an ozone-reducing strategy. Taking into account the differences in hydrocarbon emissions by vehicle model year grouping and the difference in the aromatic content between Reformulated Gasoline and Conventional Gasoline, Table 3-17 shows the emissions of aromatics by county.

Table 3-17. Annual LDV Aromatics Emissions by Model Year Grouping.

	Pre-1981 (Tons/Yr)	1981 – 1985 (Tons/Yr)	1986 – 1990 (Tons/Yr)	1991 – 1997 (Tons/Yr)	1998 – 2004 (Tons/Yr)	Total Emissions (15 counties)
Total Tons	478.35	288.46	1,117.58	2,438.05	4,052.02	8,374.46

Federal and State Authority

The U.S. EPA regulates new light-duty vehicle exhaust and evaporative emissions. The current standards, commonly referred to as the Tier 2 standards, set emissions limits for NO_x, HC, PM, and CO. In addition to regulations set by the federal government, the California Air Resources Board sets emissions standards for new light-duty vehicles certified for sale in California. California's program is called the Low Emission Vehicle Program or "LEV." California's authority to adopt its own emissions standards was established in Clean Air Act Amendments of 1965 (CAAA), which also established federal emissions standards for automobiles for the first time. Section 209 of the CAAA gives California the authority to adopt emissions standards. Section 177 provides states in nonattainment of the National Ambient Air Quality Standards outside of California with authority to adopt the California LEV standards. States other than California are preempted from adopting their own emission standards for motor vehicles—they must choose either the federal or the California standards. (See Appendix D for a more complete discussion of the division of authority to regulate mobile sources.) New York State adopted the California LEV standards in the early 1990s.

The LEV program requires automobile manufacturers to meet a fleetwide average for nonmethane organic hydrocarbons as well as a set percent requirement for advanced technology vehicles. The federal Tier 2 program requires automobile manufacturers to meet a fleetwide NO_x requirement. The EPA's MOBILE6.2 model can be used to estimate the difference in emissions between the LEV and Tier 2 fleets. MOBILE6.2 modeling demonstrates that for light-duty fleets in the Northeast, LEV fleets will emit approximately 15% less NO_x and 6% less HC in 2020 when compared to Tier 2 fleets. Particulate emissions are roughly equivalent between the two programs. Some studies have shown, however, that advanced technology vehicles emit lower levels of PM than do conventional vehicles (Southwest Research Institute, 2004).

Technical Options for Emissions Reductions

Reducing Emissions from Older Vehicles. Given the large fraction of the PM inventory in New York from older vehicles, a number of strategies are considered here to reduce emissions from these vehicles.

Establishing a Vehicle Replacement Program. CARB has proposed the development of a Voluntary Accelerated Vehicle Retirement (VAVR) program using remote sensing and high-emitter profiles as screening tools for determining participation in the program. Owners of high-emitting vehicles determined to be unrepairable, relative to the vehicle's value, are provided financial incentives to discontinue using the vehicle and turn it in for scrappage. CARB staff has proposed that PM reductions be calculated based on PM emission factors for different model year vehicles. If a viable in-use method for measuring PM is established, local air districts will have the option of measuring and quantifying PM reductions realized through vehicle retirement/replacement programs. The Carl Moyer program is being used to fund the VAVR program.[30] The methods developed by CARB to quantify the PM emissions reductions from retiring older vehicles could be used by other states adopting similar vehicle replacement programs.

Fund Repairs for Low Income New York State Residents. Owners of light-duty vehicles in New York are required to have their vehicle emissions systems tested periodically to ensure that the vehicles are meeting emissions standards. When a vehicle fails its emissions test, the owner is required to make repairs and have the vehicle retested. The emissions test does not include a PM standard. However, emission control system repairs that are mandated for failure to meet the standard for one pollutant, often have an ancillary benefit of reducing emissions for other pollutants as well. For example, if the air-fuel ratio is not optimally controlled by a properly working oxygen sensor, PM emissions will increase. Maintenance to repair failing vehicles would likely include replacing faulty oxygen sensors in many circumstances. People with low income are disproportionately affected by inspection and maintenance programs as they tend to drive older cars and have limited funds for repair and upkeep. At least two programs exist in the U.S. that fund repairs for low-income vehicle owners.

In Spokane, when vehicle owners fail an emissions test, they are referred to one of 20 repair shops participating in a special repair program, which is funded from two Supplemental Environmental Projects. A similar program in California is paid for by the General Fund. In California from 2001 to 2003, $21 million were spent to repair over 60,000 vehicles. CARB estimates that 1.6 tons per day of NO_X and VOCs have been reduced at a cost of approximately $3,000 per ton of NO_X plus HC. The Spokane County Air Pollution Control Authority estimates the cost of its program to be approximately $1,000 per ton of NO_X plus VOC reduced.

Catalyst Replacement in Older Vehicles. Vehicles sold prior to 1996 are equipped with catalytic converters that are less durable and less efficient at removing hydrocarbons from automobile exhaust, as compared to catalysts manufactured after 1996. Replacing automotive catalysts on older vehicles holds the potential to reduce aromatics and thus secondary $PM_{2.5}$ emissions from cars and light trucks. Since the estimated mass of SOA (see Table 3-16) formed in the atmosphere from light-duty vehicle precursor

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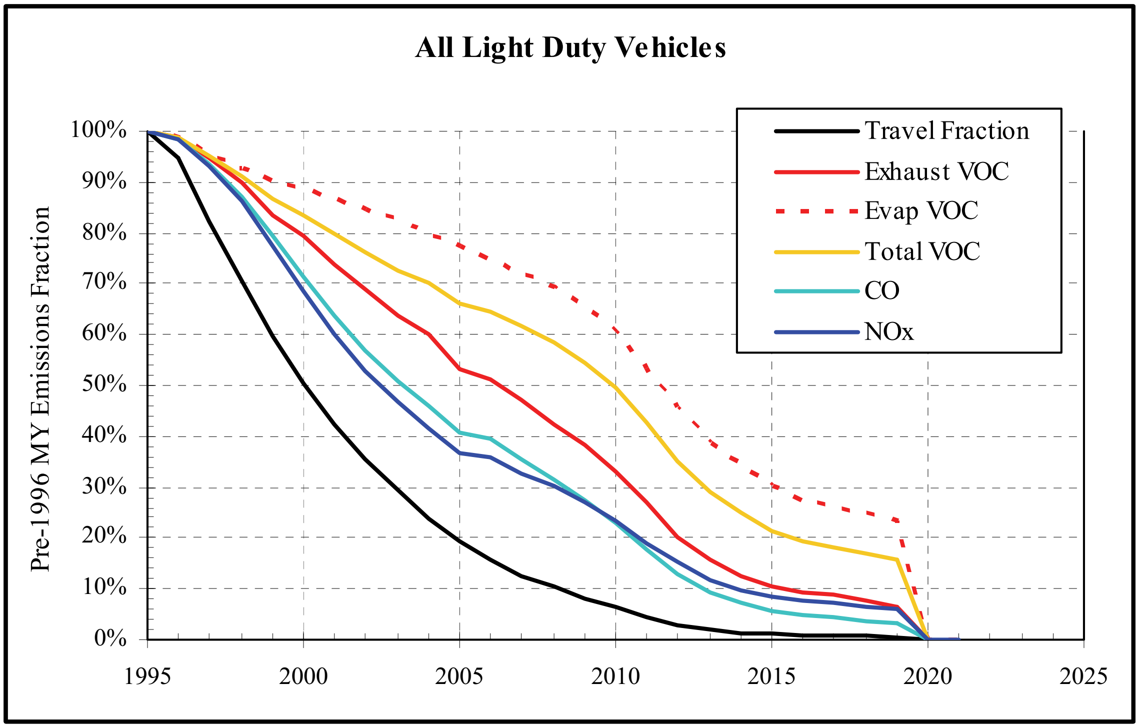
[30] http://www.arb.ca.gov/msprog/avrp/august31_wrkshp/VAVR_Calc_Guidelines.pdf

pollutants is equivalent to half the total mass of light-duty elemental plus organic $PM_{2.5}$ directly emitted (see Table 3-12 to Table 3-13), reducing precursors to SOA is an important strategy in reducing overall light-duty $PM_{2.5}$ emissions. A recent analysis conducted by CARB demonstrated that newer catalytic converters remove 50% more hydrocarbons, compared to catalysts used on pre-1996 vehicles. A recent analysis by NESCAUM of the inventory of vehicles in the Northeast shows that emissions from older vehicles will continue to constitute a large fraction of the light-duty vehicle inventory for the next 10 years. Thus, addressing emissions from this sector of the fleet is an important strategy in reducing overall emissions. Figure 3-6 shows the results of the analysis done for the eight NESCAUM states.

Figure 3-6. Fraction of Emissions from Pre-1996 Model Year (MY) Cars in the Northeast (NE fleet age distribution).



If older catalytic converters in the 15 counties evaluated in this section were replaced, the potential $PM_{2.5}$ reduction in 2007 would be approximately 5 tons per year.[31] This assumes that all cars with less efficient catalysts undergo repair, replacing the old catalytic converter with a new, more efficient model. The cost of purchasing an old-technology component is approximately \$125 to \$350. The cost of purchasing one with new technology is \$200 to \$275. This strategy could reduce $PM_{2.5}$ exposure in both urban and rural counties if it were implemented statewide. As an additional benefit, the associated HC and NO_X reductions would be very high.

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[31] This assumes the most conservative conversion rate from aromatics to PM: 0.6%

Below, additional options to reduce $PM_{2.5}$ from vehicles are discussed.

Adopting Reformulated Gasoline in Non-RFG Counties. Reformulated gasoline has lower aromatics content than conventional gasoline. As discussed above, aromatics contribute to $PM_{2.5}$ emissions as precursors for SOA, and reducing SOAs is an important strategy in reducing overall light-duty $PM_{2.5}$ emissions. In New York State, approximately 46% of VMT occurs in 50 counties that use conventional gasoline. The remaining 12 urban/suburban counties already use RFG and have done so since the early 1990s. The process for opting in to the RFG program is described in the Policy section

As can be seen from Table 3-18, the introduction of RFG as a complete replacement for conventional gasoline in non-RFG counties would reduce aromatics by 4,523 tons per year. The resulting SOA reductions would amount to 104 tons per year.[32] The cost of replacing conventional gasoline with reformulated gasoline typically is between 1 and 5 cents per gallon. The cost per ton of $PM_{2.5}$ reduced assuming a cost increment of 1 cent per gallon of gasoline would be equivalent to a $PM_{2.5}$ reduction cost-effectiveness of $270,000 per ton. If the substantial VOC plus NO_X reductions are included, the program is highly cost-effective ($1,600/ton of NOx plus HC reduced). The introduction of RFG statewide is estimated to reduce VOC plus NO_X in New York by 58.9 tons per summer day (Meszler 2006). In addition, the introduction of RFG in attainment counties would reduce $PM_{2.5}$ exposure in both urban and rural areas in upstate New York.

Table 3-18. Potential Emissions Reductions with Statewide RFG Introduction (tons per year).

	Current RFG Fraction	Potential VOC Reduction	Potential Aromatics Reduction	Potential SOA Reduction
New York	54%	17,070	4,523	104.0

Advanced Technology Light Duty Vehicles. A number of light-duty vehicle models that emit substantially lower criteria pollutants and greenhouse gases than conventional vehicles are available on the market. The LEV program requires the introduction of super ultra low emitting vehicles or "SULEVs." SULEVs are vehicles that are certified to very low tailpipe emissions standards (70 percent lower for HC and NOx). In addition, as mentioned above, SULEVs emit less $PM_{2.5}$—although they are not required to in order to be certified. The LEV program also requires the introduction of several types of advanced technology vehicles. Partial Zero Emitting Vehicles (PZEVs) and Advanced Technology Partial Zero Emitting Vehicles (AT PZEVs) must meet SULEV exhaust emissions standards and emit 35 percent fewer evaporative emissions over the life of the vehicle. Hybrid electric and natural gas light-duty vehicles currently qualify as AT PZEVs. A study conducted for NESCAUM by the Southwest Research Institute

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[32] This assumes that 30% of aromatic emissions are reduced by the introduction of RFG and 2.3% of aromatic emissions are converted to SOA.

(SWRI) to evaluate $PM_{2.5}$ and other emissions from new and aging vehicles concluded that emissions from SULEVs were 75 percent lower than from regular low emitting vehicles and Tier 2 vehicles (SWRI 2003). In addition to emitting fewer pollutants, some advanced technology vehicles provide consumers with a substantial cost savings in the form of lower monthly fuel costs. A 2004 Northeast States Center for a Clean Air Future (NESCCAF) study demonstrated that at $2.00 per gallon of gasoline, purchasers of hybrid-electric vehicles will recoup the higher initial cost of purchasing a hybrid-electric within the operating life of the vehicle.

Other vehicles currently in the pilot demonstration stage, such as plug-in hybrid-electrics, reduce emissions significantly. A study released by the New York Department of Transportation (DOT) noted that the average trip length in New York State is eight miles. Given this, the widespread introduction of hybrid-electric vehicles with plug-in capabilities could eliminate internal combustion engine emissions for a significant amount of vehicle trips. Furthermore, hybrid-electrics provide a benefit to consumers in the form of substantially lower monthly operating costs. The Electric Power Research Institute (EPRI) estimates that plug-in hybrids have the potential to reduce gasoline consumption from 50 to 80 percent.

In addition to reducing criteria pollutants and fuel consumption, hybrid-electric vehicles can potentially reduce $PM_{2.5}$ emissions related to brake wear. Studies on heavy-duty vehicles have shown that brake wear is reduced in hybrid-electric heavy duty vehicles due to regenerative braking. Since PM emissions from brake wear are significant in New York, increasing the number of hybrid-electrics might be considered as a strategy to reduce this source of $PM_{2.5}$.

Improved Technology Street Sweepers. Using improved-technology street sweepers may be an option to reduce PM from paved surfaces. While sweepers have historically been employed for general sanitation and for control of contaminants in storm water runoff, they have in recent years received increasing attention as a means to control airborne particulates. In 1997 the South Coast AQMD adopted Rule 1186, requiring cleaning of paved roads and other control measures (such as traffic calming and chemical dust suppression) for unpaved roads. This rule also provides for the testing and certification of “PM10-Efficient” street sweepers. To gain certification a sweeper must demonstrate an 80% pick-up efficiency and entrain no more than 200 mg of PM_{10} per mile. The South Coast AQMD maintains a list of certified sweepers. While these sweepers have not been evaluated for effectiveness at removal of $PM_{2.5}$, the technology may warrant further investigation since fugitive dust is a contributor to overall $PM_{2.5}$ from light-duty vehicles, and since advanced sweepers are very effective at removal of PM_{10}.[33]

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[33] Midwest Research Institute, "Background Document for Revisions to Fine Fraction Ratios Used for AP-42 Fugitive Dust Emission Factors," 2006 estimates that 15% of PM_{10} road dust is $PM_{2.5}$.

Recent improvements in street sweeper technology, spurred in part by South Coast AQMD's Rule 1186, have dramatically increased the effectiveness of commercially available sweepers at removing particulates from roads. Mechanical sweepers, which use rotating brooms, mechanical conveyance, and a liquid spray for dust suppression, are the most common type in current use; they are far less efficient than more advanced designs. Vacuum-assisted sweepers also use brooms and liquid dust suppression, but they use a vacuum to pick up and transport the material into a hopper. Regenerative-air sweepers clean by directing high-pressure air at the road surface to loosen particles, then vacuuming the particles into the hopper. Advanced dry sweepers use a mechanical brush coupled with vacuum or mechanical conveyance, but instead of liquid dust suppression they are equipped with specialized filtration to prevent dust from escaping. The combination results in very high pick-up efficiency.

The Federal Highway Administration found that the efficiencies of available sweepers vary widely. The most effective vacuum-assisted dry sweeper is capable of removing over 99% of PM10, and the least-effective mechanical sweeper actually entrains more PM than it removes. Breault et al. (2005) found that a vacuum sweeper was from 1.6 to 10 times more efficient than a mechanical sweeper for removing larger particles (63µm to 2mm in size). In-use measures may also be effective at reducing PM from paved surfaces. These include tandem sweeping (in which a mechanical sweeper is followed by a vacuum-assisted sweeper) and enforcement of an optimal sweeper vehicle speed.

The Federal Highway Administration estimates that sweeper costs may range from around $70,000 (1995 dollars) for a conventional mechanical sweeper, to $127,000 (1995 dollars) for a regenerative-air unit, to around $170,000 (1996 dollars) for a vacuum-assisted dry sweeper.

Policy Options

A number of policies could be used to spur introduction of technologies to reduce light-duty vehicle emissions. These are discussed below.

Introduction of RFG Statewide. The Energy Policy Act of 2005 established authority for New York and other states in the Ozone Transport Region to opt into the reformulated gasoline program for counties in attainment of the National Ambient Air Quality Standards. Prior to the 2005 Act, areas in attainment of the NAAQS were prohibited from opting into the program. According to the Act, governors must indicate to the EPA by way of a letter that counties in attainment will be opting into the program. A letter informing the EPA is the only action required.

Catalyst Replacement Program. The California Air Resources Board is conducting a proposed rulemaking in order to specify criteria intended to ensure that new aftermarket catalytic converters installed in existing vehicles will have sufficiently high conversion efficiency to meet vehicle emissions standards for a minimum of 50,000 miles and will be compatible with the on-board diagnostic systems in motor vehicles. If finalized by California, this requirement could be adopted by New York State under its current authority. CARB estimates that more efficient catalysts reduce HC and NO_X emissions 50 percent more than current catalysts. The amount of pollution reduced each year would decline as replacement of older vehicles with newer vehicles reduces overall VMT from older vehicles. Thus, early introduction would maximize the benefits of this program. California is also considering a voluntary replacement program. To fund part of the program being considered, the State of California will pay for replacement of an aging catalytic converter in pre-1996 vehicles. New York could consider a similar program.

Spurring the Introduction of Advanced Technology Vehicles. New York State has adopted the California LEV program and the Zero Emission Vehicle (ZEV) requirement. New York State is also spurring introduction of advanced technology vehicles through Executive Order 111, which requires state fleets to purchase these types of vehicles. A total of $35 million has been designated in Environmental Bond Act funding for purchase of hybrid and other clean vehicles in government fleets.

Municipalities and authorities in New York State are also working to introduce advanced technology vehicles. The NYPA/Th!nk Clean Commute program has placed nearly 100 electric vehicles in service. The NYPA/MEUA (Municipal Electric Utility Association) municipal and electric cooperative association has received funding toward the purchase of commercially available electric drive vehicles. Another example is the New York City hybrid taxicab program. The Taxi Commission has designated six models of hybrid-electric vehicles that can legally be operated as taxicabs in New York City. The Taxi Commission has authority to designate which vehicle models can be used as taxis in New York City; prior to the designation of hybrid electrics as eligible, these vehicles could not be used as taxis. Expanding these programs could provide opportunities to further reduce emissions. The New York City taxicab program could be expanded to other areas of the state—such as Albany—where taxis are licensed.

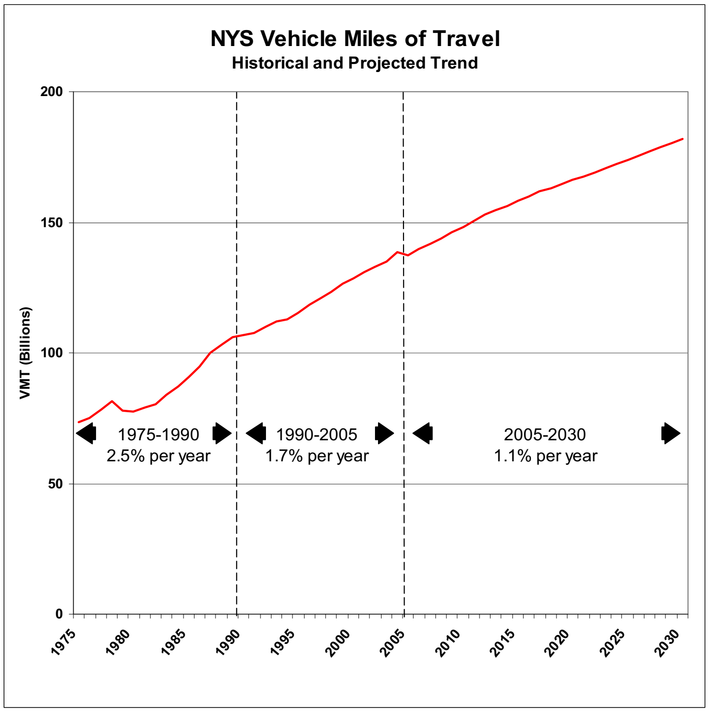
Establishing funding mechanisms for businesses that purchase low emitting, advanced technology vehicles could spur introduction of hybrids and plug-in hybrids. One example is the System Benefits Charge (SBC), which could provide a revolving loan to businesses that purchase hybrids. This could save fuel and reduce emissions. The loan could be paid back to the utility from monthly fuel savings realized from the better gasoline mileage achieved by hybrid-electric vehicles. Publicly available data on vehicle sales in northeastern states show that nearly 25% of vehicles sold in 2006 are registered as SULEVs. Given the high penetration rate and consumer acceptance of these advanced technology vehicles, putting in place a program to spur greater introduction of these vehicles is feasible.

Reducing Vehicle Miles Traveled (VMT). Vehicle miles traveled in New York will increase dramatically in 25 years: 28% or by about 40 billion miles per year between 2005 and 2030, as shown in Figure 3-7. The

VMT increases are projected to largely erase gains made in criteria pollutant emissions reductions from the light-duty motor vehicle emissions control programs at the federal level.

Figure 3-7. New York State Vehicle Miles Traveled.

Source: NYS Department of Transportation (2005)



Measures to reduce VMT growth, while difficult to implement effectively, can greatly reduce emissions and fuel consumption while providing a cost savings to consumers. A recent study evaluated 3,100 travel analysis zones in California and Illinois and concluded a doubling of housing density results in a 20 percent decrease in VMT (Burer et al 2006). The study also concluded that very large cost savings to consumers can be realized with a reduction in dependence on personal car use. A recent meta analysis of studies on the influence of density on VMT concluded that a doubling of density results in 33% fewer vehicle miles traveled (Ewing et al 2007).

Several policies are in place to encourage VMT reductions in the U.S. and in Europe. Massachusetts is encouraging so-called “Smart Growth” by providing preferential funding of public works and other grants to municipalities that adopt Smart Growth measures. Maryland has established a smart growth policy through legislation to promote an orderly expansion of growth and an efficient use of land and public services. The program capitalizes on the influence of state expenditures on economic growth and development. Priority Funding Areas are established in existing communities and places where local governments seek state investment to support future growth. Growth-related projects include programs that encourage or support growth and development such as highways, sewer and water construction, and economic development assistance. Congestion charging, a system under which tolls are charged for use of roadways within congested areas of cities, has been introduced in London, Stockholm, and Edinburgh. Congestion charging induces motorists to consider the economics of alternative modes of transportation. The London congestion charge has resulted in a 30% reduction in congestion along with a 15% reduction in particulate emissions. The congestion pricing program currently being considered for New York City by the State Legislature and the City of New York could reduce $PM_{2.5}$ emissions.

Recommendations

Based on the above analysis of particulate emissions from light-duty vehicle exhaust, brake and tire wear, road wear, and re-entrained road dust, the following viable strategies are recommended for reducing carbonaceous $PM_{2.5}$ and exposure to the same in New York State:

- Expand programs to accelerate market penetration of advanced technology vehicles such as SULEVs and hybrid-electrics. Several initiatives, including the New York City hybrid-electric taxicab program and Executive Order 111, could be expanded to encourage greater numbers of advanced technology vehicles.
- Introduce reformulated gasoline statewide as a low-cost means to reduce secondary organic aerosols in attainment areas. This strategy will improve air quality in areas outside of designated New York State nonattainment areas.
- Develop programs targeting older vehicles, accelerating their replacement, upgrading catalytic converters, and subsidizing vehicle replacement as well as motor vehicle inspection and maintenance (I&M)-related repairs for low-income vehicle owners.
- Consider development of new or expansion of existing strategies to reduce vehicle miles traveled in the New York nonattainment areas. New York City's "plaNYC" provides a plan for reducing congestion and vehicle miles traveled in the city (City of New York 2007). A smart growth program, patterned after the Massachusetts program, and London’s congestion pricing program, could serve as effective models for smart growth and congestion pricing.
- Evaluate the potential of new-technology street sweepers to cost-effectively reduce $PM_{2.5}$. If promising, consider requirements for the purchase of regenerative air or vacuum assist sweepers.

ONROAD HEAVY-DUTY VEHICLES: TRUCKS AND BUSES

This section discusses the current and projected inventory of direct $PM_{2.5}$ emissions from onroad heavy duty vehicles in New York State. It reviews available regulatory authority to address emissions from new and existing vehicles, and the control strategies that state and local authorities could consider to reduce emissions from existing heavy-duty vehicle fleets. Some of the strategies discussed involve the use of various diesel retrofit technologies, which are described in Appendix B.

Sector Profile

The onroad heavy-duty vehicle sector is composed of a wide variety of vehicles, from 18-wheel tractor-trailer combinations, to school and transit buses, to dump trucks and refuse haulers. The vast majority are powered by diesel engines. Gasoline powered heavy-duty vehicle $PM_{2.5}$ emissions in New York State total less than 15% of the $PM_{2.5}$ emissions from diesel trucks and buses. Heavy-duty vehicles are categorized by weight class. In terms of numbers of vehicles, and especially fuel used annually, the heaviest Class 8 vehicles dominate. The majority of these Class 8 trucks are long-haul tractor-trailers used to move goods over the nation's highways. Other examples of Class 8 vehicles include transit buses and refuse haulers. As shown in Table 3-19, in 2002, there were over 2,082,600 Class 8 trucks on the road in the U.S. (U.S. Census Bureau 2004), and they used over 17 billion gallons of diesel fuel (Davis 2004).

The number of miles traveled annually by Class 8 trucks is expected to increase by approximately 40 percent through 2020 (EPA 2000a). While future emissions regulations will mitigate the air quality impacts of increased miles traveled, heavy-duty onroad vehicles are, and will remain, a significant contributor to direct $PM_{2.5}$ emissions.

Table 3-19. 2002 U.S. Heavy Truck Statistics.

Source: Census Bureau 2004, Davis 2004

Class	# Trucks (thousands)	VMT* (million miles)	Fuel Use (million gal)
2B	396.7	5,031.2	318.2
3	621.1	8,428.6	1,075.1
4	287.3	4,184.2	533.7
5	291.1	3,949.2	503.7
6	855.8	11,361.3	1,449.1
7	419.1	5,726.7	995.9
8	2,082.6	100,167.0	17,420.3

* VMT = Vehicle Miles Traveled

The vast majority of $PM_{2.5}$ emissions from this sector comes from the combustion of diesel fuel in diesel engines. $PM_{2.5}$ emissions from tire and brake wear are much less important from heavy-duty vehicles than from cars and light trucks, accounting for less than 3% of the $PM_{2.5}$ emissions. In New York State in 2003, there were 11,800 registered truck-tractors, 61,000 buses (including school buses), and 39,000 farm trucks (U.S. DOT-FHA 2006). Exhaust from heavy-duty onroad diesel vehicles creates over 3,000 tons of direct $PM_{2.5}$ emissions annually in New York State; it accounts for approximately 51% of the direct $PM_{2.5}$ emissions from all onroad vehicles and approximately 2% of total direct $PM_{2.5}$ emissions (including emissions from nonroad vehicles and from stationary sources). Within the New York City tri-state metropolitan nonattainment area, diesel exhaust from trucks and buses is relatively more important compared to other emission sources, accounting for almost 5% of total direct $PM_{2.5}$ emissions. On a mass basis this $PM_{2.5}$ from onroad diesel vehicles is composed of approximately 75% elemental carbon and 23% organic carbon (EPA 2005g). The proportion of organic carbon varies depending on engine technology level, fuel, duty cycle, and the state of engine tuning (EPA 2004h). Older vehicles with two-stroke engines, and poorly maintained newer vehicles, are likely to emit PM with a higher organic carbon content due to the combustion of greater amounts of lube oil in the engine. Most of the remainder of the $PM_{2.5}$ mass (approximately 2%) is sulfates.

There are almost ten times as many buses and farm vehicles as truck-tractors registered in New York State, and these vehicles are relatively more important to the on-road emissions inventory in New York State than nationally. However, in-state vehicle registrations do not give a complete picture of potential emissions from truck-tractors, since goods movement within New York can take place with out-of-state vehicles. The 2002 Commodity Flow Survey conducted by the U.S. Department of Transportation indicates that 250 million tons of freight shipments originated in New York in 2002, and that 90% of this tonnage was carried by truck. In the same year, 286 million tons of freight originating in other states was delivered into New York, and 81% of this tonnage was carried by truck. Nationally, only 67% of freight tonnage is carried by truck (U.S. DOT-BTS 2005). It is important to note that given the large amount of truck traffic from vehicles registered outside of the State, a regional approach to reducing truck emissions may be needed.

Tractor-trailers account for a significant portion of vehicle traffic on many State roadways. Vehicle counts conducted by the New York State Department of Transportation between 1999 and 2004 indicate that on some segments of interstate highways heavy vehicles account for as much as 35% of the total traffic, with a statewide average of 26% in rural areas and 14% in urban areas (NYSDOT 2005). Many of these trucks are tractor-trailers used for goods movement. Truck traffic contributes to elevated $PM_{2.5}$ exposure near roadways.

Sector Emissions

According to the EPA’s 2002 National Emissions Inventory, over 60% of the heavy-duty onroad vehicles in New York are new enough (built in 1994 and later) that their engines were designed to comply with the most stringent EPA PM standards now in effect (0.1 g/bhp-hr). About 15% of vehicles have engines that were regulated to a more lenient standard, and between 14% and 22% of vehicles have engines that are essentially unregulated for PM emissions because they were built prior to 1989 (see Table 3-20). Targeting the oldest vehicles in New York’s heavy-duty fleet could substantially reduce $PM_{2.5}$ emissions from this sector.

Table 3-20. Age Distribution of Heavy-Duty Onroad Vehicles in 2002 National Emissions Inventory (New York State).

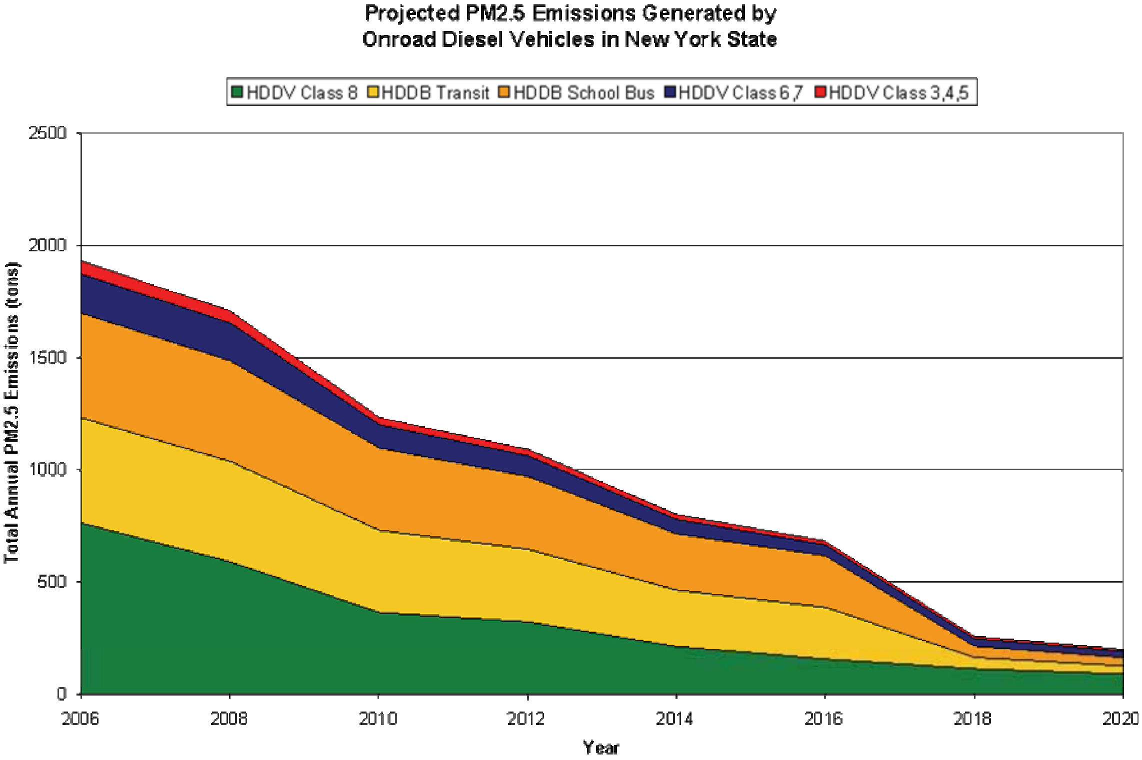
Source: EPA 2002 National Emissions Inventory

	Vehicle Type			
Year of Manufacture	**HDV Class 3 ,4**	**HDV Class 5,6,7**	**HDV Class 8**	**Transit & School Buses**
Later than 2003	13%	11%	11%	10%
1994 – 2003	54%	51%	51%	56%
1991 – 1993	9%	10%	10%	13%
1989 – 1990	5%	6%	6%	6%
Earlier than 1989	19%	22%	22%	14%

Figure 3-8 shows a breakdown of emissions by vehicle class for the years 2006 to 2020, based on data from the EPA's National Emissions Inventory). Approximately one-third of heavy-duty truck $PM_{2.5}$ emissions in 2006 are from Class 8 trucks—those weighing more than 26,000 pounds. Approximately 25% of $PM_{2.5}$ emissions are from transit buses, one-quarter from school buses, and 10% from heavy-duty Classes 3 to 7. Figure 3-8 shows that Class 8 trucks, school buses, and transit buses also dominate the heavy-truck $PM_{2.5}$ inventory in the latest year emissions were modeled, 2020. The data used to generate this graph do not account for the extensive transit bus retrofit program in New York City. To date, approximately 4,000 transit buses have been retrofitted with particulate filters; an additional 2,000 other New York City buses—the remainder of the City's fleet of buses—are scheduled to be retrofitted. After this program is completed, only 2,000 non-retrofitted transit buses will remain in New York State.

Figure 3-8. Expected Change in $PM_{2.5}$ Emissions from Onroad Diesels Through 2020.

Source: U.S. EPA 2002 National Emissions Inventory



Because the emissions reductions achieved from the transit bus retrofit program were not factored into the modeling analysis, emissions from transit buses are in reality much lower than pictured in Figure 3-8. It is also important to note that national default mileage values for school buses were used for this modeling, and school bus mileage in New York State is somewhat lower than the national average. Because of this, school bus $PM_{2.5}$ emissions in New York State are also likely lower than the amount shown in Figure 3-8. If these factors are taken into consideration, Class 8 trucks represent the majority of heavy-duty truck $PM_{2.5}$ emissions in New York State and will continue to do so until 2020.

While emissions from class 8 trucks dominate the $PM_{2.5}$ inventory, from an exposure standpoint, diesel vehicles that tend to be old, operate in densely populated areas, carry large numbers of passengers or carry passengers that are categorized as "sensitive populations" are a concern. Examples are school buses, tour buses operating in New York City, private tour buses. A priority could be placed on retrofitting and reducing idling emissions from these vehicles.

Federal and State Authority

The EPA has primary authority to regulate emissions from new heavy-duty trucks and buses, while California has authority to adopt its own emissions standards for these vehicles, and other states can adopt either the California or the federal standards. See Appendix D for a more complete discussion of the division of authority to regulate mobile sources.

Because the on-road heavy-duty fleet turns over much more slowly than the light-duty fleet, there are still significant numbers of vehicles with unregulated or marginally regulated pre-1990 engines. Even so, over the next 15 years current and pending EPA regulations will begin to have a significant effect. Based on normal fleet turnover, the EPA estimates that annual PM emissions from heavy-duty onroad vehicles will fall by 37% through 2010, and by 84% through 2020, as new vehicles enter the fleet and older vehicles are retired. This decline will occur despite a projected 40% increase in annual heavy-duty vehicle miles traveled through 2020. To reduce exposure to onroad diesel $PM_{2.5}$ emissions in the near term, the use of clean fuels, retrofits, engine rebuilds, engine repowering, and vehicle replacement programs will be needed.

As discussed further below, New York State and some New York counties and municipalities currently have regulations related to particulate emissions from onroad diesel vehicles. New York State law prohibits onroad heavy-duty vehicles from idling more than five minutes, with some exceptions (NYSR 217-3). New York has also set limits for smoke opacity from diesel vehicles and has implemented an opacity-based inspection and maintenance program for these vehicles, which applies in counties that are designated as being in nonattainment of the National Ambient Air Quality Standards for ozone (NYSL Art 19-0320; NYSR 217-5). New York State requires that for certain centrally fueled state fleets, a specified percentage of new heavy-duty vehicles purchased after 1998 must be “clean fuel vehicles” (NYSL Art 19-0319).

New York Governor Pataki recently signed into law the "Diesel Emission Reduction Act of 2006." The law will require that, beginning in 2008, highway and construction vehicles owned by the State, operated on behalf of the State, or leased by a State agency or State and regional authority be retrofitted with best available control technology. Assuming the Act will affect 30% of construction equipment and require retrofit with particulate filters, approximately 1,000 tons of carbonaceous $PM_{2.5}$ could be reduced in the NAA from this measure.

The New York City Administrative Code prohibits the emission of an “air contaminant” with greater than a certain smoke density, which applies to all sources, including onroad vehicles (NYC AC 24-141:142). It specifically prohibits a motor vehicle from emitting a “visible air contaminant” while stationary for longer than 10 seconds or after the vehicle has moved more than 90 yards after being stationary (NYC AC 24-143). New York City prohibits any motor vehicle from idling for more than three minutes (with some exceptions) and prohibits buses from idling while stopped at a “terminal point on an established route” (no time limit specified) (NYC AC 24-163). Suffolk County also has idling and opacity limits.

In addition, New York City has mandated that all city-owned diesel vehicles, vehicles used to fulfill the city's solid waste and recyclable materials contracts, school buses, and sightseeing buses use ultra-low-sulfur diesel fuel and best-available retrofit technology to reduce PM emissions (NYC AC 24-163.4, 163.5, 163.6, 163.7). These mandates have different starting dates and phase-in schedules. Finally, beginning in 2005 the city has mandated that at least 20% of buses purchased by the city be alternative fuel buses. Another city regulation requires that a program be implemented to test the feasibility of alternative fuel street-sweeping vehicles (NYC AC 24-163.2). Rockland and Westchester Counties have also passed best-available retrofit technology requirements for heavy-duty vehicles.

The City of Buffalo also has established limits on emissions of "dense smoke" from sources that include on-road vehicles (BUF CC 66-3; 66-25). These restrictions are similar to those in effect in New York City. With some exceptions, the Laws of Rockland County require the county to purchase diesel vehicles that use the best available retrofit technology (BART), or that meet equally stringent PM standards, and require such vehicles to use ultra-low-sulfur diesel fuel (no more than 15 ppm sulfur). Rockland County is subject to a purchasing schedule that would require all county-owned diesel vehicles to be low-emitting by 2012 (LRC, chap. 137).

Technical Options for Emissions Reduction

Retrofits. As described in Appendix B, there are a number of retrofit technologies that can be installed on existing diesel vehicles to reduce PM emissions. The two most common types of devices are diesel oxidation catalysts (DOCs), which can reduce PM from a diesel vehicle by 10-30 percent, and diesel particulate filters (DPFs), which can reduce PM from a diesel vehicle by 80-90 percent. When applied to an older onroad vehicle, a DOC can reduce annual PM emissions by 2 to 20 pounds depending on duty cycle, and VOCs up to 80%. A DPF can reduce annual PM emissions by 13-60 pounds. In general, DOCs are much easier to install on a vehicle than a DPF, however, DOCs do not reduce primary elemental carbon emissions. Conversely, uncatalyzed DPFs do not reduce primary organic carbon or VOC emissions. Thus, DPFs should be catalyzed whenever possible since primary organic carbon and secondary organic aerosols (formed from VOCs) contribute to health problems and poor air quality. The use of active DPFs can result in a fuel economy penalty of a few percent. Passive filters do not lower fuel economy.

Using alternative fuels or improving diesel fuel quality is also an important strategy in reducing diesel PM emissions. The three main options for reducing diesel emissions are not mutually exclusive. Some commercially available retrofit devices work better in conjunction with, or even require the use of, an alternative diesel fuel. Commercial products are also available that combine engine modifications with retrofit devices, while engine repowering can always be combined with retrofits to further reduce emissions. The more viable retrofit options include tailpipe controls, such as diesel oxidation catalysts and diesel particulate filters, and crank case ventilation filters.

Idling Reduction. The EPA estimates that idling Class 8 trucks use, on average, 0.82 gallons of diesel fuel per hour and produce 3.7 grams per hour of PM emissions. Cab climate control and electrical power on tractor-trailer trucks are usually supplied by the truck's diesel engine. For this reason, long-haul truckers often keep their trucks idling all night as they rest in the sleeper cab. This idling is exempt from current New York State and New York City idle restriction laws.

The new federal engine standards require 2008 and newer model year heavy-duty diesel engines to be equipped with a nonprogrammable engine system that automatically shuts down the engine after five minutes of idling. Alternatively, the engine emissions can meet a stringent NO_X idling emission standard. California's in-use truck standards require operators of both in-state and out-of-state registered sleeper-berth-equipped trucks to manually shut down their engine when idling more than five minutes at any location within California beginning in 2008. CARB staff anticipate that truck manufacturers will respond by installing auxiliary power units (APUs) or other methods to supply power to auxiliary loads as standard equipment on trucks equipped with sleeper cabs. APUs and other methods of providing necessary electrical power for climate control, lighting, and other amenities while reducing or eliminating the need for extended overnight idling are discussed in Appendix B. EPA testing has shown that use of these technologies can reduce total PM emissions by 90–99% compared to idling the truck's main engine.

These idling reduction methods require additional equipment to be installed either on the truck, at truck stops, or both. The EPA's review of currently available devices indicates that on-board idle reduction technologies can cost between \$500–\$8,500 per truck, with the auxiliary engine systems (which provide the greatest flexibility) generally costing \$5,000 or more. However, the payback period for these devices can be less than two years as a result of fuel savings (see Appendix C). Services at truck stops that provide cabin heating and cooling, electricity, and other amenities to eliminate the need for on-board equipment generally charge between \$0.50–\$1.50 per hour, depending on the services provided and the contracting arrangements that apply (EPA 2005d). With diesel fuel costing \$2.50/gallon, truckers using these services instead of idling their main engines can save \$0.50–\$1.50 per hour.

NYSERDA recently demonstrated on-board electrification technology at a truckstop in Wilton, NY, in collaboration with Shurepower, LLC (NYSERDA, 2005). A class 8 truck was retrofitted with electric accessories, and a truck parking space was electrified with shore power infrastructure. The demonstration was successful, and user feedback was very positive, though total operating experience was limited. NYSERDA estimated that 17.4 pounds of PM were avoided over the course of one year, with an average use of 24%. Also recently, a collaborative led by Clean Air Communities demonstrated off-board idle-reduction technology at the Hunt's Point market in the Bronx, NY (NESCAUM, 2004). Twenty-eight parking spaces were fitted with IdleAire electrification units. The systems were shown to operate successfully, and 126 pounds of PM were avoided during the year-long demonstration. However, as with the Wilton project, total usage was limited. These projects illustrate both the potential of idle reduction technologies to reduce emissions from long-haul trucks, and some of the challenges involved in introducing a new infrastructure to an industry with well-established practices.

Although implementation remains a challenge, the air quality benefits of idle reduction may be significant. Recent research at the University of California – Davis compared energy and emissions impacts of idling and several idle-reduction technologies (Solomon, 2006). The study found that all IR options can provide substantial reductions in lifecycle emissions and energy use. It also found that emissions from electrification depend strongly on the fuel mix of the local electric grid, and that comparative impacts of idling and IR strategies may vary with climate and location of truckstops and powerplants. For example, New York State has a relatively low fraction of coal in its generation mix, so total lifecycle PM emissions from electrification are comparatively low; however, New York's coal plants are disproportionately located near areas of high population density, which may offset the benefits of electrification in some cases. The study found that fuel-fired cabin heaters may provide dramatic benefits over idling, especially in locations with high heating loads, such as in New York. The greatest overall energy and emissions benefits were found with hybrid systems, combining cabin heaters with APUs, battery-electric cooling, or TSE.

Clean Fuels. Some alternative formulations of diesel fuel produce 20 to 50% lower emissions of PM in existing engines than standard on-road diesel fuel, and they can often be used without any physical changes to the engine. The most common of these cleaner fuels (e.g., emulsified diesel, oxy-diesel, bio-diesel, synthetic diesel) are described in Appendix B.

While it is technically possible to modify existing diesel engines to operate on natural gas, using a purpose-built natural gas engine is generally preferred. Given the cost and the current limited availability of natural gas fueling stations, conversion to natural gas is most appropriate for fleet vehicles that are centrally fueled and that operate within a limited geographic area.

Cost-effectiveness figures for some of the options discussed above are provided in Table 3-21.

Table 3-21. Select Control Strategies for Trucks and Buses.

Source: See Appendix B &C

Control Strategy	Annual Reductions (lbs/vehicle)	Cost/ton
DOC retrofit	3 - 20	$29,000 - $415,000
DPF retrofit	21 - 60	$60,000 - $243,000
Emulsified Fuel	4 - 33	$95,000 - $1,500,000
Biodiesel Fuel	3 - 13	$159,000 - $1,600,000
Cabin heater	5	0[34]

Policy Options

New York State has funded vehicle retrofits and replacements using funds raised from the sale of State general obligation bonds. Since 1996, New York State has operated a grant program using funds from the 1996 voter-approved Clean Water/Clean Air Bond Act. Approximately $230 million were obligated for a wide range of air quality projects, including retrofits and fuel conversions of onroad diesel vehicles (NYSDEC 2005).

In New Jersey, a diesel retrofit program adopted by the State Legislature in June 2005 and approved by a substantial margin in the November 2005 general election will be funded by reapportioning part of the State corporation business tax that is currently dedicated to the remediation of leaking underground fuel storage tanks. This is expected to generate $14 million per year for retrofits (Iavarone 2005).

In California, CARB has finalized four regulations aimed at reducing diesel truck emissions by requiring the use of clean fuels and retrofit devices. They include: transit bus, trash trucks, transportation refrigeration units (TRUs), and bus and truck idling regulations. These rules will reduce diesel PM emissions by 85% in the affected vehicles. While the New York Diesel Emission Reduction Act of 2006 aims to reduce emissions from vehicles operated by or leased to the State, vehicles operated by municipalities will not be affected. Thus, adopting the CARB regulations could provide an additional emissions benefit. In addition, some of the vehicles regulated by CARB include private fleets that service privately owned industry. Trash trucks that haul commercial trash are an example. The transit bus regulation would likely not provide much of an additional emissions benefit in New York State, given that approximately 6,500 of the State's 8,000 transit buses have been or will be retrofitted with particulate filters.

In 2004, CARB adopted new in-use emissions rules for transportation refrigeration units (TRUs) that essentially mandated either the use of alternative fuels or retrofit of existing units with aftertreatment

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[34] If fuel savings are factored in, a cabin heater would result in an approximate $6,000 savings over a seven-year period for each truck operator. It would take approximately 57 trucks equipped with cabin heaters to achieve a one ton $PM_{2.5}$ reduction over a seven year period. The total cost savings would be more than $300,000 if 57 trucks are equipped with cabin heaters.

devices. TRUs are used to cool trailers that carry frozen or perishable goods, and they typically are powered by their own small diesel engines (generally smaller than 25 horsepower). The engines used in TRUs have their own EPA regulatory regime, with standards that are generally more lenient than those for the engines used in trucks. The CARB regulations set an in-use standard more stringent than the EPA standard for new engines, and will therefore require retrofit of existing units built after 2001 with aftertreatment, beginning in 2008.

The CARB trash truck regulation requires waste collection companies and government agencies that operate their own waste collection fleets to phase-in cleaner trucks from 2004 through 2010. By 2015, according to CARB's research, the trash truck regulations will help reduce PM emissions from these vehicles by as much as 85% from their 2000 levels.

The existing voluntary retrofit program funded by New York’s Environmental Bond Act could be expanded to private fleets and supplemented with additional funding—through the sale of new bonds or by dedicating some other revenue stream. A number of states have also used funds collected in environmental enforcement actions to fund diesel retrofit projects. New Jersey’s program is of particular interest—the combination of mandatory retrofits of private fleets with state funding is an approach that New York could follow as well. Given the current vehicle mix in New York, it would be appropriate to focus voluntary retrofit efforts on large Class 8 vehicles, including tractor-trailer trucks, as well as refuse trucks and charter buses.

Mandatory Retrofits. New York City has mandated that all city-owned diesel vehicles, vehicles used to fulfill the city’s solid waste and recyclable materials contracts, school buses, and sight-seeing buses use ultra-low sulfur diesel fuel and best available retrofit technology to reduce emissions.

The 2005 New Jersey program requires BACT retrofits for all diesel school buses, commercial buses, garbage trucks that are publicly owned or used on a publicly funded contract, and other publicly owned onroad diesel equipment. Unlike the California and New York City legislation, the New Jersey program includes provisions to reimburse fleet owners for the cost of the retrofits.

New York State and New York City have already implemented limited retrofit programs for onroad heavy-duty diesel vehicles. To achieve greater emissions reductions, these existing programs could be made more stringent or expanded. The mandatory New York City retrofit requirements could be extended to other private vehicle fleets, or could cover fleets statewide. New York State could also adopt the CARB rules for transportation refrigeration units.

Other possible mandatory actions to reduce emissions from existing diesel vehicles include maximum vehicle age requirements for contracted government services such as school busing, and increased taxes or registration fees for older vehicles. These types of policies are designed to encourage retirement of the oldest, most polluting diesel vehicles.

Inspection and Maintenance. Numerous studies have shown that a minority of vehicles create a disproportionate share of vehicle emissions (Niemeir et al. 2004). These high emitters are vehicles that are poorly maintained, misadjusted, or have malfunctioning emissions control equipment. Inspection and maintenance programs are designed to identify high emitters and to require vehicle owners to repair them. New York State has a periodic heavy-duty vehicle inspection program for all vehicles greater than 8,500 lbs gross vehicle weight. Annual emission inspections are required for vehicles registered in the nine-county New York Metropolitan Area. This program could be expanded to the entire State.

A study of 20 vehicles that had failed a smoke opacity test of the type used in the New York program indicated that after repair, both smoke opacity and PM mass emissions had fallen by over 30% on average. The most common repairs performed in the study were replacement of injectors and injector pumps. Repair costs ranged from $85 to $2,053 per vehicle, with an average cost of $1,088 (McCormick et al. 2003).

As noted, New York State already has a diesel inspection and maintenance program, but it could be strengthened to yield additional PM reductions. It could be extended statewide rather than applying only to trucks registered in counties designated as nonattainment. Additionally, the maximum acceptable opacity limits could be lowered.[35] While the current limits[36] are consistent with those used in other heavy-duty diesel inspection and maintenance programs, they are fairly lenient.

As noted in Appendix D, California has mandated on-board diagnostic (OBD) systems on new diesel vehicles beginning in the 2010 model year. New York's adoption of these regulations would eventually allow the use of on-board diagnostic equipment to identify high emitters in a diesel inspection and maintenance (I&M) program, as in many current light duty I&M programs. However, such a program would be unlikely to have a significant impact prior to 2020.

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[35] Although an extended or more stringent Inspection and Maintenance (I&M) program might yield additional reductions in PM emissions, it is unlikely that New York would be successful in arguing for credit from EPA under its State Implementation Plan (SIP). EPA's current MOBILE6.2 emissions inventory modeling software does not include any assumptions about high-emitting diesel vehicles, and does not have the ability to quantify the benefits of a diesel I&M program. In addition, EPA has not issued guidance as to how a state might otherwise quantify the emissions reductions resulting from a diesel I&M program for purposes of developing a SIP.

[36] Maximum opacity: 70% for 1973 and older vehicles; 55% for 1974-1990 vehicles; 40% for 1991 and newer vehicles.

Idle Reduction. In order to reduce or eliminate unnecessary idling, approximately 26 states and cities, including both New York State and New York City, have enacted laws that restrict the idling of diesel vehicles (ATRI 2004). New York State law sets an idling limit of five minutes, while the New York City ordinance is more restrictive, limiting idling to no more than three minutes (NYSR 217-3; NYC AC 24-163).

New York could encourage or mandate the use of idle reduction technologies in the same way that it encourages or mandates vehicle retirement, engine repowering, retrofit controls, and fuel switching. Despite relatively short payback periods, a vehicle owner's shortage of capital funds could impede adoption of these on-truck technologies. The development of a revolving loan fund could spur technology adoption and may be more appropriate than a grant program. The EPA has issued guidance as to how a state can take credit for the benefits of an idle reduction program (EPA 2004a). EPA's SmartWay program has had some success in encouraging the use of small business loans for purchase of technologies that improve fuel economy, such as aerodynamic drag reduction devices. More expensive approaches that reduce idling - such as APUs - have not been successful. This suggests that for low-cost idle reduction technologies, such as cabin heaters, a revolving loan fund might work but that higher cost options may not be successful.

New York was one of the first states to install the necessary equipment to provide electricity from the grid to power truck accessory systems at truck stops within the state. Four New York sites can now provide on-site power to trucks with sleeper cabs, including one each on the westbound and eastbound sides of the New York State Thruway east of Syracuse, one at the Hunt's Point Market in the Bronx, and one on the Adirondack Northway (I-87), a major truck route from Albany, NY, to Montreal, Quebec (NYSERDA 2004). More of these installations will provide greater opportunities for individual truckers to reduce their idling.

Reducing Fuel Usage. There are a number of changes that do not involve engine technology that will reduce fuel use from onroad heavy-duty diesel vehicles, particularly Class 8 tractor-trailer trucks. These include the maintenance of proper tire inflation and the reduction of highway speeds. New York could reduce allowable highway speeds within the state, or better enforce existing limits. The State could also mandate and enforce proper tire inflation for highway trucks, in conjunction with current enforcement of highway truck weight limits.

Other technologies that can be applied to highway trucks to reduce fuel use include low-rolling-resistance tires and aerodynamic designs for both tractors and the trailers they pull. The EPA has developed a voluntary partnership program with the transportation industry, called the SmartWay Transport Partnership, to focus attention on these technologies and their benefits (EPA 2005e). Under the SmartWay partnership, the EPA is working with partners to package some of these fuel-saving technologies along with emissions reduction retrofit devices (DOC or DPF) into "SmartWay Upgrade Kits." The EPA estimates that a kit combining an idle reduction device, low-rolling-resistance tires, aerodynamic treatments for the trailer, and a DPF can increase fuel economy by 20% while reducing PM emissions by up to 90%. Based on fuel savings, net monthly cost savings (i.e., cost of fuel saved minus the loan cost) for the truck operator would be over $100 per truck.

Ten states in the U.S. have state infrastructure banks (SIBs) that provide loans for road building and other projects. The funds are typically controlled by state departments of transportation. New York is one of the states that has a SIB. Oregon officials recently announced that retrofits with SmartWay upgrade kits would be eligible for Oregon SIB loans. A similar approach to funding the retrofit of heavy-duty vehicles in New York could be explored. Because Class 8 trucks emit approximately 40% of all heavy-duty onroad $PM_{2.5}$ in New York, it is important to develop financing mechanisms aimed at reducing emissions from this sector.

<u>Recommendations</u>

Inventory data for heavyduty onroad truck $PM_{2.5}$ emissions in New York indicate that 40% of $PM_{2.5}$ emissions are from Class 8 trucks, 25% from transit buses, 15% from school buses, and 6% from Class 6 and 7 trucks. This does not account for the fact that nearly three-quarters of all New York transit buses have been or will be retrofitted with diesel particulate filters. In light of the substantial particulate contributions from this sector, the following viable strategies are recommended for reducing carbonaceous $PM_{2.5}$ and exposure to the same in New York State:

- Adopt the California regulations for truck refrigeration units (TRUs), idling reduction, transit bus, and trash trucks that require using clean fuels and retrofit devices. Together these could provide significant $PM_{2.5}$ reductions.
- Fund retrofit of 1990s and newer model year school buses and replacement of older school buses to reduce emissions from and exposure to school bus $PM_{2.5}$ emissions. In the NAA, approximately 500 tons of carbonaceous $PM_{2.5}$ could be reduced each year if all school buses were retrofitted with particulate filters.
- Evaluate travel patterns of in-state versus out-of-state trucks, to assist in the development of policies to reduce heavy-duty truck $PM_{2.5}$ emissions. Retrofiting class 8 trucks with particulate filters could reduce up to 1,000 tons of carbonaceous $PM_{2.5}$ in the NAA each year.
- Develop incentives for the use of idle reduction technologies at trucks stops - given that currently available technologies such as truck stop electrification and cabin heating provide PM reductions at a cost savings to the truck operator.
- Reduce emissions from trucks and buses that carry passengers and operate in densely populated areas - since these vehicles are a concern from an exposure standpoint.
- Since organic $PM_{2.5}$ and secondary $PM_{2.5}$ formation from VOCs contribute to overall $PM_{2.5}$, ensure that particulate filters are catalyzed.

NONROAD ENGINES

This section discusses the current and projected inventory of $PM_{2.5}$ emissions from nonroad sources in New York. It reviews available regulatory authority to address these emissions, and the control strategies that State and local authorities could consider. This section also provides information on regulatory authority and emissions standards that will be referred to in the later marine ports section.

Sector Profile

The major source of organic and elemental carbon from nonroad sources comes from large diesel equipment. This includes agricultural tractors and combines, airport ground-support equipment, construction and mining equipment (e.g., loaders, tractors, pavers, generators, compressors), commercial and industrial equipment (e.g., generators, compressors, refrigeration units), logging equipment, rail locomotives, marine ships, cargo handling equipment, and aircraft. Gasoline-powered heavy-duty nonroad equipment accounts for only 2.5% of direct $PM_{2.5}$ emissions from the nonroad sector in New York State. $PM_{2.5}$ emissions from tire and brake wear in nonroad equipment are also insignificant. The last two categories, airport and port-related sources, are discussed in later sections of this report. This equipment includes engines in a wide variety of sizes and configurations, from as small as 20 horsepower to as large as 5,000 horsepower or more. While agricultural equipment is concentrated in rural areas, other nonroad diesel engines contribute significantly to urban air quality concerns. Table 3-22 provides detailed information on the U.S. population of nonroad equipment:

Table 3-22. Estimate of U.S. Population of Nonroad Equipment by Equipment Type.

Source:U.S. EPA, *Control of Emissions of Air Pollution from Nonroad Diesel Engines and Fuel* (2004)

Equipment Type	Population (x1000)
Diesel	
Agricultural	3,208
Airport Ground Support Equipment	22
Commercial/Industrial	10,456
Construction/Mining	3,017
Logging Equipment	330
Gasoline	
Lawn & Garden	108,460
Recreational	25,488

Non-road engines were not subject to emission standards until 1996, when the first (Tier 1) standards took effect. The Tier 1 PM standard for non-road compression-ignition engines was established at 0.4 grams per brake-horsepower-hour (g/b-HP-hr) and applied to engines with horsepower ratings at or above 175 HP.

Tier 2 PM standards for the same class of non-road engines were established at 0.15 g/b-HP-hr, effective with the 2003 model year. At the same time that Tier 2 PM standards were established, EPA established the 2006 model year date for Tier 3 standards but subsequently decided to keep the PM emission standard at the same level. Tier 4 PM standards for non-road engines between 75 and 750 HP have been established at 0.01 g/b-HP-hr and will be phased in from 2011 to 2013. Consequently over time, emissions from non-road engines will be considerably cleaner.

According to the EPA's 2002 National Emissions Inventory, the annual direct $PM_{2.5}$ emissions from nonroad equipment in New York totaled almost 12,000 tons – twice the amount from onroad vehicles (including both heavy-duty trucks and light-duty cars). This is approximately 8% of the total direct $PM_{2.5}$ inventory, including stationary sources. Within the New York City Metropolitan Nonattainment Area (NYC NAA), nonroad emissions are even more important – totaling over 9,000 annual tons and accounting for over 19% of all annual direct $PM_{2.5}$ emissions. The EPA data for direct $PM_{2.5}$ emissions for major categories of nonroad equipment in New York are shown in Table 3-23.

Table 3-23. 2002 Nonroad Emissions Inventory.

Source U.S. EPA, *Final 2002 National Emissions Inventory* (2006)

	Direct $PM_{2.5}$ Emissions (tons/year)						
			Other Urban Areas		Typical Rural Counties		
Source	**New York State**	**NYC NAA**	**Erie County**	**Monroe County**	**Greene County**	**Lewis County**	**Cattaraugus County**
Construction & Mining	3,581	4,030	165	144	8	10	30
Recreation (Gasoline)	2,377	1,036	57	204	11	18	8
Commercial Marine	1,791	273	27	2	3	0	0
Lawn/garden (Gasoline)	998	1,377	62	72	1	1	3
Agricultural	714	42	15	14	3	16	15
Locomotives	350	151	34	19	4	0	2
Other	2,154	2,554	115	105	3	4	8
Total	11,965	9,463	475	560	33	49	66

Sector Emissions

Construction Equipment. The single largest nonroad diesel source is construction equipment, which accounts for over 20% of mobile source $PM_{2.5}$ emissions statewide and over 28% of mobile source $PM_{2.5}$ emissions in the New York City metropolitan nonattainment area (NYC NAA). Annual $PM_{2.5}$ emissions from construction equipment totaled over 4,000 tons within the nonattainment area in 2002—almost as much as the $PM_{2.5}$ emissions from all onroad vehicles (including gasoline cars and diesel trucks and buses).

On a mass basis, $PM_{2.5}$ from nonroad diesel equipment is composed of approximately 74% elemental carbon and 23% organic carbon (EPA 2005g). Depending on engine technology level, fuel, duty cycle, and the state of engine tuning, there may be slightly more or less organic carbon (EPA 2004h). Older nonroad equipment with two-stroke engines, and poorly maintained newer engines, are likely to emit PM with a higher organic carbon content due to the combustion of greater amounts of lube oil in the engine. Most of the remainder of the $PM_{2.5}$ mass (approximately 3%) is sulfates. Diesel PM from nonroad equipment is likely to have slightly higher sulfate content than PM from onroad diesel vehicles due to the higher sulfur content of current nonroad fuel.

Road building and other infrastructure projects are particularly intensive in their use of large diesel equipment. Projects let by the New York Department of Transportation statewide in 2005 reveal that while over 40% of the total value was for projects within the nonattainment area of New York City, Long Island, and the lower Hudson Valley, there was also significant road construction activity in other parts of the state, most notably in the middle of the state in the county that includes Syracuse. NYSERDA is currently working on an inventory of construction equipment emissions in New York.

Figure 3-9. Expected Change in Construction Equipment $PM_{2.5}$ Emissions Through 2020

Source: EPA NONROAD Model (http://www.epa.gov/otaq/nonrdmdl.htm)

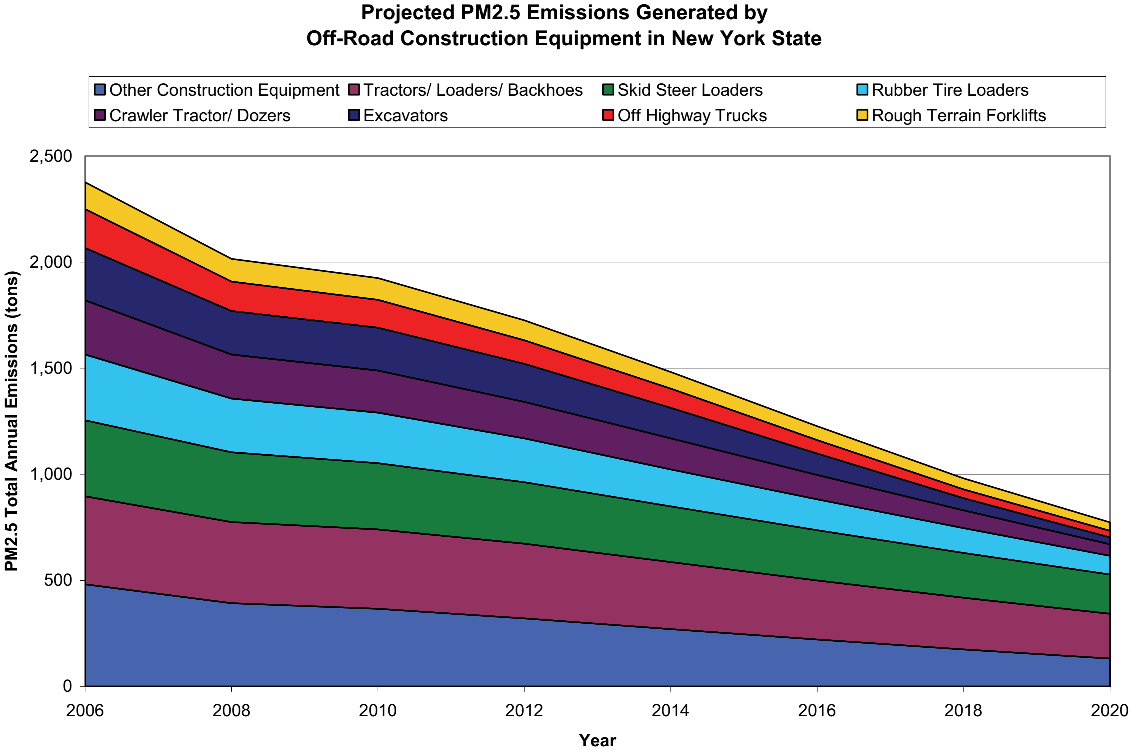

Figure 3-9 shows the projected amount of fine particulate emissions from different types of construction equipment in New York State between 2006 and 2020. The graph shows that emissions from seven types of vehicles account for almost 80% of $PM_{2.5}$ emissions from construction equipment in New York State: excavators, off-highway trucks, rough-terrain forklifts, rubber tire loaders, tractors/loaders/backhoes, dozers, and skid steer loaders.

As further shown in Figure 3-9, these equipment types are expected to continue to produce the majority of $PM_{2.5}$ emissions from this sector through 2020, despite overall reductions based on fleet turnover. Three categories of equipment in 2020 (rubber tire loaders, tractors/loaders/backhoes, and skid steer loaders) will emit about 60% of total construction equipment emissions.

According to the EPA's nonroad emissions model, approximately one-third of this equipment in most categories is new enough that its engines were built to comply with the latest Tier 2 or Tier 3 emissions standards, with the majority of the rest built to comply with Tier 1 standards. With the exception of tractors/loaders/backhoes and skid steer loaders, a relatively small percentage of equipment is old enough that its engines are completely uncontrolled (see Table 3-24).

Table 3-24. Age Distribution of Construction Equipment.

Source: EPA NONROAD Model (http://www.epa.gov/otaq/nonrdmdl.htm)

	Year of Manufacture			
Equipment Type	**Later than 2003 (Tier 2/Tier 3)**	**2001-2003 (Tier 1)**	**1996 – 2000 (Tier 1 Phase-in)**	**Earlier than 1996 (Uncontrolled)**
Excavators	43%	33%	22%	3%
Rubber Tire Loaders	29%	25%	32%	14%
Tractors/Loaders/Backhoes	19%	17%	25%	39%
Crawler Tractor/Dozer	33%	29%	30%	8%
Skid Steer Loaders	18%	17%	24%	41%
Other	29%	26%	29%	16%

The slow turnover rate of nonroad diesel equipment means that even the more stringent rules will not begin to have a major impact for many years. Based on normal fleet turnover, the EPA estimates that annual PM emissions from nonroad diesel engines will fall by 20% through 2010 and by 56% through 2020, or less than a third of the percent reduction expected from the onroad fleet in the short term (EPA 2000b, 2004h). Because particulate filter-based standards will not go into effect until 2010 and later for nonroad diesel engines, efforts to retrofit engines manufactured prior to 2010 will yield significant PM reductions.

Nonroad Gasoline Engines. Nonroad gasoline engines include lawn and garden equipment and recreational equipment such as all-terrain vehicles and marine gasoline engines. Table 3-23 shows that recreational gasoline engines emitted 2,377 tons of $PM_{2.5}$ in New York State in 2002, second only to emissions from construction equipment. Lawn and garden equipment $PM_{2.5}$ emissions were the fourth largest source of nonroad engine $PM_{2.5}$ emissions in 2002. Together these engines contributed 30% of nonroad engine $PM_{2.5}$ emissions in New York State in 2002. Emissions from small gasoline nonroad engines often occur in close proximity to the equipment operator. For example, some lawn and garden equipment—such as weed trimmers—are suspended by a strap at the operator's shoulder. Thus the engine and exhaust emissions occur within about two feet of the operator's breathing zone. Chain saw engines emit pollution at arm's length from the operator. The California Air Resources Board estimates that operating a chain saw for 2 hours is equivalent (in HC emissions) to driving ten 1995 vehicles 250 miles each.

Locomotive Emissions. There are 36 freight rail companies operating in New York State, with almost half of the freight track operated by CSX Transportation and Norfolk Southern Corporation, the two U.S. Class 1 railroads operating in the state. (A Class 1 railroad is defined by the Surface Transportation Board as one with at least $272 million in operating revenue in 2002). The remaining track is operated by two Canadian railroads, four regional railroads, 21 local railroads, and seven switching and terminal companies (DOT-BTS 2004). In addition to the freight rail track, New York State has over 2,000 miles of commuter rail track (DOT-BTS 2006), the vast majority of which is within New York City and the surrounding counties and is operated by the Metropolitan Transportation Authority. New Jersey Transit also operates commuter rail in New York City and its environs, and AMTRAK operates train service along the Northeast Corridor through New York City.

The vast majority of commuter locomotives operating within the New York City Metropolitan Nonattainment Area are electrified, including New York City Transit subways, Long Island Railroad commuter trains, Metro North commuter trains, and AMTRAK trains. (NYC Transit operates a small number of diesel work trains, but all passenger cars are electric.) The locomotives that pull New Jersey Transit commuter trains through northern New Jersey and into Manhattan are diesel-powered, as are some of the Long Island Railroad locomotives that operate on the eastern end of Long Island in Suffolk County.

Locomotive emissions account for only 2% of annual mobile source $PM_{2.5}$ emissions within New York State, and just over 1% in the New York City Metropolitan Nonattainment Area. Their contribution is higher in other parts of the state, including Erie County (Buffalo), where locomotives account for almost 4% of mobile source $PM_{2.5}$ emissions, and rural Greene County, where they account for over 5%. However, reducing $PM_{2.5}$ emissions from locomotives may provide an important public health benefit in terms of reducing exposure. This is because diesel locomotives operate in highly congested urban areas: all of New Jersey Transit's commuter rail lines terminate either at Manhattan's Pennsylvania Station, or in Hoboken, NJ, just across the Hudson River from lower Manhattan. A total of 111diesel-powered New Jersey Transit trains go into and out of Manhattan on a typical weekday (NJT 2006). In addition, $PM_{2.5}$ exposures at train stations where diesel locomotives operate can be very high. Ambient monitoring was conducted by NESCAUM on a commuter rail station platform in Boston, frequented by diesel locomotives. The mean

$PM_{2.5}$ level monitored on the platform over a 3-hour period during rush hour was 546 μ/m^3. Peak levels were even higher (NESCAUM, 2004). These measurements indicate that large numbers of commuters riding locomotives (as well as other passenger vehicles such as ferries and transit buses) can be exposed to elevated levels of $PM_{2.5}$ during their commutes and for short periods while waiting for trains or ferries.

Federal and State Authority

EPA has adopted regulations that prohibit state and local governments from enforcing any rules that would "affect a locomotive manufacturer's or remanufacturer's design." EPA regulations specifically prohibit three categories of state and local controls for any locomotive, for eight years after purchase or remanufacture (equivalent to 1.33 times the "average" useful life of six years before remanufacturing). These categories include 1) emissions standards, 2) non-federal in-use testing programs, and 3) mandatory emissions control retrofit requirements (EPA 1997a). As such, New York State would seem to be precluded from imposing *mandatory* retrofit requirements on locomotives used in the state.

In 2003, the New York City Council adopted Local Law 77 (NYC AC 24-163.3), which mandates the use of ultra-low-sulfur diesel fuel and requires retrofits for all diesel-powered construction equipment greater than 50 horsepower used on all publicly funded construction projects in New York City. The law also requires ultra-low-sulfur diesel and retrofits on publicly owned construction equipment. The requirements were phased in between June 2004 and December 2005, depending on the location and size of the project (NYCDDC 2004). Other New York jurisdictions, including Westchester County, have begun to pursue similar laws. A statewide version of this regulation was signed into law on August 16, 2006, by Gov. Pataki. The new law (The Diesel Emission Reduction Act of 2006) requires that by January of 2011, any diesel-powered heavy-duty vehicle or equipment owned by, operated by or on behalf of, or leased by a State agency or State and regional public authority shall be powered by ultra-low-sulfur diesel fuel and be equipped with best available control technology verified by the EPA to reduce PM and NO_X emissions. New York City also requires that vehicles transporting solid waste material (both on- and nonroad) be retrofitted with emissions control devices (Local Law 40).

Other applicable regulations include: New York City's NYC AC 24-141:142, which prohibits the emission of an air contaminant with greater than a certain smoke density; it applies to all sources, including nonroad equipment. The City of Buffalo has limits on emissions of "dense smoke" from sources that include nonroad equipment (BUF CC 66-3; 66-25). It specifically mandates that the boilers used "in connection with steam shovels, locomotive cranes, or used for stationary work" use only "anthracite coal, coke, gas, or other smokeless fuel" (BUF 66-16). Rochester prohibits the emissions of smoke greater than a specified density from any "locomotive, tug, boat…" and prohibits emissions of "unlawful density" from locomotives operating in the city (RC 100-3, 100-5). Suffolk County also has opacity limits. With some exceptions, the Laws of Rockland County require the County to purchase diesel vehicles that use the best available retrofit technology (BART) or that meet equally stringent PM standards, and require such vehicles to use ultra-low-sulfur diesel fuel (no more than 15 ppm). The County is subject to a purchasing schedule that would require all county-owned diesel vehicles to be low-emitting by 2012 (LRC, chap. 137).

Approximately 10 to 15 percent of construction activity in New York State is subject to Local Law 77. (Assuming 25% of construction spending statewide is in New York City and 60% of that is for infrastructure projects. See Sector Profile above).With the signing of New York State's Diesel Reduction Act of 2006, approximately 60% of all construction projects in New York State will be subject to a retrofit requirement. This assumes that all infrastructure projects are paid for by State or local agencies. Residential and commercial buildings, which make up the remainder of projects, will not be affected by the laws. Because some contractors that work on infrastructure construction projects may also work on residential or commercial projects, the law may reduce emissions from private projects as well. One of the first projects to occur under the Local Law 77 requirement is the expansion of the Croton Reservoir access for New York City. Eighteen particulate filters have been installed to date in equipment working on the site. A total of 35 to 40 pieces of equipment will be retrofitted with best available control technology as part of the project.

Technical Options for Emissions Reduction

Locomotives. The retrofit and alternative fuel technologies discussed in Appendix B that are applicable to onroad diesel trucks and diesel construction equipment are also potentially applicable to locomotive engines, although they are not as well developed commercially for the locomotive application. An active DPF system that has been commercialized in Europe has seen limited demonstrations in California,[37] and DOCs from several companies are under testing or have seen limited demonstrations in California and in Boston, MA. Further commercialization of these technologies would benefit from additional funding for proof-of-concept demonstrations.

During remanufacture or rebuilding of a locomotive, the diesel engine can be upgraded with new technology to reduce PM emissions. Some relevant engine technologies already available include improved pistons, cylinder liners, and piston rings; these reduce lube oil consumption and therefore reduce PM emissions. Under current EPA regulations, all new and remanufactured locomotives must meet PM emission standards when new or rebuilt, as indicated in Table 3-25. One of the major beneficial effects of this action by EPA is the upgrading of emission controls on older locomotives at the time of remanufacture.

Another important approach for reducing locomotive emissions is idle reduction, particularly for switchyard locomotives. Argonne National Laboratory estimates that a typical switchyard locomotive idles

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[37] In Europe, this device has been commercialized for four-stroke locomotive engines. Many U.S. locomotive engines are two-stroke engines, which pose additional technical challenges.

up to 75% of the time, and that idling consumes 27% of the total fuel used (Argonne 2005). In some cases, these locomotives are allowed to idle for long periods of time to maintain engine oil and coolant temperatures; in other cases, they are left to idle to provide a relatively small amount of electrical power for auxiliary equipment. To a lesser extent, commuter locomotives, and even line haul locomotives, are sometimes left to idle for extended periods in switchyards or at stations for the same reasons.

Table 3-25. PM Emission Standards (g/b-hp-hr) for Locomotives.
Source: EPA, 63 FR 18978, April 16, 1998

Tier & Duty Cycle	PM Emission Std	Model Yr (New)	Model Yr (Re-mfg)
Tier 0 (Line-Haul)	0.60	2001	1973-2001
Tier 0 (Switch)	0.72	2001	1973-2001
Tier 1 (Line-Haul)	0.45	2002-2004	2002-2004
Tier 1 (Switch)	0.54	2002-2004	2002-2004
Tier 2 (Line-Haul)	0.20	2005+	2005+
Tier 2 (Switch)	0.24	2005+	2005+

As discussed in Appendix B, a number of alternative systems can provide the necessary electrical power for auxiliary loads on locomotives (auxiliary power units). Systems also exist that monitor various engine parameters and automatically shut down the engine when idling is unnecessary and restart the engine as required (shutdown/restart systems). In addition, at least one company makes a hybrid-electric switcher locomotive that uses a small diesel engine and a large battery pack, thus significantly reducing both emissions and fuel use. The EPA's review of currently available devices indicates that these idle reduction systems cost between $4,000 and $35,000 to install on a locomotive, depending on the type of device used. Their use can reduce annual PM emissions by up to 90 pounds and annual fuel use by up to 7,700 gallons from a single locomotive. Given the amount of fuel that can be saved by reducing or eliminating idling, many of these systems have payback periods of six to 20 months (Gaines 2005; also see Appendix C). Comparisons of emission control strategies for locomotives are shown in Table 3-26.

In limited cases, it may also be cost-effective to modify existing engines to incorporate more modern design elements and equipment—such as improved fuel injectors—that result in lower emissions. This is best done during a major overhaul, which generally occurs several times during the life of a typical nonroad engine. This may be particularly effective for locomotive engines, for which Tier 1 upgrade kits are available for older Tier 0 engines.

Table 3-26. PM2.5 Emissions Control Strategies for Locomotives.

Source: See Appendices B &C

Equipment Type	Control Strategy	Annual Reductions (lbs/vehicle)	Cost/ton
Locomotives	DOC retrofit	Up to 500	\$15,000 - \$125,000
	DPF retrofit	Up to 1,500	\$35,000 - \$100,000
	Idle Reduction: APU on switcher	90	\$20,000 annual savings (2 year payback period)
	Idle reduction: hybrid switcher	110	\$25,000 annual savings (8 year payback period)

Given that most control technologies for locomotives (with the exception of particulate filters) will reduce only a fraction of $PM_{2.5}$ emissions, levels of $PM_{2.5}$ in passenger cabins and on station platforms could remain very high. Because of this, alternative approaches to reducing in-cabin exposures should be explored. These include cabin sealing or filtration of intake air into cabins. Rerouting of exhaust pipes may also provide some benefit—in the form of $PM_{2.5}$ reductions—to passengers.

There may be targeted opportunities to use natural gas as fuel in specific types of equipment. The West Coast Collaborative and South Coast Air Quality Management District in California have provided \$2.2 million to repower an existing locomotive with a natural gas engine and a liquefied natural gas fuel system (WCC 2005b). Potential applications of electric power to replace diesel fuel in nonroad engines include hybrid-electric locomotive yard switchers.

Nonroad Construction. Given the higher baseline PM emissions for nonroad equipment, retrofit technologies are generally more cost-effective than for trucks and buses (see Appendix C). However, retrofit technologies may more easily be applied to onroad engines, primarily because a greater number of retrofit technologies have been verified for use in highway engines. Also, the generally less complex configuration of the truck chassis/engine combination, fewer muffler configurations in trucks, and less variation in PM emissions make retrofits easier in trucks as compared to nonroad engines. Some new nonroad engines are both cleaner and more fuel efficient than older engines, so repowering to reduce emissions can provide additional emissions benefits from fuel savings. A detailed discussion of technical options for nonroad engine emissions reductions is presented in Appendix B. These options are similar to the ones for highway diesel engines.

Policy Options

Adopting California Standards. California can create its own requirements for some types of existing nonroad vehicles, but other states cannot. With respect to nonroad equipment, New York and the other states can only adopt rules that California has imposed on existing nonroad vehicles. In 2004, CARB adopted mandatory retrofit requirements for the portable diesel engines used in California to power agricultural pumps, airport ground-support equipment, oil drilling rigs, portable generators, and some other nonroad equipment. The rule requires phased reductions in PM emissions from these engines to achieve a 95% reduction by 2020 (CARB 2005). These are the first mandatory retrofit requirements California has adopted for any type of nonroad diesel engine, although CARB is evaluating a similar program for other nonroad construction equipment. New York State could adopt these current and future California rules. New York State is, along with other states, preempted from adopting California's emission standards for lawn and garden equipment. In 2003, a rider on a Veterans Administration/Housing and Urban Development (VA HUD) appropriations bill inserted by Sen. Kit Bond (R-MO) altered state authority under the Clean Air Act. Prior to this amendment, any state outside of California could adopt California's nonroad engine standards for engines smaller than 50 horsepower. The amendment removed this authority. As part of the final language in the VA HUD bill, the EPA was tasked with issuing emissions standards for lawn and garden equipment. A proposed regulation has not yet been promulgated.

Nonroad Construction Equipment. A New York State voluntary retrofit program for construction equipment would likely be most effective if targeted toward one or more of the seven types of construction vehicles that currently account for almost 80% of $PM_{2.5}$ emissions from construction equipment in the state. These include excavators, off-highway trucks, rough terrain forklifts, rubber tire loaders, tractors/backhoes, bulldozers, and skid steer loaders. Such a program need not target the oldest pieces of equipment. Equipment with Tier 1 or Tier 2/3 engines would benefit from all of the retrofit and alternative fuel options discussed in Appendix B. For example, a DOC retrofit or use of an alternative fuel could reduce annual PM emissions by up to 20 pounds per vehicle; a DPF retrofit could reduce annual PM emissions by up to 65 pounds per vehicle. As shown in Table 3-24, over 70% of construction equipment in New York State is assumed to be new enough to be equipped with Tier 1 or Tier 2/3 engines.

Two equipment types—tractors/backhoes and skid steer loaders—are assumed to have a much higher percentage of older vehicles with uncontrolled engines than the other types. For these equipment types, it would likely be relatively more effective to provide incentives under a voluntary program for early vehicle retirement rather than for retrofits or switching fuel.

Given the diversity of nonroad equipment designs and applications, evaluating machines on a piece-by-piece basis is still necessary for DPF installation. Experience has shown that rear-engine machines, such as wheeled loaders, are easier to retrofit with particulate filters, since installing a filter on the back of a piece of equipment does not impede the operators' line of sight. Skid steer loaders are difficult to retrofit because of low exhaust temperatures, space constraints, operator line-of-sight issues, and other considerations. Fuel changes or engine upgrades may be the most practical approach for skid steer loaders. After 2010, when all construction equipment will be required to use fuel with very low sulfur levels, there will be fewer impediments to DPF retrofits.

> **Demonstration grants.** The same grant programs used to fund replacement or retrofit of onroad vehicles can be used to fund nonroad projects. The EPA has made demonstration grants for nonroad projects under its National Clean Diesel Campaign, most recently funding retrofits of commuter rail locomotives, construction vehicles, agricultural equipment, rubber-tired gantry cranes, and port straddle carriers (EPA 2005b). New York State could create a similar voluntary retrofit grant program specifically for construction equipment. Such a program could potentially be funded with any remaining New York State Environmental Bond Act funds, or with funding from a different source.
>
> **Low sulfur fuel.** At approximately 3,000 ppm, the current sulfur content of nonroad diesel fuel may impede the use of effective retrofit PM emissions controls on nonroad engines. Sulfur adversely affects the catalytic processes used to reduce PM emissions. DOCs do not work well if fuel sulfur is greater than 500 ppm, and the even more effective DPFs require fuel with no more than 50 ppm sulfur (see Appendix B).
>
> While EPA regulations will eventually require nonroad diesel fuel sulfur reductions, the implementation dates lag those for onroad fuel. Beginning in June 2007 when nonroad diesel will be required to contain no more than 500 ppm sulfur, DOC retrofits will become practical on virtually all construction equipment. Prior to 2010 when nonroad diesel fuel sulfur levels will be reduced below 50 ppm, either voluntary or mandatory use of onroad diesel fuel will be required in order to implement DPF retrofits on construction equipment. For this reason, New York City's Local Law 77 includes a mandate for the early adoption of reduced-sulfur fuels.

Several mandatory programs that California has adopted could be adopted by New York State under Section 209 of the Clean Air Act Amendments. Note that broad mandatory retrofit requirements are harder to impose on privately owned nonroad equipment than on privately owned onroad vehicles. Unlike onroad vehicles that use public roadways, nonroad equipment is usually not required to be registered with the State. One of the reasons that California was able to impose mandatory retrofit requirements on portable engines is that in California these engines are required to be registered.

Locomotives. The fact that idle-reduction technologies provide annual fuel and cost savings make them amenable to the use of a revolving loan fund to spur technology adoption. Such a fund typically provides low-interest loans for the purchase and installation of qualifying emissions reduction technologies. As the original recipients repay their loans, the funds are used to make additional loans. In 2005, the West Coast Diesel Collaborative developed such a fund to finance idle-reduction technologies on locomotives. This fund was established with $150,000 from the EPA and $450,000 from the nonprofit foundation Climate Trust (WCC 2005a).

In 2005, the California Air Resources Board entered into a voluntary agreement with the BNSF Railway Company and the Union Pacific Railroad Company to reduce PM emissions in California rail yards. Among other things, the agreement requires the railroads to equip all locomotives with "automatic idling reduction devices" by June 2008. While voluntary, this agreement includes monetary penalties for failure to implement its various requirements (CARB 2005e). New York could take a similar approach. As discussed above, almost half of the freight track in New York is operated by CSX Transportation and Norfolk Southern Corporation, as are 20 major switchyards. This concentration of ownership limits the number of signatories required for agreement. The EPA has issued guidance on how a state can take credit for the benefits of a locomotive idle-reduction program that would result from such an agreement (EPA 2004f).

Lawn and Garden Equipment. California has proposed stringent emissions standards for lawn and garden equipment. When implemented, they will require catalytic converters and improved air fuel control; they will reduce HC and NOx emissions from weed trimmers and chain saws by 74% and lawn mowers 67% beginning in 2010. No estimate for reductions in PM2.5 emissions are provided in the California regulatory documents, but the engine changes required to reduce HC and NOx will undoubtedly reduce PM emissions. In addition, secondary PM formation will be greatly reduced as a result of this rule. Given the recently imposed restriction on state authority to adopt California emission standards for this category of nonroad engines, a stringent emissions standard at the federal level is critical to states in order to reduce PM and other emissions from this sector.

Recreational Engines. Offroad motorcycles, all-terrain vehicles, recreational marine engines (gasoline and diesel), and specialty vehicles such as go-carts are included in this category. For some categories in this class, emissions remain unregulated by the EPA (stern-drive and inboard gasoline marine engines, for example). For many nonroad gasoline engines, the installation of catalysts similar to those used in automobiles is technically feasible. Efforts are needed at the federal level to introduce catalyst-based standards for nonroad gasoline engines where they are not currently required. CARB has established standards for recreational marine engines that will reduce emissions by 66% beginning in 2007. These standards could be adopted by New York.

Recommendations

Taken together, $PM_{2.5}$ emissions from nonroad engines are the greatest source of mobile source-related $PM_{2.5}$ in New York State. Construction equipment emits the greatest amount of $PM_{2.5}$ in the State. Front-end loaders, backhoes, and skid steer loaders will produce the majority of construction equipment-related $PM_{2.5}$ emissions between 2006 and 2020 in New York State. Reducing construction-related $PM_{2.5}$ emissions will result in the greatest reduction of $PM_{2.5}$ mass from nonroad diesel engines in New York. However, retrofit programs still need to be evaluated on a case-by-case basis. $PM_{2.5}$ exposures resulting from locomotives and small nonroad gasoline engines can also be substantial. The following specific actions are recommended:

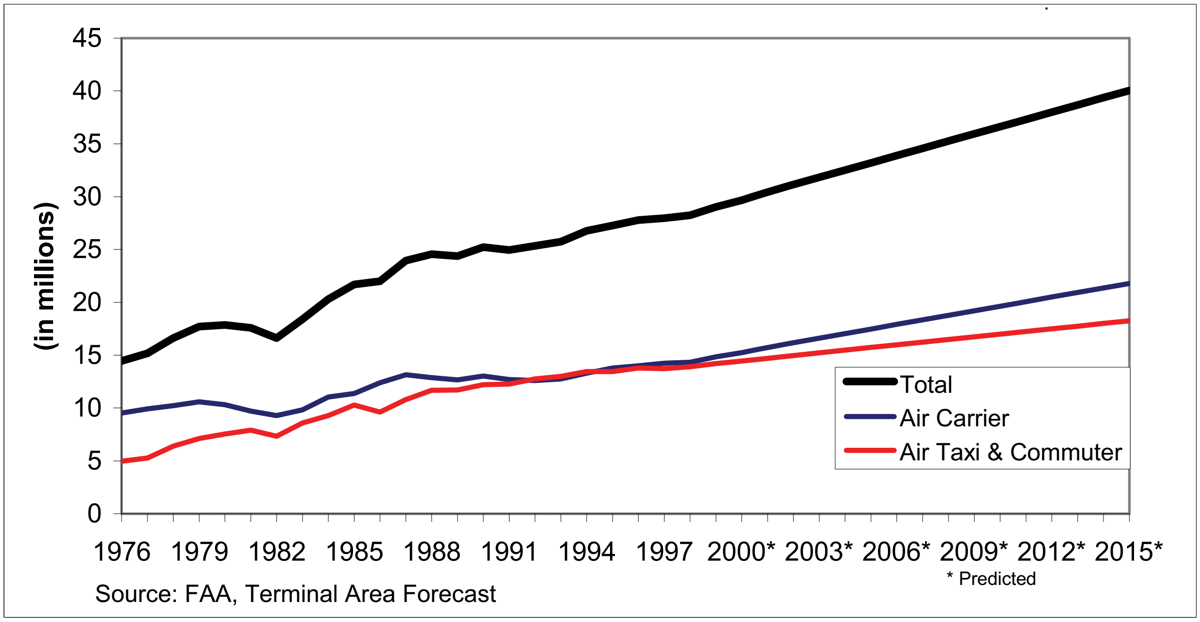
- Develop incentives for installing particulate filters on construction equipment. Particulate filters can nearly eliminate EC and OC emissions from nonroad diesel equipment but not all equipment is suited for filter installation. Rear engine machines can more easily be retrofitted with particulate filters than front engine machines. For machines that cannot be retrofitted with particulate filters, retrofit with oxidation catalysts should be encouraged or required.
- Promote installation of idling reduction technologies in locomotives. This is one of the most cost-effective means to reduce $PM_{2.5}$ emissions from nonroad diesel engines. Two-year payback times are typical.
- Retrofit locomotives with particulate control devices such as oxidation catalysts and particulate filters. Controlling diesel commuter rail locomotive $PM_{2.5}$ pollution could significantly reduce commuter exposure at rail stations. Reducing freight locomotive emissions will reduce $PM_{2.5}$ exposure in and around rail yards in New York. Evaluate the use of air filtration systems for air intake to reduce on-board exposure in commuter rail cars.
- Encourage the EPA to promulgate a stringent regulation to reduce $PM_{2.5}$ emissions from lawn and garden equipment.
- Adopt California's emission standards for nonroad recreational engines. The State has already adopted the CARB jet ski rule and could expand the rule to include other types of marine recreational engines.

AIRPORTS AND AIRCRAFT

Nationally, the number of aircraft operations (defined as one takeoff or one landing) has grown from approximately 15 million in 1976 to almost 30 million in 2000, a cumulative growth of over 100% (NESCAUM 2003). While emissions from most mobile source sectors are declining due to the implementation of more stringent control programs, the growth in activity at airports combined with relatively lax aircraft engine emission standards are resulting in increased pollution from airports. As shown in Figure 3-10, aircraft activity is expected only to grow, resulting in this emissions category becoming even more significant in the future. Airport emission sources can be grouped into five general categories:

- Aircraft Engine Operations, which include taxiing, idling, takeoff, climb-out, and approach
- Ground Support Equipment & Ground Power Units, which include equipment used to service the aircraft, including units supplying electricity and air conditioning to parked aircraft, baggage tugs, and belt loaders
- Stationary Equipment, which includes emergency power generators and aircraft engine testing
- Ground Access Vehicles, which include trucks, personal cars, and shuttle buses operating on airport property, including those which travel on and off the property
- Construction Equipment, which includes heavy-duty nonroad equipment

Figure 3-10. Projected Growth in Aircraft Activity.



Source: FAA, Terminal Area Forecast

Sector Profile

There are 13 commercial service airports in New York State, most of which are in federally designated nonattainment areas. The New York Metropolitan Area (NYMA) has the two largest airports (John F. Kennedy [JFK] and LaGuardia) in terms of annual aircraft operations. While located in New Jersey, a 14th airport (Newark Liberty) is within the multi-state NYMA nonattainment area. These are ranked by level of activity in Table 3-27. Three of these airports (JFK, Newark, and LaGuardia) are among the largest in the United States. In 2004, JFK, Newark, and LaGuardia were ranked eighth, twelfth, and twentieth respectively in terms of aircraft boardings in the U.S.; they represented 85% of the total boardings for the 14 airports discussed in this chapter (BTS 2006). Appendix E includes profiles for each of the airports listed in the table below, along with information on emissions modeling for airports.

Sector Emissions

Comprehensive $PM_{2.5}$ inventories for major airports in the Northeast are not available. However, data from Los Angeles International Airport (LAX) may be comparable to major airports in New York State in terms of the relative contribution of PM emissions from the various emission source categories. According to the August 2005 Draft South Airfield Improvement Project Environmental Impact Report (EIR) for LAX (LAX 2005), the proportion of emissions from the above general categories (excluding construction emissions) are as follows: 27% from aircraft, 23% from , ground support equipment and ground access vehicles, 23% from stationary equipment, and 27% from ground access vehicles.

Table 3-27. Boardings at New York Airports (2004).
Source: Bureau of Transportation Statistics

State	Airport	Boardings
NJ	Newark Liberty	15,827,675
NY	John F. Kennedy	18,586,863
NY	LaGuardia	12,312,561
NY	Buffalo Niagara	2,206,385
NY	Albany	1,536,263
NY	Greater Rochester	1,364,869
NY	Syracuse Hancock	1,130,236
NY	Long Island MacArthur	986,103
NY	Westchester County	462,981
NY	Stewart Newburgh	250,006
NY	Greater Binghamton	133,894
NY	Elmira Corning	97,122
NY	Ithaca Tomkins	72,383
NY	Chautauqua Jamestown	11,276

Aircraft Engine Emissions. Beginning in 2004, the National Aeronautics and Space Administration (NASA) commenced its Aircraft Particle Emissions Experiments (APEX), assessing particulate emissions of black carbon, organic carbon, and sulfates in aircraft engine exhaust plumes downstream from engines operating on the ground at varying power settings (Wey 2006). APEX showed that black carbon particle counts were highest during idle and high thrust conditions and minimum at situations corresponding to cruise and approach operations. Black carbon particle mass was nonlinearly dependent on engine power. Typically it was less than 20 mg per kg of fuel consumed over the 4-7% power range but greater than 200 mg per kg of fuel consumed at 85% of thrust and above (i.e., equivalent to take-off and climb-out conditions). At the higher thrust levels ($\geq 85\%$) particulate emission mass was virtually all black carbon (i.e., virtually no organic carbon).

At lower thrust levels, BC mass and OC mass were roughly equivalent. Significant gas-to-particle conversion occurred in the exhaust plume as it cooled. Higher sulfur levels in the fuel caused the OC particle count to increase (i.e., sulfate particles functioned as condensation nuclei).

Ground Support Equipment (GSE). The airport GSE category is comprised of a large variety of vehicles and equipment that service aircraft during ground operations. The airlines own approximately 75% of GSE equipment, and the average age of the equipment is 9.4 years. Airline ownership of the GSE makes it problematic for an airport authority to establish a complete inventory of equipment at any one particular facility. Table 3-28 provides a comparison of the various airports in New York in terms of numbers of equipment and total emissions.

Table 3-28. $PM_{2.5}$ Emissions from GSE and Aircraft.[38]

Source: U.S. EPA-NEI 2006; Watz 2006; McGarry 2006

State	Airport	Total GSE Units (2002)	GSE PM (2002 – TPY)	Aircraft PM (2002 – TPY)
NJ	Newark Liberty	3730	74.3	41.3
NY	John F. Kennedy	892.4	58.4	19.40
NY	LaGuardia			
NY	Buffalo Niagara	69.5	3.0	6.81
NY	Albany	35.0	1.5	6.55
NY	Greater Rochester	46.6	2.0	6.31
NY	Syracuse Hancock	38.3	1.6	6.26
NY	Long Island MacArthur	48.9	2.1	10.20
NY	Westchester County	16.6	0.7	7.46
NY	Stewart Newburgh	10.3	0.4	5.31
NY	Greater Binghamton	2.2	0.1	2.75
NY	Elmira Corning	3.4	0.1	1.80
NY	Ithaca Tomkins	3.2	0.1	1.34
NY	Chautauqua Jamestown	5.4	0.2	0.62

Most GSE is powered by gasoline or diesel fuel; smaller percentages are powered by natural gas (CNG/LNG) and electricity (FAA Vision-100). A list of commonly used types of GSE and their function are described in Table 3-29.

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[38] The results from using national default methods (e.g., numbers of airport operations, aircraft NO_X emissions) to estimate GSE populations and PM emissions from aircraft and GSE can vary significantly from the results derived through a more specific study of an individual airport.

Table 3-29. Description of GSE Types and Function (NESCAUM 2003).

GSE Type	GSE Function	GSE Type	GSE Function
Aircraft Pushback Tractor	Used to push aircraft back from the terminal or to tow aircraft to and from the hangar.	Cargo Loader	Used to transfer containers, skids, and pallets to the aircraft's hold.
Baggage Tug	Used to tow luggage trailers from the terminal to the aircraft and back.	Ground Power Unit (GPU)	Ground-based mobile generator that supplies electricity to the aircraft while parked.
Belt Loader	Used to transfer baggage from trailers on the tarmac to and from the aircraft's hold.	Lifts	Used for moving cargo and equipment around the airport, storage areas, or hangars.
Bobtail	A truck cab with no cargo bed mounted to the chassis. Used for a variety of operations.	Service Trucks	Generally onroad vehicles that provide a variety of aircraft support operations.

In 2004, the EPA finalized its rule to reduce emissions from nonroad diesel engines, integrating engine and fuel controls as a system to gain the greatest emission reductions. The new engine standards, to be phased in between 2008 and 2014, will reduce PM and NO_X emissions by 90% and will apply to new nonroad diesel-powered equipment at airports. Given the durability of diesel GSE, however, achieving near-term reductions will require efforts to reduce emissions from existing engines.

Federal and State Authority

Aircraft engine emission standards are set internationally by the International Civil Aviation Organization (ICAO). ICAO is a United Nations body created in 1947 under the Convention on International Civil Aviation (often termed the "Chicago Convention"). To date, 188 nations have signed the Chicago Convention and are thus contracting States to the ICAO. The EPA also sets emissions standards for aircraft engines; historically the U.S. has elected to conform its aircraft engine emissions standards to those developed by ICAO. Under the current international process through the ICAO, the introduction of new aircraft engine emissions standards holds little promise to reduce overall aircraft emissions. ICAO standards for aircraft engine emissions have been in place since 1981, covering emissions of NOX, CO, and HC. PM emissions are not addressed by the standards. In 1998, a more stringent NOX standard was recommended. These standards were adopted by EPA in November 2005 (EPA-FR 2005). While the new standards affect engines designed after 2003, the standards do not apply to engines currently in production and designed prior to 2003.

Emissions standards for ground service equipment are set at the federal level by the EPA and by the California Air Resources Board. States outside of California, under authority established by Section 209 of the Clean Air Act, can adopt the California standards for nonroad engines in lieu of the federal standards.

Technical Options for Emissions Reduction

Aircraft Engines. There are methods to change how aircraft are operated that can have the effect of reducing emissions. However, all are under immediate control of the pilot, and ultimately the choice is up to the airlines as to whether to instruct pilots to use these techniques. These include minimizing the use of reverse thrust, single-engine taxi, and shutting off main engines while idling. Such techniques typically are beyond the control of the airport operator as the party normally responsible for reducing emissions on the airport property. As an example of effectiveness, Delta Airlines' implementation of single-engine taxiing at its Atlanta hub led to a $5.9 million reduction in fuel costs in 1995 alone, as well as a proportional reduction in emissions (Amin 2001). Some aircraft are unable to perform single-engine taxi, and on capable aircraft, crews must be trained in the technique. Given the limited authority states have to control aircraft-related emissions, this section will focus on reducing emissions from ground service equipment and ground access vehicle emissions.

Ground Service Equipment. Technical options to reduce emissions from ground service equipment are similar to those for other nonroad land-based diesel engines: retrofits, rebuilds, replacement, and clean fuels. One replacement option is particularly cost-effective for reducing GSE emissions: gate electrification. Gate electrification in some cases pays for itself in fuel savings within two years of installation. Also, the use of alternative fuels is possible given that ground service equipment is centrally fueled. Operation of factory-built, dedicated compressed natural gas (CNG) and liquid propane gas (LPG) ground service equipment reduces emissions of NO_X, HC, CO, and PM relative to gasoline and diesel-powered equipment. Conversions from diesel or gasoline to CNG/LPG provide NO_X and PM reductions but can sometimes increase HC and CO emissions.

Several major U.S. airports are converting their GSE fleets from diesel to electric equipment. Notable are projects undertaken at Dallas-Fort Worth (DFW) and San Francisco International (SFO) Airports (FAA ILEAV 2006). The primary impetus is to achieve significant reductions in ozone precursor pollutants and to save on the cost of diesel fuel. Typically, there is a net cost savings over the life of the equipment. However, there is a considerable initial outlay involved with purchasing the GSE and the charging equipment. For example, a project at DFW in which 146 baggage tugs and 10 belt loaders were replaced with new electric models cost about $4.3 million (FAA ILEAV 2006).

Table 3-30. Hypothetical GSE Electrification Project for Newark Liberty International Airport.

Source: Ricondo 2006, EPA-GSE 1999, Koroniades 2006

Unit Type	Unit Count	Unit Cost ($)	Total Cost ($ x 10^6)	Tons PM$_{2.5}$ Reduced (9 year life)	Cost Effectiveness ($/Ton)
Baggage Tug	14	$27,500	$0.39	7.0	$55,000
GPU	200	$30,800	$6.2	159.8	$38,548
Pushback Tractor	200	$105,000	$21.0	177.2	$118,510
Start Unit	150	$30,800	$4.6	106.5	$43,380
Service Truck	246	$49,500	$13.1	83.8	$155,943
Cargo Loader	150	$34,100	$5.1	57.6	$88,802
Totals	978	--	$50.3	591.9	Avg: $85,062

Airport-related emissions are a very small part of the overall PM$_{2.5}$ inventory for New York State. Outside of the New York metropolitan area, the airports are so small that their contribution to overall PM$_{2.5}$ hardly registers in the inventory. A viable GSE replacement program would need to focus on the three large airports in the metropolitan area in order to achieve PM$_{2.5}$ reductions of any significance. Table 3-30 and Table 3-31 present examples of reductions that could be achieved through a hypothetical strategy in which diesel-powered GSE are replaced with electric models. While unit cost and cost-effectiveness of the projects outlined in the tables below may be comparable to similar projects which target other non-road sectors, the overall PM emissions from GSE are relatively low. For example, Table 3-23 shows that there are more than 4000 tons of PM$_{2.5}$ from construction and mining equipment in the New York metropolitan area. In contrast, Table 3-28 shows that there are only about 130 tons of PM$_{2.5}$ emissions from GSE in the metropolitan area. Therefore, a GSE electrification project, particularly if it was being done for the primary purpose of achieving significant PM$_{2.5}$ reductions in the metropolitan area, may be a lower priority than a construction equipment retrofit project.

Table 3-31. Hypothetical GSE Electrification Project for JFK & LaGuardia Airports.

Source: Ricondo 2006, EPA-GSE 1999, McGarry 2006

Unit Type	Unit Count	Unit Cost ($)	Total Cost ($ x 10^6)	Tons PM$_{2.5}$ Reduced (9 year life)	Cost Effectiveness ($/Ton)
Baggage Tug	263	$27,500	$7.2	136.6	$52,947
GPU	150	$30,800	$4.6	117.2	$39,420
Pushback Tractor	127	$105,000	$13.3	110.1	$121,117
Start Unit	46	$30,800	$1.4	32.1	$44,137
Belt Loader	145	$34,100	$4.9	31.6	$156,472
Cargo Loader	68	$34,100	$2.3	25.5	$90,933
Totals	799	--	$33.7	453.1	Avg: $74,377

Ground Access Vehicles. Alternative fuel vehicles are a viable option for fleets that do not leave the airport property and can refuel easily, such as shuttle buses. Improving the flow of vehicles circulating through the airport can reduce emissions from idling. However, the same flow improvements may actually result in increased use and increased circulation (i.e, rather than parking) of private automobiles by drivers who would otherwise be averse to airport congestion. An individual's choice of transportation mode, traveling to and from the airport depends both on the the availability and price of parking and the ease of circulation. Increasing the price of parking in an effort to discourage the use of private vehicles for airport access may just encourage vehicle circulation.

San Francisco International Airport is an example of an airport authority that has made a major commitment to reducing emissions from GSE and GAVs by converting to alternative fuel technologies and promoting the use of public transit. For example, the airport authority has converted or replaced 500 airfield vehicles with natural gas, propane, or electric powered units, and their on-airport electric Air Train system has eliminated 600 daily rental car shuttle bus trips (SFO 2005).

Policy Options

A variety of regulatory and policy options exist for states, localities, and airport operators to control airport-related emissions. Innovative programs have been initiated at many airports around the world (Amin 2001).

Onroad vehicles contribute significantly to overall emissions generated at airport facilities. Emissions from vehicles visiting the airport (autos, taxis, vans, buses, etc.) are usually only counted after the vehicles enter the airport property. Vehicle use by airport employees, airport visitors, and ground transportation providers, and the transportation of people and goods between airport facilities produce onroad mobile emissions at an airport. Because employee trips occur daily, one potentially viable strategy to curb emissions is to find ways to limit these trips or reduce their congestion impact. Some mitigation measures to be considered include:

- Implementing variable shifts as a congestion reduction measure to mitigate the employee "rush hour" effect
- Providing rideshare and carpool incentives for employees
- Providing transit or alternative mode incentives or subsidies for employees
- Increasing long-term parking fees for airport visitors
- Establishing idle time limits for passenger vehicles
- Providing passenger and employee satellite parking with shuttle bus service
- Establishing idle time limits for taxis and buses
- Establishing idle restrictions for delivery, service, and commercial vehicles
- Improving circulation management for on-call vans and shuttles
- Switching to alternative fuels for airport shuttle buses

- Replacing gas- and diesel-powered shuttle buses with electric shuttles
- Switching to alternative fuels for delivery, service, and commercial vehicles as well as for taxis and rental cars
- Extending rail service to the airport, or implementing shuttle bus service from rail to airport.

A passenger's ground transportation choice depends on a number of factors including cost, convenience, and availability. For example, multi-stop city buses may be a very inexpensive option for travel from an airport to a city center, but they have inherent disadvantages. Typically they are not set up to handle quantities of luggage, they take longer to reach a destination as compared to other transportation modes, they are not available on demand, and their stopping points typically do not coincide with the passengers' specific destinations. Other forms of public transportation (e.g., express bus, door-to-door van, rail), where available, may have a higher convenience factor, but typically at a higher cost. In the U.S., private transportation (e.g., personal cars, rental cars, taxis) largely dominates the airport transportation sector. Alternatively fueled onroad vehicles, like alternatively fueled ground service vehicles, require a financial investment and incentives for operators.

The Vision 100—Century of Aviation Reauthorization Act (Vision 100), signed into law in December 2003, established a voluntary program to reduce airport ground emissions at commercial service airports in air quality nonattainment and maintenance areas. The provisions are intended to help airports meet their obligations under the Clean Air Act and to assist regional efforts to meet National Ambient Air Quality Standards (NAAQS). Vision 100 directs the Federal Aviation Administration (FAA) to issue guidance, describing eligible airport low-emission modifications and improvements and how airport sponsors should demonstrate program benefits (FAA NPIAS 2004).

The FAA is implementing Vision 100 airport emission provisions in a single program called the Voluntary Airport Low Emission (VALE) program (FAA VALE 2005). Participation in the VALE program is entirely voluntary for the airport sponsors and state air quality agencies. The goal of the VALE program is to reduce air emissions generated by ground transportation sources at airports. The program is designed to provide sponsors with financial and regulatory incentives to increase their investments in proven low-emission technology. The program encourages the use of alternative fuel vehicles (AFVs) and other low-emission technologies that are particularly suited to the airport environment.

Funding for the VALE program is provided through two airport assistance programs. The FAA Airport Improvement Program (AIP) provides grants to airports from the Aviation Trust Fund; the Passenger Facility Charges (PFC) program approves locallyimposed fees from airline passengers for eligible airport development. These programs offer substantial resources to airports for low-emission activities but only if such activities represent a higher priority for the airport than other needed airport development. Vision 100 also requires the EPA, in consultation with the FAA, to issue guidance on how airports can receive airport emission reduction credits (AERCs) for VALE projects and how to apply those credits to future airport projects in order to meet regulatory requirements under the Clean Air Act. The eligibility guidelines, requirements, and procedures for the VALE program are based on established AIP and PFC program regulations, the experience of the FAA with the Inherently Low Emission Airport Vehicle pilot program (FAA ILEAV 2006), and the statutes and orders governing airport development. Through the program, states have the opportunity to realize the environmental and public benefits of early reductions in airport emissions in exchange for granting emission credits to airport authorities.

To date, ozone nonattainment issues have been the driving force for airport authorities to develop VALE projects. Short of a finding that ground equipment is contributing significantly to PM nonattainment, there is little incentive for an airport authority to initiate a VALE project that specifically targets PM emissions. As illustrated in Table 3-32, if New York were considering a VALE type project to address a PM nonattainment problem, ultimately it would be determined not to be cost-effective for the relatively small amount of PM reductions that would be achieved. Implementing a VALE project requires an airport authority to come up with a funding match—and to make the decision not to use that amount of their annual federal funds allotment for other purposes such as noise abatement. Consequently, relatively few airport authorities are applying for VALE projects (Stanco 2006), and those that do apply are focusing more on the potential NO_X and hydrocarbon reduction benefit. To date, Albany International Airport is the only New York State airport that is implementing VALE projects. Table 3-32 and Table 3-33 summarize the vehicle changes occurring under Phase I of the Albany VALE project, including cost-effectiveness and pollutants reduced. Table 3-33 emphasizes the fact that the impetus for this particular project is to reduce ozone precursor pollutants, not $PM_{2.5}$.

Table 3-32. VALE Project Costs – Albany International Airport.

Source: Albany 2005

Equipment	Technology	Increased Cost
M-65 Dump Truck	BAF Hybrid Retrofit	$30,000
M-44 Sweeper	BAF Hybrid Retrofit	$30,000
M-46 Sweeper	BAF Hybrid Retrofit	$30,000
CNG Fuel Station	Storage and Dispensing	$167,000
E-450 Shuttle Bus	BAF CNG w/ oxidation cat	$20,200
E-450 Shuttle Bus	BAF CNG w/oxidation cat	$20,200
Baggage Tractor	Electric Replacing Diesel	$8,843
Baggage Tractor	Electric Replacing Diesel	$8,843
Baggage Tractor	Electric Replacing Diesel	$8,843
GSE Recharging Station	Posi-Charge Sys and Stand	$140,868

Table 3-33. Pollutants Reduced and Cost-Effectiveness of Albany VALE Project.

Source: Albany 2005)

Pollutant	Lifetime Reductions (Tons)	Cost Effectiveness ($/Ton)
NO_x	42.4	$9,949
Hydrocarbons	6.8	$61,867
Ozone	49.2	$8,570
Particulate	None Determined	None Determined

Federally funded projects must conform to the state implementation plan (SIP); i.e., they must not adversely affect timely attainment and maintenance of NAAQS, cause or contribute to new violations of an air quality standard, increase the frequency or severity of an existing violation, or delay timely attainment of a standard or required interim emissions reductions or milestones. In nonattainment areas, a specific airport emissions budget, including an emissions growth factor, may be established in the SIP (Amin 2001). Airport emissions budgets must include emissions from aircraft, ground service vehicles, onroad mobile sources, and many other sources.

In a conformity analysis, both direct and indirect project emissions must be addressed. According to 40 CFR Part 51, "direct emissions" means "those emissions of a criteria pollutant or its precursors that are caused or initiated by the federal action and occur at the same time and place as the action." Indirect emissions are "those emissions of a criteria pollutant or its precursors that (1) are caused by the federal action but may occur later in time and/or may be further removed in distance from the action itself but are still reasonably foreseeable; and (2) the federal agency can practicably control and will maintain control over due to a continuing program responsibility of the federal agency."

If a federal project's emissions are expected to exceed the budget outlined in the SIP, there are three options to achieve conformity. The first is to use emissions offsets so there is no net increase in emissions. Mitigation measures, a second method, are designed to decrease emissions related to the project itself below a certain *de minimis* level. An example of this is to use alternative fuel in ground service vehicles to offset increased aircraft activities. In contrast to offsets, mitigation measures are directly related to and simultaneous to the project. Offset and mitigation measures must be enforceable commitments, adhering to a timetable. Another way to achieve conformity is to modify the SIP. This requires that the current SIP is being implemented, all mitigation measures are being implemented, and that the emissions decrease will occur before the associated increase resulting from the federal action.

Given the projected growth in emissions at these airports, it is relevant to consider alternative options to reduce aircraft emissions, such as emissions bubbles (CCAP 2005). A "bubble" is a conceptual limit placed around total emissions, either for the airport as a whole or for a distinct category of sources or operations within the airport (e.g., aircraft, APUs, GSE, GAVs, and stationary sources). Emissions within the bubble are then limited by a defined cap or budget. Emissions from any individual source within the bubble may vary as long as the overall cap or budget is not exceeded. The emission limit may be fixed, decline over time, or allow for growth.

Efforts at the national and international levels to increase the stringency of engine emission standards could play a role in reducing air pollution from aviation and in driving technology development. “Cap-and-trade” or airport “bubble” approaches have the potential to limit airport-related emissions, provide flexibility in achieving reductions, and encourage the use and development of cleaner technologies. The operators of Logan Airport in Boston have established a cap on airport emissions; any increases that result from airport activity must be offset by on-airport emissions reductions, reductions near the airport, or by purchasing emissions credits.

Fee-based strategies, such as increased or variable landing fees, are another potentially useful tool that officials at the state, local, and airport level can use to reduce emissions. Variable aircraft landing fees have been implemented at Zurich and Geneva Airports in Switzerland, and at 19 airports in Sweden. The fees are emissions-based and result in a greater charge being levied on higher polluting aircraft entering those airports. Regulatory approaches, such as 1) promoting or requiring the purchase of cleaner alternatives when fleet vehicles or equipment are replaced or added; and 2) developing a declining fleet emissions target that can be used to achieve emissions reductions from ground service equipment and ground access vehicles.

Recommendations

In light of the low particulate emission contributions from this sector when compared to other sectors, there are few strategies to recommend for reducing carbonaceous $PM_{2.5}$. However, a few strategies may merit further consideration because they have other desirable co-benefits. For example, with jet fuel prices at record highs, there is renewed interest in reducing fuel consumption. Strategies such as single-engine taxiing and greater use of push-back tugs have the combined benefit of reducing fuel consumption and reducing emissions.

- Make use of grants programs and other financing options (e.g., VALE Program, New York Power Authority) to fund conversion GSE from gasoline and diesel-powered engines to electric.
- Convert ground access vehicles to using alternative fuels or retrofit with particulate filters. Vehicles such as shuttle buses are well suited to this approach.

MARINE PORTS

This section discusses sources of $PM_{2.5}$ emissions at marine ports in New York State. It describes relevant regulatory authority for addressing these sources, and control strategies available to New York State and local authorities to reduce $PM_{2.5}$ emissions from these operations.

Sector Profile

Together, the seven major marine cargo terminals located around New York Harbor constitute a significant portion of the largest "port" in New York, and the third largest in the U.S., based on total annual freight handled as well as on the number of annual ship calls. The marine terminals in New York Harbor received over 9,700 ships in 2003 and handled 146 million short-tons of freight (DOT-BTS 2004). Only the ports of Los Angeles and Long Beach together, as well as the port of Houston, received more ship calls, and only the ports of Houston and South Louisiana handled more tonnage.

Figure 3-11. Marine Terminals in the Port of New York/New Jersey.

Source: PANYNJ 2006

Four of the marine terminals in New York Harbor, including the largest, are in New Jersey, and three are in New York City. Six of the seven are leased to private operators and maintained by the Port Authority of New York and New Jersey (PANYNJ) while one is privately owned (the Global Marine Terminal). With the exception of the Global Marine Terminal, which is located in Bayonne, NJ, these New York Harbor marine terminals are shown in Figure 3-11.

In addition, there is significant marine vessel traffic in New York Harbor that is not associated with the facilities managed and maintained by the PANYNJ. These include vessels calling at various privately owned petrochemical facilities (New York Harbor is one of the leading petroleum handling ports in North America) and smaller liquid or solid bulk facilities, and those vessels that call at the port only to refuel. There are also vessels passing through the port en route to up-river destinations.

As is happening nationally, freight traffic into and out of the Port of New York/New Jersey continues to grow. In 2005, the volume of containerized cargo handled by PANYNJ marine terminals increased 7.6% compared to the previous year (PANYNJ 2006a).

The New York City-owned Passenger Ship Terminal on the Hudson River in Manhattan is the sixth largest in the U.S. In 2003 it handled 212 ship calls and boarded over 400,000 cruise passengers (DOT-BTS 2004). The City's Brooklyn Cruise Terminal as well as a cruise ship facility in Bayonne, New Jersey, have recently started operating. Therefore, marine-related emissions from these facilities are not included in these vessel statistics. In addition to these freight and cruise terminals, a number of companies operate commuter ferries and sightseeing excursion boats within the harbor from multiple terminals on each side of the Hudson River. After September 11, 2001, when train service to lower Manhattan was disrupted, private ferry traffic doubled in New York Harbor, to 1,000 trips per day (NYSERDA 2003). Other sources of diesel emissions within New York Harbor include emissions from harbor craft, such as municipal and privately owned ferries, tugboats, towboats, and marine construction vessels.

Other marine ports in New York State are significantly smaller than New York Harbor. Table 3-34 lists the five largest ports in New York State. As can be seen from the table, the second largest port—the port of Albany—was ranked as the 71st largest port nationally in 2000, but it handled only 4% of the freight tonnage handled in New York Harbor. Port Jefferson, on the eastern end of Long Island, handles automobile ferry service across Long Island Sound to Connecticut. Ports in New York State are owned and operated under a number of different arrangements. The Port of Buffalo is privately owned by Gateway Metroport, while the Port of Albany is operated by the Albany Port District Commission. The Port of New York/New Jersey is a mix of publicly and privately owned facilities. Included in this mix are the marine terminals that are maintained by the PANYNJ, which it leases to private operators, making it a "landlord port."

Table 3-34. Cargo Volumes of New York Ports, 2000.

Source: DOT-BTS 2006

Port	U.S. Rank	Cargo Volume (mill short tons)
New York/ New Jersey	3	138.7
Albany	71	6.1
Port Jefferson	108	2.8
Buffalo	119	2.2
Hempstead	148	1.2

Sector Emissions

In 2003, the PANYNJ developed an inventory of emissions from commercial marine traffic within New York Harbor in 2000 (PANYNJ 2003a).[39] The results of this inventory are shown in Table 3-35. According to this inventory, in 2000 marine vessels emitted 461 tons of $PM_{2.5}$ within the New York City

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[39] The geographic area of the inventory includes all of the New York and New Jersey counties within the New York City nonattainment area. For ocean-going ships it includes all emissions within a zone extending out three nautical miles from shore.

nonattainment area. The largest contributors were ocean-going vessels, which emitted almost half of the total $PM_{2.5}$ from marine vessels within the harbor. Emissions from auxiliary engines, which provide power to ocean-going vessels while they are in the nonattainment area (in port), represent approximately one-third of the emissions from ocean-going vessels. The second largest contributors were towboats used to move bulk freight barges around the harbor (31%), followed by ferry and excursion boats (12%). This inventory did not include emissions from vessels calling at some facilities in New York Harbor such as the New York City Passenger Ship Terminal.

PANYNJ also developed an inventory of landside emissions from the material-handling operations within the PANYNJ container and automarine terminals in the Port of New York/New Jersey in 2002. The inventory did not include emissions from the onroad trucks that call to drop off and pick up containers and import/exported vehicles at the terminals, or line-haul locomotives similarity used to haul freight to and from the terminals (PANYNJ 2003b). The results of the PANYNJ landside emissions inventory are shown in Table 3-36. The table shows that the great majority of emissions from nonroad equipment at the Port are from cargo-handling equipment. Since the Automarine operations use trucks, which are considered onroad vehicles, the nonroad emissions in the table for this category are zero.

Table 3-35. New York Harbor Commercial Marine Emissions, 2000.

Source: PANYNJ 2003

Vessel Type	$PM_{2.5}$ (tons/yr)	% of Total
Ocean-going Vessels	215.7	47%
Towboats	142.1	31%
Assist Tugboats	34.1	7%
Ferry & Excursion	55.2	12%
Government	4.4	1%
Dredging Operations	9.8	2%
TOTAL	461.4	100%

Table 3-36. Port of New York/New Jersey Nonroad Landside Emissions, 2002 (PANYNJ marine terminal operations).

Source: PANYNJ 2003

Source	$PM_{2.5}$ (tons/yr)	% of Total
Nonroad Material Handling Equipment	124.5	98%
Locomotive	2.1	2%
Automarine Operations	0.0	0%
TOTAL	126.6	100%

$PM_{2.5}$ emissions from various drayage (truck) operations at the Port are a potentially large contributor to total $PM_{2.5}$ emissions in New York Harbor. The PANYNJ estimated that 121million miles are driven each year by drayage trucks traveling within a 75-mile radius of the port. However, the study also indicates that drayage truck miles traveled associated directly with port activities constitute only 1.8% of these truck miles in the region (PANYNJc).

PNYNJ "Green Ports" and Other Emission Reductions Initiatives. Recent actions undertaken as part of the PANYNJ Green Ports Initiative and the Comprehensive Port Improvement Plan have resulted in significant reductions in PM emissions at the Port. These recent efforts include several components: (1) crane electrification, (2) replacement of nonroad cargo handling equipment (CHE) with new pieces that come equipped with cleaner highway engines, (3) shifting movement of goods from truck to rail (on-dock rail), (4) terminal redevelopment, (5) truck gate electrification, and (6) longer gate hours of operation (PANYNJ 2006 "The Environment"). A PANYNJ study to evaluate the change in cargo-handling equipment emissions between 2002 and 2004 found that PM emissions went down 32% in this time period (Starcrest 2005). The reduction in emissions occurred at the same time that the number of pieces of equipment in use and containers moved went up by 19% and 25% respectively.

In a separate effort, three New York Harbor retrofit and rebuild projects have demonstrated significant emissions reductions. These include: the retrofit of one Staten Island ferry with selective catalytic reduction (SCR) and the rebuilding of two Staten Island ferries using engines with cleaner emissions standards. Also included is a project to retrofit 42 private ferries in New York Harbor with diesel oxidation catalysts and the rebuilding of tugboat engines used between Port Jefferson and Bridgeport. The Staten Island SCR project is aimed at reducing NO_x and was done to offset emissions from the Harbor Deepening Project under the Harbor Air Mitigation Plan (HAMP). The private ferry retrofit project is expected to reduce PM emissions by 11 tons per year when all retrofits are completed in 2006 (Ralbovsky and Silecchia 2006)[40] Retrofits on 17 boats have been completed to date.

The inventory work described above has revealed some shortcomings in the understanding of port-related emissions. For example, while much of the inventory information cited in this section is from the PANYNJ, a large fraction of port-related emissions comes from facilities not associated with PANYNJ facilities. This is borne out by statistics on ocean-going vessel calls to New York Harbor. DOT data shows that over 9,000 ocean-going vessels called at New York Harbor in 2003, while PANYNJ data shows that just over 5,000 calls were made to PANYNJ facilities.

Federal and State Authority

Some New York counties and municipalities have regulations related to particulate emissions from existing marine equipment. New York City (NYC AC 24-164) restricts the sulfur content of fuel used in marine vessels to no more than 0.2% by weight (2,000 ppm) for No. 2 fuel oil and 0.3% (3,000 ppm) for No. 4 fuel

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[40] This project was funded with a $5 million grant from the Federal Transit Administration and $1.8 million in federal Congestion, Mitigation, and Air Quality program funds.

oil and residual oil. This fuel sulfur restriction does not apply to "ocean-going vessels engaged in international or interstate trade" but does apply to tugboats, workboats, ferries, and other harbor craft (NYC AC 24-169). The City of Buffalo has limits on emissions of "dense smoke" from sources that include marine vessels (BUF CC 66-3; 66-25). Rochester also prohibits the emissions of smoke greater than a specified density from any "locomotive, tug, boat…" (RC 100-3, 100-5). Marine vessels are excluded from New York State requirements for diesel inspection and maintenance (NYSL Art 19-0320; NYSR 217-5), as well as from diesel vehicle idling restrictions (NYSR 217-3). Additional authority for some control options in New York State is described below under specific technical options.

Technical Options for Emissions Reduction

This subsection presents information on technical approaches to reduce emissions from mobile sources operating at ports. It includes options for those sectors where the greatest opportunity to reduce emissions exists within the New York Harbor, followed by opportunities that provide lesser reductions.

Harbor Craft. Taken together, emissions from towboats, ferries, and assist tugs constituted the largest source of PM_{10} emissions in New York Harbor in 2000 (see Table 3-35 and Table 3-36). Emissions standards for harbor craft engines are set by both the EPA and by the California Air Resources Board. Table 3-37 shows the federal emissions standards for typical Category 1 (C1) harbor craft engines. Category 1 Engines are engines with rated power at or above 37 kW but with a specific displacement of less than five liters per cylinder. These engines are similar to land-based nonroad diesel engines that are used in applications ranging from skid-steer loaders to large earth moving machines. For a complete discussion of emissions standards see Appendix B.[41] Several emissions control strategies for harbor craft engines are described below.

Table 3-37. Marine C1 Engine Emissions Standards.

(EPA2004)

Certification Level	EPA PM (g/kw-Hr)
Tier 0	0.6
Tier 1	0.3
Tier 2	0.12
Tier 3	Proposal expected in 2007

California Harbor Craft regulation. In 2006, CARB proposed a harbor craft rule that in the first phase would require either engine replacement with a Tier 2 engine, or installation of the highest level verified diesel emissions reduction technology. In phase 2, the rule would require replacement with a Tier 3 marine engine. New York has the authority to adopt this regulation. It is

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[41] Tier 0 represents uncontrolled emissions. Tier 1 standards went into effect in 1999 to 2000. Tier 2 standards went into effect in 2004-2005.

important to note that harbor craft emissions are currently being reduced through SEP and Harbor Deepening Project funding, as noted above. However, should emissions reductions become required through future regulation, then it is possible that some of these funding sources would remain available.

Retrofit with an Oxidation Catalyst or Particulate Filter. The ferry retrofit project in New York Harbor (described above) has demonstrated that harbor craft can be retrofitted with DOCs. A smaller subset of harbor vessels can be retrofitted with DPFs. Space constraints, age of engine, state of maintenance, exhaust temperature, and other conditions need to be assessed in order to ensure that harbor craft engines can be equipped with retrofit devices. In addition, it is essential to have access to ultra-low sulfur diesel fuel in order for a DPF to perform properly. For very old, two-stroke engines, retrofitting may not be an option due to high ash levels in the exhaust. For these engines, options listed below are preferable. A detailed discussion of retrofit technologies is presented in Appendix G.

Repower C1 Engines to Tier 2 Standards. PANYNJ funded the repowering of five private tugboats used in New York Harbor (EPA 2005c). This project was done as part of a general conformity requirement offset increased NOx emissions from a 10-year dredging project in New York Harbor, and it is focused on NOx reduction. However, the project will significantly reduce PM emissions from the affected vessels. This program can serve as a model for an expanded effort to reduce PM from marine vessels that operate within New York Harbor and other New York waters.

Alternative Fuels. Alternative formulations of diesel fuel, discussed in Appendix B, can be used in marine and land-side nonroad diesel engines. Often the substitution can be made without any physical changes to the engine.

Low sulfur Fuel. Requiring the use of low-sulfur fuel in advance of the 2012 regulatory requirement will reduce sulfate emissions and facilitate the introduction of emissions control devices such as particulate filters. As mentioned above, there is a NYC law regulating fuel sulfur levels for harbor craft engines. This law could be amended to require low-sulfur fuel.

Table 3-38 shows potential harbor craft emissions reductions that can be achieved using some of the approaches described above.

Table 3-38. Control Options for Harbor Craft ($PM_{2.5}$ tons reduced each year).

Harbor Vessel Type	2002 Baseline Emissions ($PM_{2.5}$ tons/yr)	DOC retrofit (once verified technology available)	Repower Tier 2	Repower Tier 3 (when available)*
Tugboats	176	33	106	158
Ferries/excursion	55	14	33	50
Total	231	58	139	208

* assumes a 90% reduction from Tier 2 levels (assumption made by CARB in its Harbor Craft rule).

The cost for retrofitting ferries with DOCs is approximately $15,000 per ton. Costs for repowering tugboats to a Tier-2 emissions level are $83,000 per ton of PM reduced. This analysis assumes that the cost of repowering a tugboat is $450,000 and the vessel operates for 10 years after it is repowered (PANYNJ 2005).

Land Side Nonroad Engines/Cargo Handling Equipment (CHE). Nonroad engines operating at ports include terminal tractors, forklifts, cranes, straddle carriers, top-loaders, empty-container handlers, and other equipment. Straddle carriers and top loaders are used to move cargo containers around a marine terminal. The PANYNJ cargo handling equipment inventory (2003) found that almost 50% of nonroad machines inventoried were terminal tractors. An additional 15% were straddle carriers, 11% were forklifts, 9% were top-loaders, and 7% were empty-container handlers. The year of manufacture of this equipment ranged from 1970 to 2003 (PANYNY 2003a). A majority of the engines were larger than 175 horsepower.

CARB CHE Emissions Standards. In 2005 CARB adopted a cargo-handling emissions regulation. The regulation calls for the replacement or retrofit of existing engines with ones that use Best Available Control Technology (BACT), and will require, beginning January 1, 2007, that newly purchased, leased, or rented cargo handling equipment emit low levels of PM and NOx. The regulation will reduce particulate emissions from new CHE engines by 85 percent. Section 209 of the Clean Air Act amendments provides states outside of California with authority to adopt California's nonroad engine standards for engines greater than 175 horsepower. (Section 209 prohibits states from adopting the standards for smaller engines.) Legislation to adopt the standards is not required in New York as existing authority is sufficient. Since a majority of CHE is 175 horsepower or more (PNYNJ 2005), New York could achieve substantial particulate reductions by adopting the CARB CHE regulation.

Purchase CHE with Onroad Heavy-Duty Engines. Replacing older pieces of equipment with new ones equipped with cleaner engines can provide substantial emissions benefits as demonstrated by the PANYNJ. Between 2002 and 2004, a large shift to cleaner, Tier 2 CHE engines was made by PANYNJ marine terminal tenants. In 2002, 3% of CHE engines were Tier 2 engines, 70% were Tier 1 engines, and 27% had uncontrolled emissions. In 2004, 36% were Tier 2 engines, 49% Tier 1, and only 15% uncontrolled (PNYNJ 2005). Encouraging all PANYNJ tenants to replace older CHE engines and expanding the modernization program to non-PANYNJ facilities could significantly reduce emissions in New York Harbor.

While the current program to equip nonroad machines with onroad engines is resulting in the replacement of older engines with cleaner engines, the new engines do not require DPFs. Beginning in January of 2007, new heavy-duty onroad (Tier 4) diesel engines will be equipped with DPFs and will emit 90% less PM2.5 than current models. Manufacturers of cargo-handling equipment indicate that ordering yard tractors with new, 2007 engines is feasible, but not other types of cargo handling equipment.

Retrofitting. Three nonroad retrofit devices have been verified by CARB and could be used in cargo-handling equipment. As with marine engines, substantial reductions in organic and elemental carbon can be achieved by retrofitting with particulate filters and oxidation catalysts.

Electrification. Given the mix of fuels used for electricity production in New York State, substitution of electric power for diesel power in land-side nonroad equipment produces net PM reductions of 30– 80%, depending on the size of the engine.[42] Port tenants are already electrifying cranes at the PNYNJ. This approach could be expanded to other facilities that use cranes in New York Harbor. In addition, other land-side material-handling equipment in the port operates within a very limited geographic area for its entire life. Thus, the range limitations associated with battery-powered mobile electric vehicles are not a barrier to electrification {here}. In particular, small and medium-sized forklifts are commercially available with electric drives and are currently being used by PANYNJ tenants where compatible with operations.

Emulsified Diesel Fuel and Biodiesel.

See discussion of these approaches in Appendix B.

Table 3-39 shows potential cargo handling equipment emissions reductions that can be achieved using some of the approaches described above:

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[42] M.J. Bradley & Associates analysis based on New York State electrical generation units included in EPA Emissions and Generation Resource Integrated Database, and PM emissions factors for electricity generation from EPA AP-42, Compilation of Air Pollutant Emissions Factors. PM factors for nonroad equipment based on gram per brake horsepower-hour PM emission standards.

Table 3-39. Control Options for CHE ($PM_{2.5}$ tons reduced each year).

Certification level	Tier 2 Upgrade Tons reduced	DOC + Crank Case Device Tons reduced	DOC + emulsion Tons reduced	DPF (combifilter) Tons reduced	Replace with onroad 2007 engines Tons reduced
Tier 0	14	4	N/A	N/A	N/A
Tier 1	41	10	20 tons	34	15
Tier 2	N/A	4	N/A	12	8
Total	22 tons	23 tons	26 tons	46 tons	24 tons

In Table 3-39 the potential tons of $PM_{2.5}$ that could be reduced from cargo handling equipment at the PNYNJ by using different control measures is calculated. Three retrofit technologies verified by either the EPA or by CARB were evaluated: DOC plus Spiracle, DOC plus emulsified fuel, and an active particulate filter. In addition, two rebuilding and repowering approaches were evaluated: (1) rebuilding engines to Tier 2 standards and (2) replacing yard-hauler nonroad engines with onroad engines. In this analysis, it was assumed that Tier 0 and Tier 1 engines can be upgraded to Tier 2 engines. The DOC plus spiracle option is verified for use in Tier 0, Tier 1, and Tier 2 engines. The DOC plus emulsion combination is verified for use only in Tier 1 engines, and the active particulate filter is verified for use in Tier 1 and Tier 2 engines.

The above analysis assumes that all of the cargo-handling equipment eligible for retrofit with a DOC, closed-crankcase ventilation system, or active particulate filter can be retrofitted. As is always the case when retrofitting diesel engines, low exhaust temperatures, space constraints, maintaining the equipment operator's line of sight and other issues can make equipment ineligible for retrofit. Cost-effectiveness data for the above options are presented in Table 3-40.

Table 3-40. Cost-Effectiveness of PM Control Measures for CHE (dollars per ton of $PM_{2.5}$ reduced).

Reduction Measure	Cost per ton of PM reduced	Reduction Measure	Cost per ton of PM Reduced
Tier 1 DOC retrofit	$23,000	Tier 2 DOC retrofit	$42,000
Tier 1 DPF retrofit	$18,000	Tier 2 DPF retrofit	$33,000
Tier 1 Emulsion + DOC	$321,000	Tier 2 Emulsion + DOC	$280,000
Tier 0 DOC	$32,000	Tier 2 Equip. replacement	$308,000
Tier 0 DPF	$25,000	Tier 4 Equip. replacement	$205,000
Tier 0 Emulsion + DOC	$105,300	Tier 4 onroad engine specification (yard tractors)	$16,000

The analysis only includes capital costs for DOCs, closed-crankcase systems, and DPFs. NESCAUM did not assume any increase in annual costs due to increased maintenance or fuel penalty. Also, the cost-effectiveness calculation does not discount annual costs for emulsified fuel for the Emulsified+DOC option for later years.

As can be seen from the table, the most cost-effective option for retrofit with DOCs and DPFs are the Tier 0 and Tier 1 engines. For emulsified fuel plus DOC, the most cost-effective option is to retrofit Tier 0 engines. Cost-effectiveness for retrofit with hardware (DOC, DPF, and crankcase-ventilation systems) is dependent on both the baseline emissions factor and annual use (hrs). For emulsified fuel, the baseline emissions factor matters exclusively since annual use does not affect dollar-per-ton values. For a DOC or DPF there is a fixed cost that must be paid off —the more it is used the more favorable the cost effectiveness. By contrast, emulsified fuel has no fixed cost, so all the costs are variable.

The Tier 2 and Tier 4 equipment replacement calculations provide cost-effectiveness figures for replacing the entire piece of equipment. In this case, NESCAUM calculated the cost of replacing older, higher emitting yard tractors (Tier 0 or Tier 1) with newer, cleaner yard tractors (Tier 2 or Tier 4 engines). This assumes the cost for a new yard tractor is $65,000 per vehicle (PNYNJ 2005). If yard tractors are gradually replaced with tractors that have 2007-compliant onroad engines (equipped with particulate filters), the cost-effectiveness for PM reduction is more favorable ($15,000 per ton of PM reduced). This assumes an incremental cost of $5,000 more for the specification of an onroad engine as compared to a non-road engine in the same power category. In this case, however, the emissions reductions will be realized more slowly given that the rate of equipment turnover can be slow.

Trucks. Thousands of truck trips are made to terminals each day to drop off and pick up cargo. The total is significantly more if all parts of the truck trip are included. Options for reducing emissions from truck engines are provided below. Details for all of these options are included in Appendix B.

> **Retrofitting.** A large number of retrofit devices have been verified by either CARB or the EPA for use in onroad engines. According to EPA and CARB certification data, up to 90% reduction in $PM_{2.5}$ emissions from trucks can be realized by retrofitting.
>
> **Repowering.** Repowering older engines with 2007compliant onroad truck engines will reduce $PM_{2.5}$ 90%.
>
> **Operational Changes.** As mentioned earlier, the PANYNJ tenants have extended their gate operating hours, are improving rail access to reduce truck trips, and have electrified gates at entries into terminals. Further improvements could be made that reduce queuing and attendant idling. These include improving the layout of terminal gates and staging areas, the use of gate-

appointment scheduling systems, providing incentives for off-peak delivery and pick-up, and the use of common pools of terminal tractors by multiple terminal operators. Some of these have been initiated by PANYNJ tenants. Mandatory idling restrictions for queued trucks can be difficult to enforce, although major terminal operators are pursuing voluntary approaches, such as posting "no idling" signage in the vicinity of their facilities. Another possibility might be to provide incentives for off-peak delivery and pick-up. However, warehouses would then have to operate additional shifts, and concerns have been expressed over after-hours noise impacts on nearby communities.

Greater Use of Rail. The Port of New York/New Jersey currently has three dedicated rail terminals and two intermodal support yards that comprise the ExpressRail intermodal system (PANYNY 2006b). PANYNJ's 2006-2009 investment program includes expansion and upgrade of all five of these facilities to increase capacity. Approximately 14% of the containers shipped through the port are currently moved by rail to/from their inland destinations (PANYNJ 2006b). Increasing rail trips further will reduce PM emissions from trucks.

Create Common Truck Chassis Pool. Drayage trucks are typically hired by shippers or their consignees. Chassis switching by drivers moving containers from multiple operators can add up to an hour per trip and contributes to congestion on port roadways. In October 2004 the Port of Virginia became the first U.S. port to require that all of its terminal operators participate in a common chassis pool, meaning that any vehicle can be used to move any container. This has reduced the number of individual trucks stored and used within the port by 20%—a reduction of 5,000 chassis. Local drivers who had previously been able to move only two to three containers per shift are now completing up to ten moves a day, thereby increasing their income and improving service to customers while reducing total fuel use and emissions (EPA 2005ce.)

Recommendations

Efforts to reduce emissions of direct $PM_{2.5}$ from New York ports could focus on reducing diesel exhaust emissions from harbor craft and cargo-handling equipment. Although New York has limited ability to reduce emissions from ocean-going ships, it does have numerous opportunities to reduce emissions from commercial harbor craft, which in New York Harbor account for over 40% of port-related $PM_{2.5}$ emissions. Efforts to reduce land-based emissions should focus on port-based nonroad material-handling equipment, which accounts for over 20% of port-related $PM_{2.5}$ emissions within New York Harbor. These efforts should focus on expanding the successes already achieved at the PNYNJ. Finally, further study is needed to determine the amount of fine particulates emitted from trucks operating at the Port. If inventory results show trucks are a significant source, then controls may be considered. Specific recommendations to implement these efforts are given below.

Reduce Harbor Craft Emissions:

- Focus on reducing harbor craft emissions by expanding DOC retrofit and Tier 2 repower programs already underway.
- Amend NYC law NYC AC 24-164 so that it requires the use of onroad diesel fuel in harbor craft engines.

Expand PNYNJ Programs to Other Facilities in New York Harbor and Other New York Ports:

- Estimate emissions from non-PNYNJ terminals and operations.
- Evaluate how the CHE Tier 2 engine, crane electrification, on-dock rail, and reducing truck-idling time programs at the PNYNJ can be expanded to the other private facilities at the Port of New York (Global Marine Terminal, Cruise ship terminal, and petrochemical facilities such as Exxon and BP).
- Evaluate how the CHE Tier 2 program and other programs can be exported to other ports in the tri-state area.
- Evaluate how the on-dock rail, crane electrification, expanded port hours, and other changes can be expanded to other ports in New York Harbor and in the tri-state area.

Further Reduce Emissions from PNYNJ Facilities:

- Establish an understanding of the role of trucking logistics. The PNYNJ is planning to conduct a study to estimate emissions from trucks at the Port, and information on trucking logistics is needed for such estimates. As part of the Gateway Cities project on the West Coast, efforts to reduce drayage emissions are underway. Some of the West Coast initiatives may be transferable to the East Coast, but an analysis of contractual arrangements and intermediaries is needed in order to understand if the programs can be used in New York.
- Evaluate CHE emissions improvements periodically and consider adoption of the CARB CHE regulation. Legislation is not required in order for New York to adopt California's emissions standards, since existing authority allows for adoption without further legislation.
- Conduct pilot programs to evaluate potential strategies to reduce emissions from CHE that cannot be repowered with Tier 4 2007 compliant highway diesel engines.

CONCLUSIONS

A range of control options are available to reduce primary carbonaceous $PM_{2.5}$ emissions in New York State. Eight sectors were evaluated for this report: commercial meat-cooking operations, residential fuel combustion and electricity use, light-duty motor vehicles, onroad heavy-duty vehicles, nonroad engines, airports and aircraft, marine ports, and other mobile sources. (The last sector is included under the discussion of nonroad engines.) Specific control measures considered most promising are summarized for each sector. The combined strategies outlined could reduce 10,000 tons of $PM_{2.5}$ in the NAA each year, or one-quarter of direct carbonaceous $PM_{2.5}$ emissions. The estimate does not take into account the potentially large additional reductions in secondary organic $PM_{2.5}$ that could be realized through reductions in motor vehicle and other gasoline engine hydrocarbon emissions. A suite of additional options is outlined in the chapter.

According to inventory data, the largest single contributor in New York City to primary $PM_{2.5}$ emissions is nonroad engines (24%), followed by residential fuel combustion (19%), charbroiling (18%), onroad heavy-duty diesel engines (13%), other mobile sources, which include recreational marine and nonroad gasoline engines (10%), and automobiles (7%). Approximately 58% of primary carbonaceous emissions in the New York City metropolitan nonattainment area are from mobile sources and 42% from stationary sources. Approximately 92% of primary elemental carbon comes from mobile sources. Charbroiling dominates the organic carbon inventory in the NAA. Charbroiling is an urban issue, as emissions from charbroiling are not a large contributor in areas outside of the New York City nonattainment area.

In rural areas, residential wood burning dominates the organic carbon inventory. This is an issue of exposure rather than $PM_{2.5}$ attainment, since the greatest emissions from wood burning take place outside of the New York City nonattainment area. Diesel engines and gasoline cars contribute only about 5% of rural-area $PM_{2.5}$ emissions because sources like residential fuel combustion (primarily wood burning) tend to dominate the rural inventory. However, $PM_{2.5}$ tons emitted from diesel engines and gasoline cars outside of the NAA are equivalent to tons emitted within the nonattainment counties. Thus, exposures upstate may be high in urban areas such as Albany, Syracuse, and Buffalo.

Policy options to reduce $PM_{2.5}$ emissions from nonroad engines include the adoption of retrofit requirements similar to a proposed CARB rule, to be voted on by the CARB Board} in spring, 2007. Assuming one-third of the construction equipment in the NAA is retrofitted with a mix of technologies, approximately 1,000 tons of $PM_{2.5}$ could be reduced each year. A loan fund for installation of anti-idling devices for switchers and locomotives could produce additional cost-effective $PM_{2.5}$ reductions.

Residential fuel combustion represents the next largest $PM_{2.5}$ source in the metropolitan area and is the single largest source of emissions in the more rural areas. A strategy to attain the $PM_{2.5}$ NAAQS and reduce public exposure to $PM_{2.5}$ should also address emissions from this source. Major options include requiring controls for outdoor wood boilers and wood stove change-out programs; these could result in emissions reductions on the order of several thousand tons statewide.

$PM_{2.5}$ associated with charbroiling (largely meat grilling) comprises the next largest source of the carbonaceous $PM_{2.5}$ in the nonattainment area. Approximately 10% of the $PM_{2.5}$ comes from chain-driven charbroilers and 74% from underfired charbroilers. If restaurants that use chain-driven charbroilers were required to retrofit their equipment with catalytic oxidizers, a reduction of 275 tons of $PM_{2.5}$ per year could be realized. The South Coast Air Quality Management District Rule 1138 could be used as a model. The State could also consider a requirement to retrofit underfired charbroilers with control technologies, which could yield an emissions reduction of 3,000 tons, assuming a 100% compliance rate.

Another important combined category, light- and heavy-duty vehicles emissions, account for 20% of $PM_{2.5}$ emissions in the NAA. This category includes road dust, which comprises 50% of the total. As discussed in the chapter, motor vehicle emissions have been linked in a number of studies to elevated $PM_{2.5}$ exposures—both in urban and more rural areas. Several recommendations stand out for heavy-duty vehicles. Approximately 75% of heavy-duty truck $PM_{2.5}$ emissions are from Class 8 trucks and school buses combined. Retrofitting candidate vehicles with particulate filters could reduce $PM_{2.5}$ by 1,500 tons per year. CARB's truck retrofit regulations and the New Jersey school bus retrofit requirement provide good models. New York could also consider adopting a requirement similar to California's truck refrigeration regulation. If implemented in the New York City NAA, this would reduce $PM_{2.5}$ by approximately 1,300 tons of annually.

For light-duty vehicles, New York could consider a subsidized vehicle repair program for low-income vehicle owners, since older vehicles emit the majority of passenger car $PM_{2.5}$ emissions. Improved maintenance would address emissions of primary $PM_{2.5}$ as well as hydrocarbons and their associated secondary $PM_{2.5}$. Two programs, one in California and the other in Spokane, Washington, provide a template for such a program. A congestion pricing program such as the one implemented in London, could be considered for New York City. This may reduce light-duty exhaust $PM_{2.5}$ emissions along with brake, tire, road wear, and road dust by 15%, and could reduce $PM_{2.5}$ by 500 tons per year. The New York City Council would need to pass a regulation to establish this program.

In order to reduce road dust related to motor vehicles, a pilot demonstration could be conducted to determine the effectiveness of new-technology street sweepers. The 1997 South Coast Air Quality Management District (SCAQMD) Rule 1186 provides a model that could be applied to urban areas in New York State. Substantial $PM_{2.5}$ reductions may be possible, as PM_{10} reductions of 90% have been demonstrated.

A measurable fraction (4%) of carbonaceous $PM_{2.5}$ in the New York metropolitan area comes from lawn and garden equipment. The State could work with the EPA to develop a stringent emissions control program for lawn and garden equipment similar to the program that CARB has finalized. That rule requires a 70% reduction in NO_x and HC, which will also result in both primary and secondary $PM_{2.5}$ reductions.

Mobile sources operating at airports and ports emit approximately 5% of carbonaceous $PM_{2.5}$ in the metropolitan area. One approach to reduce emissions is to require electrification or a best available control technology (BACT) retrofit of ground service equipment, using the CARB ground service equipment regulation as a model. Two other CARB rules require emissions reductions from harbor craft and cargo handling equipment; these could be adopted by New York State. Finally, expansion of emission reduction initiatives at the PANYNJ could be expanded to other facilities in New York Harbor. These programs taken together could eliminate approximately 350 tons of $PM_{2.5}$ per year.

The emissions control strategy recommendations presented here rely on current information on emission inventories, ambient measurements, and the state of technology. Although a considerable body of knowledge exists, substantial gaps remain. Gaps in understanding how emission control technologies can be applied more broadly also exist. For example, a pilot study on control technologies for underfired charbroilers could assist policy makers in developing a strategy to reduce emissions from this large, uncontrolled source sector. Use of particulate filters in cargo-handling equipment, marine harbor craft engines, and locomotives are needed. A demonstration project for vacuum-assist street sweepers could point toward a cost-effective approach to reduce road dust emissions. At the same time it is important to keep in mind that there is little uncertainty regarding the public health implications of exposure to $PM_{2.5}$, even at levels below federal standards. This by itself should serve as sufficient impetus to move forward with the most potentially viable emission control programs.

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Appendix A

NEW YORK STATE $PM_{2.5}$ INVENTORY

INTRODUCTION

The Environmental Protection Agency (EPA), with input from the states, develops a comprehensive National Emissions Inventory that includes details on emissions of criteria pollutants from various mobile and stationary source categories. The national inventory is also broken out to the county level. The latest inventory is for 2002, and a detailed database is available for download at http://www.epa.gov/ttn/chief/net/2002inventory.html. Figure A-1 shows the New York City (NYC) Metro $PM_{2.5}$ Nonattainment Area, which is the area in and around New York City that has been designated by EPA as being in nonattainment for $PM_{2.5}$. It includes the five boroughs of New York City as well as five surrounding counties in New York State (Nassau, Suffolk, Westchester, Orange, and Rockland), ten counties in northern New Jersey (Bergen, Essex, Hudson, Mercer, Middlesex, Monmouth, Morris, Passaic, Somerset, and Union) and two counties in southern Connecticut (Fairfield and New Haven).[1]

Figure A-1. New York City PM2.5 Nonattainment Area.

Source: EPA 2006b

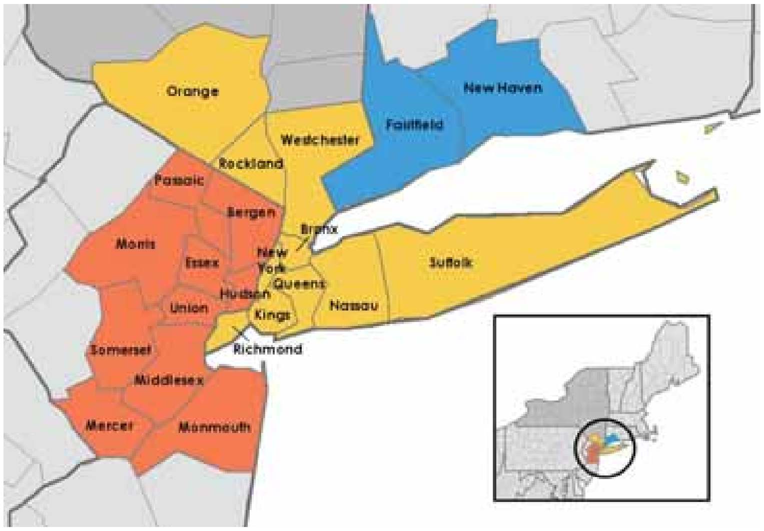

Table A-1 to Table A-4 below show details of the primary $PM_{2.5}$ portion of the inventory for New York State and select areas within the State. All of the data shown in to Table A-4 were taken from the National Emissions Inventory database downloaded from the above address.

Table A-1 includes high-level highlights that illustrate the major differences in $PM_{2.5}$ sources geographically. Table A-2 includes a detailed inventory for New York State as a whole, as well as for the New York City $PM_{2.5}$ Nonattainment Area,[43] Erie County (which includes the City of Buffalo), and Monroe County (which includes the city of Rochester). Figure A-1 shows the designated three-state nonattainment area for $PM_{2.5}$. Table A-3 includes a detailed inventory for three representative rural counties in New York: Greene County, Lewis County, and Cattaraugus County. As shown in Figure A-2, these counties are in the lower Hudson Valley, north central New York, and western New York, respectively. The year 2000 population density of these counties ranged from 21 to 76

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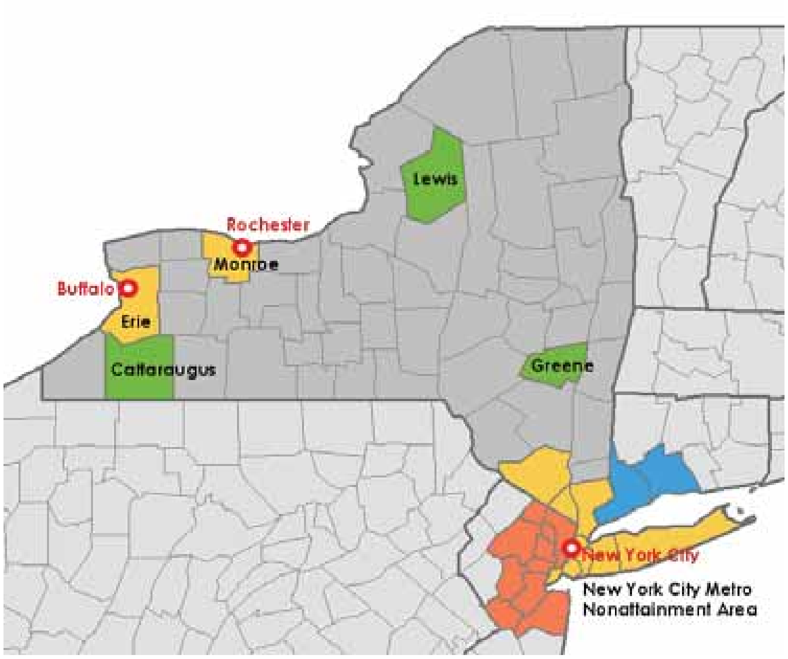
[43] EPA Green Book Web site (EPA 2006b).

persons per square mile (ppsm), compared to 401 ppsm for the state as a whole, 66,946 ppsm in Manhattan, 910 ppsm in Erie County, and 1,115 ppsm in Monroe County.[44] Table A-4 breaks out the emissions inventory for the New York City Metro $PM_{2.5}$ Nonattainment Area into the portions attributable to the New York, New Jersey, and Connecticut counties that it comprises.

Figure A-2. Counties Included in Table A-1 to Table A-4.



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[44] Data on county population density is from the U.S. Census Bureau (http://quickfacts.census.gov/qfd/states/36/36029.html).

Table A-1. Highlights of 2002 $PM_{2.5}$ Inventory, New York State and Select Areas.

	NYS		NYC METRO PM2.5 NONATTAINMENT AREA (1)		OTHER NYS URBAN AREAS				TYPICAL NYS RURAL COUNTIES					
	All Counties				ERIE County (Buffalo)		MONROE County (Rochester)		GREENE COUNTY		LEWIS COUNTY		CATTARAUGUS COUNTY	
	Tons/yr	% of Total	Tons/yr	% of Total	Tons/yr	% of Total	Tons/yr	% of Total	Tons/yr	% of Total	Tons/yr	% of Total	Tons/yr	% of Total
ONROAD	5,898	3.8%	4,689	***9.5%***	411	5.4%	282	4.3%	41	2.7%	13	0.5%	50	1.7%
NONROAD	11,965	7.6%	9,463	***19.2%***	475	6.2%	560	8.6%	33	2.2%	49	2.0%	66	2.2%
STATIONARY	139,032	88.6%	35,179	71.3%	6,716	88.3%	5,676	87.1%	1,421	***95.1%***	2,361	***97.4%***	2,878	***96.1%***
TOTAL	**156,895**		**49,331**		**7,602**		**6,518**		**1,494**		**2,423**		**2,995**	
ONROAD Car & Light Truck	2,131	1.4%	1,866	***3.8%***	143	1.9%	122	1.9%	12	0.8%	4	0.2%	7	0.2%
ONROAD Diesel Truck & Bus	3,095	2.0%	2,419	***4.9%***	221	2.9%	129	2.0%	24	1.6%	8	0.3%	37	1.2%
NONROAD Agricultural	714	0.5%	42	0.1%	15	0.2%	14	0.2%	3	0.2%	16	0.7%	15	0.5%
NONROAD Construction	3,681	2.3%	4,030	***8.2%***	165	2.2%	144	2.2%	8	0.5%	10	0.4%	29	1.0%
NONROAD Lawn & Garden	998	0.6%	1,377	***2.8%***	62	0.8%	72	1.1%	1	0.1%	1	0.0%	3	0.1%
NONROAD Marine	1,791	1.1%	273	0.6%	27	0.4%	2	0.0%	3	0.2%	-	0.0%	-	0.0%
NONROAD Rail	350	0.2%	151	0.3%	34	0.5%	19	0.3%	4	0.3%	0	0.0%	2	0.1%
NONROAD Recreational	2,377	1.5%	1,036	2.1%	57	0.7%	204	***3.1%***	11	0.8%	18	0.7%	8	0.3%
STATIONARY Residential/Wood	60,584	***38.6%***	6,610	***13.4%***	1,675	22.0%	1,019	15.6%	686	***45.9%***	1,881	***77.6%***	2,128	***71.1%***
STATIONARY Residential/Oil	1,285	0.8%	1,426	2.9%	4	0.1%	7	0.1%	7	0.5%	3	0.1%	2	0.1%
STATIONARY Electric Generation	13,395	8.5%	7,013	14.2%	2,080	***27.4%***	1,666	***25.6%***	4	0.2%	2	0.1%	6	0.2%
STATIONARY Charbroiling	4,099	2.6%	4,598	***9.3%***	188	2.5%	144	2.2%	10	0.7%	4	0.2%	23	0.8%

(Table A-1 Cont.) Highlights of 2002 $PM_{2.5}$ Inventory, New York State and Select Areas.

	NYC Metro PM 2.5 Nonattainment Area (NAA)					
	NY COUNTIES (1)		**NJ COUNTIES (1)**		**CT COUNTIES (1)**	
	Tons/yr	**% of Total**	**Tons/yr**	**% of Total**	**Tons/yr**	**% of Total**
ONROAD	2,495	9.0%	1,555	10.6%	639	8.9%
NONROAD	4,918	***17.8%***	3,205	***21.9%***	1,341	***18.8%***
STATIONARY	20,167	73.1%	9,846	67.4%	5,166	72.3%
TOTAL	**27,580**		**14,606**		**7,145**	
ONROAD Car & Light Truck	994	***3.6%***	665	***4.6%***	207	***2.9%***
ONROAD Diesel Truck & Bus	1,223	***4.4%***	805	***5.5%***	391	***5.5%***
NONROAD Agricultural	16	0.1%	23	0.2%	3	0.0%
NONROAD Construction	2,405	***8.7%***	1,162	***8.0%***	463	6.5%
NONROAD Lawn & Garden	608	2.2%	593	***4.1%***	176	2.5%
NONROAD Marine	65	0.2%	30	0.2%	178	***2.5%***
NONROAD Rail	33	0.1%	115	0.8%	3	0.0%
NONROAD Recreational	456	1.7%	251	1.7%	329	***4.6%***
STATIONARY Residential/Wood	1,555	5.6%	2,632	***18.0%***	2,423	***33.9%***
STATIONARY Residential/Oil	962	3.5%	182	1.2%	282	3.9%
STATIONARY Electric Generation	4,108	14.9%	2,614	17.9%	290	4.1%
STATIONARY Charbroiling	2,709	***9.8%***	1,479	***10.1%***	410	5.7%

Table A-2. 2002 $PM_{2.5}$ Inventory, New York State and Select NYS Urban Areas.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5											
			NYS			NYC METRO PM2.5			OTHER NYS URBAN AREAS					
			All Counties			NONATTAINMENT AREA (1)			ERIE County (Buffalo)			MONROE County (Rochester)		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
ONROAD VEHICLES	Light Duty Gasoline Car (LDGC)	Exhaust	542.1	3.0%	0.3%	504.3	3.6%	1.0%	34.4	3.9%	0.5%	30.6	3.6%	0.5%
		Brake Wear	432.1	2.4%	0.3%	447.3	3.2%	0.9%	25.2	2.8%	0.3%	20.7	2.5%	0.3%
		Tire Wear	163.0	0.9%	0.1%	168.4	1.2%	0.3%	9.5	1.1%	0.1%	7.8	0.9%	0.1%
		Sub-Total	*1,137.2*	*6.4%*	*0.7%*	*1,120.1*	*7.9%*	*2.3%*	*69.0*	*7.8%*	*0.9%*	*59.1*	*7.0%*	*0.9%*
	Light Duty Gasoline Truck (LDGT)	Exhaust	544.3	3.0%	0.3%	373.0	2.6%	0.8%	43.3	4.9%	0.6%	37.3	4.4%	0.6%
		Brake Wear	326.0	1.8%	0.2%	270.9	1.9%	0.5%	22.4	2.5%	0.3%	18.8	2.2%	0.3%
		Tire Wear	123.2	0.7%	0.1%	101.9	0.7%	0.2%	8.4	1.0%	0.1%	7.1	0.8%	0.1%
		Sub-Total	*993.4*	*5.6%*	*0.6%*	*745.7*	*5.3%*	*1.5%*	*74.1*	*8.4%*	*1.0%*	*63.2*	*7.5%*	*1.0%*
	Heavy Duty Gasoline Truck (HDGV)	Exhaust	388.7	2.2%	0.2%	229.2	1.6%	0.5%	27.6	3.1%	0.4%	16.4	2.0%	0.3%
		Brake Wear	27.6	0.2%	0.0%	18.3	0.1%	0.0%	1.9	0.2%	0.0%	1.1	0.1%	0.0%
		Tire Wear	16.3	0.1%	0.0%	9.5	0.1%	0.0%	1.1	0.1%	0.0%	0.7	0.1%	0.0%
		Sub-Total	*432.6*	*2.4%*	*0.3%*	*256.9*	*1.8%*	*0.5%*	*30.6*	*3.5%*	*0.4%*	*18.2*	*2.2%*	*0.3%*
	Motorcycle (MC)	Exhaust	8.4	0.0%	0.0%	7.6	0.1%	0.0%	0.6	0.1%	0.0%	0.6	0.1%	0.0%
		Brake Wear	2.0	0.0%	0.0%	2.4	0.0%	0.0%	0.2	0.0%	0.0%	0.2	0.0%	0.0%
		Tire Wear	0.2	0.0%	0.0%	0.4	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*10.6*	*0.1%*	*0.0%*	*10.4*	*0.1%*	*0.0%*	*0.8*	*0.1%*	*0.0%*	*0.8*	*0.1%*	*0.0%*
	Light Duty Diesel Car (LDDV)	Exhaust	52.2	0.3%	0.0%	57.6	0.4%	0.1%	2.3	0.3%	0.0%	1.8	0.2%	0.0%
		Brake Wear	0.9	0.0%	0.0%	1.3	0.0%	0.0%	0.1	0.0%	0.0%	0.0	0.0%	0.0%
		Tire Wear	1.0	0.0%	0.0%	0.5	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*54.1*	*0.3%*	*0.0%*	*59.4*	*0.4%*	*0.1%*	*2.4*	*0.3%*	*0.0%*	*1.9*	*0.2%*	*0.0%*
	Light Duty Diesel Truck (LDDT)	Exhaust	166.8	0.9%	0.1%	74.2	0.5%	0.2%	12.1	1.4%	0.2%	9.5	1.1%	0.1%
		Brake Wear	6.0	0.0%	0.0%	2.8	0.0%	0.0%	0.5	0.1%	0.0%	0.4	0.0%	0.0%
		Tire Wear	2.0	0.0%	0.0%	1.0	0.0%	0.0%	0.2	0.0%	0.0%	0.1	0.0%	0.0%
		Sub-Total	*174.8*	*1.0%*	*0.1%*	*78.0*	*0.6%*	*0.2%*	*12.8*	*1.4%*	*0.2%*	*9.9*	*1.2%*	*0.2%*
	Heavy Duty Diesel Truck (HDDT)	Exhaust	1,807.1	10.1%	1.2%	1,739.4	12.3%	3.5%	126.0	14.2%	1.7%	78.1	9.3%	1.2%
		Brake Wear	20.6	0.1%	0.0%	24.5	0.2%	0.0%	1.5	0.2%	0.0%	0.9	0.1%	0.0%
		Tire Wear	25.2	0.1%	0.0%	28.5	0.2%	0.1%	1.8	0.2%	0.0%	1.1	0.1%	0.0%
		Sub-Total	*1,853.0*	*10.4%*	*1.2%*	*1,792.3*	*12.7%*	*3.6%*	*129.2*	*14.6%*	*1.7%*	*80.0*	*9.5%*	*1.2%*
	Heavy Duty Diesel Bus (HDDB)	Exhaust	1,234.4	6.9%	0.8%	621.9	4.4%	1.3%	91.6	10.3%	1.2%	48.5	5.8%	0.7%
		Brake Wear	5.1	0.0%	0.0%	2.8	0.0%	0.0%	0.4	0.0%	0.0%	0.2	0.0%	0.0%
		Tire Wear	2.6	0.0%	0.0%	1.6	0.0%	0.0%	0.2	0.0%	0.0%	0.1	0.0%	0.0%
		Sub-Total	*1,242.1*	*7.0%*	*0.8%*	*626.3*	*4.4%*	*1.3%*	*92.2*	*10.4%*	*1.2%*	*48.9*	*5.8%*	*0.8%*
		TOTAL EXHAUST	**4,743.9**	*26.6%*	*3.0%*	**3,607.2**	*25.5%*	*7.3%*	**337.9**	*38.2%*	*4.4%*	**222.9**	*26.5%*	*3.4%*
		TOTAL BRAKE	**820.3**	*4.6%*	*0.5%*	**770.3**	*5.4%*	*1.6%*	**51.9**	*5.9%*	*0.7%*	**42.2**	*5.0%*	*0.6%*
		TOTAL TIRE	**333.5**	*1.9%*	*0.2%*	**311.7**	*2.2%*	*0.6%*	**21.3**	*2.4%*	*0.3%*	**16.9**	*2.0%*	*0.3%*
		TOTAL ON-ROAD	**5,897.7**	**33%**	**3.8%**	**4,689.2**	**33%**	**9.5%**	**411.1**	**46%**	**5.4%**	**282.0**	**33%**	**4.3%**

Table A-2. (Cont.) 2002 $PM_{2.5}$ Inventory, New York State and Select NYS Urban Areas.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5											
			NYS			NYC METRO PM2.5 NONATTAINMENT AREA (1)			OTHER NYS URBAN AREAS					
			All Counties						ERIE County (Buffalo)			MONROE County (Rochester)		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
NONROAD EQUIPMENT	Aircraft	Jet Fuel	376.2	2.1%	0.2%	322.1	2.3%	0.7%	26.4	3.0%	0.3%	19.5	2.3%	0.3%
		Sub-Total	*376.2*	*2.1%*	*0.2%*	*322.1*	*2.3%*	*0.7%*	*26.4*	*3.0%*	*0.3%*	*19.5*	*2.3%*	*0.3%*
	Airport GSE	Gasoline	0.2	0.0%	0.0%	230.9	1.6%	0.5%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	57.0	0.3%	0.0%	131.9	0.9%	0.3%	1.1	0.1%	0.0%	0.9	0.1%	0.0%
		LPG	0.1	0.0%	0.0%	0.2	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.1	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*57.3*	*0.3%*	*0.0%*	*363.0*	*2.6%*	*0.7%*	*1.1*	*0.1%*	*0.0%*	*0.9*	*0.1%*	*0.0%*
	Agricultural	Gasoline	1.3	0.0%	0.0%	0.1	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	712.5	4.0%	0.5%	41.5	0.3%	0.1%	14.8	1.7%	0.2%	13.9	1.6%	0.2%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.1	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*713.9*	*4.0%*	*0.5%*	*41.6*	*0.3%*	*0.1%*	*14.8*	*1.7%*	*0.2%*	*13.9*	*1.7%*	*0.2%*
	Commercial	Gasoline	173.6	1.0%	0.1%	210.9	1.5%	0.4%	7.8	0.9%	0.1%	5.4	0.6%	0.1%
		Diesel	746.3	4.2%	0.5%	912.0	6.4%	1.8%	33.7	3.8%	0.4%	23.2	2.8%	0.4%
		LPG	5.3	0.0%	0.0%	6.5	0.0%	0.0%	0.2	0.0%	0.0%	0.2	0.0%	0.0%
		CNG	3.7	0.0%	0.0%	4.4	0.0%	0.0%	0.2	0.0%	0.0%	0.1	0.0%	0.0%
		Sub-Total	*928.8*	*5.2%*	*0.6%*	*1,133.8*	*8.0%*	*2.3%*	*42.0*	*4.7%*	*0.6%*	*28.9*	*3.4%*	*0.4%*
	Construction & Mining	Gasoline	108.8	0.6%	0.1%	118.3	0.8%	0.2%	4.9	0.5%	0.1%	4.3	0.5%	0.1%
		Diesel	3,571.3	20.0%	2.3%	3,910.6	27.6%	7.9%	159.9	18.0%	2.1%	139.7	16.6%	2.1%
		LPG	0.8	0.0%	0.0%	0.8	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*3,680.8*	*20.6%*	*2.3%*	*4,029.7*	*28.5%*	*8.2%*	*164.8*	*18.6%*	*2.2%*	*144.0*	*17.1%*	*2.2%*
	Industrial	Gasoline	7.2	0.0%	0.0%	8.0	0.1%	0.0%	0.6	0.1%	0.0%	0.8	0.1%	0.0%
		Diesel	604.8	3.4%	0.4%	660.8	4.7%	1.3%	40.1	4.5%	0.5%	47.9	5.7%	0.7%
		LPG	56.1	0.3%	0.0%	62.8	0.4%	0.1%	4.3	0.5%	0.1%	6.3	0.8%	0.1%
		CNG	4.2	0.0%	0.0%	4.6	0.0%	0.0%	0.3	0.0%	0.0%	0.5	0.1%	0.0%
		Sub-Total	*672.3*	*3.8%*	*0.4%*	*736.2*	*5.2%*	*1.5%*	*45.2*	*5.1%*	*0.6%*	*55.5*	*6.6%*	*0.9%*
	Lawn & Garden	Gasoline	824.7	4.6%	0.5%	1,118.7	7.9%	2.3%	52.3	5.9%	0.7%	58.1	6.9%	0.9%
		Diesel	172.9	1.0%	0.1%	257.3	1.8%	0.5%	9.8	1.1%	0.1%	13.7	1.6%	0.2%
		LPG	0.4	0.0%	0.0%	0.6	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*998.0*	*5.6%*	*0.6%*	*1,376.6*	*9.7%*	*2.8%*	*62.1*	*7.0%*	*0.8%*	*71.8*	*8.5%*	*1.1%*

Table A-2 (Cont.) 2002 $PM_{2.5}$ Inventory, New York State and Select NYS Urban Areas.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5											
			NYS			NYC METRO PM2.5			OTHER NYS URBAN AREAS					
			All Counties			NONATTAINMENT AREA (1)			ERIE County (Buffalo)			MONROE County (Rochester)		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
NONROAD EQUIPMENT (cont)	Logging	Gasoline	5.7	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	14.0	0.1%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*19.7*	*0.1%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*
	Marine	Gasoline	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	1,791.3	10.0%	1.1%	273.1	1.9%	0.6%	27.2	3.1%	0.4%	2.3	0.3%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*1,791.3*	*10.0%*	*1.1%*	*273.1*	*1.9%*	*0.6%*	*27.2*	*3.1%*	*0.4%*	*2.3*	*0.3%*	*0.0%*
	Rail	Gasoline	0.1	0.0%	0.0%	0.1	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	349.5	2.0%	0.2%	150.7	1.1%	0.3%	34.4	3.9%	0.5%	19.1	2.3%	0.3%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*349.6*	*2.0%*	*0.2%*	*150.8*	*1.1%*	*0.3%*	*34.4*	*3.9%*	*0.5%*	*19.1*	*2.3%*	*0.3%*
	Recreational	Gasoline	2,321.7	13.0%	1.5%	1,006.3	7.1%	2.0%	54.9	6.2%	0.7%	200.5	23.8%	3.1%
		Diesel	55.5	0.3%	0.0%	29.6	0.2%	0.1%	1.7	0.2%	0.0%	3.5	0.4%	0.1%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*2,377.2*	*13.3%*	*1.5%*	*1,035.9*	*7.3%*	*2.1%*	*56.6*	*6.4%*	*0.7%*	*204.0*	*24.2%*	*3.1%*
		TOTAL JET FUEL	**376.2**	***2.1%***	***0.2%***	**322.1**	***2.3%***	***0.7%***	**26.4**	***3.0%***	***0.3%***	**19.5**	***2.3%***	***0.3%***
		TOTAL GASOLINE	**3,443.3**	***19.3%***	***2.2%***	**2,693.2**	***19.0%***	***5.5%***	**120.6**	***13.6%***	***1.6%***	**269.1**	***32.0%***	***4.1%***
		TOTAL DIESEL	**8,075.1**	***45.2%***	***5.1%***	**6,367.5**	***45.0%***	***12.9%***	**322.5**	***36.4%***	***4.2%***	**264.3**	***31.4%***	***4.1%***
		TOTAL LPG	**62.7**	***0.4%***	***0.0%***	**70.9**	***0.5%***	***0.1%***	**4.6**	***0.5%***	***0.1%***	**6.5**	***0.8%***	***0.1%***
		TOTAL CNG	**7.9**	***0.0%***	***0.0%***	**9.1**	***0.1%***	***0.0%***	**0.5**	***0.1%***	***0.0%***	**0.6**	***0.1%***	***0.0%***
		TOTAL NON-ROAD	**11,965.1**	***67%***	***7.6%***	**9,462.9**	***67%***	***19.2%***	**474.5**	***54%***	***6.2%***	**560.1**	***67%***	***8.6%***

Table A-2 (Cont.) 2002 $PM_{2.5}$ Inventory, New York State and Select NYS Urban Areas.

SECTOR	Type	Source	PRIMARY PM2.5											
			NYS			NYC METRO PM2.5 NONATTAIN AREA (1)			OTHER NYS URBAN AREAS					
			All Counties						ERIE County (Buffalo)			MONROE County (Rochester)		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
EXTERNAL FUEL COMBUSTION	Residential	Coal	42.8	0.03%	0.03%	17.4	0.05%	0.04%	0.7	0.01%	0.01%	0.4	0.01%	0.01%
		Bagasse	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Distillate Oil	1,285.0	0.92%	0.82%	1,426.3	4.05%	2.89%	4.3	0.06%	0.06%	6.8	0.12%	0.10%
		Lignite	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Liquid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	2.0	0.00%	0.00%	2.4	0.01%	0.00%	0.1	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	80.6	0.06%	0.05%	87.0	0.25%	0.18%	7.6	0.11%	0.10%	5.1	0.09%	0.08%
		Petroleum Coke	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Solid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood/Bark Waste	60,583.6	43.58%	38.61%	6,610.1	18.79%	13.40%	1,675.3	24.94%	22.04%	1,018.8	17.95%	15.63%
		Sub-Total	*61,993.9*	*44.6%*	*39.5%*	*8,143*	*23.1%*	*16.5%*	*1,688*	*25.1%*	*22.2%*	*1,031*	*18.2%*	*15.8%*
	Electric Generation	Coal	10,830.2	7.79%	6.90%	4,356.8	12.38%	8.83%	2,061.3	30.69%	27.12%	1,666.2	29.35%	25.56%
		Bagasse	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Distillate Oil	0.9	0.00%	0.00%	3.1	0.01%	0.01%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Lignite	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Liquid Waste	1.7	0.00%	0.00%	0.2	0.00%	0.00%	0.7	0.01%	0.01%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	764.6	0.55%	0.49%	830.4	2.36%	1.68%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Petroleum Coke	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.3	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	1,797.6	1.29%	1.15%	1,822.0	5.18%	3.69%	17.8	0.26%	0.23%	0.0	0.00%	0.00%
		Solid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood/Bark Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*13,395.0*	*9.6%*	*8.5%*	*7,013*	*19.9%*	*14.2%*	*2,080*	*31.0%*	*27.4%*	*1,666*	*29.4%*	*25.6%*
	Industrial / Commercial / Institutional	Coal	5,650.1	4.06%	3.60%	2,219.3	6.31%	4.50%	447.9	6.67%	5.89%	1,007.2	17.74%	15.45%
		Bagasse	0.0	0.00%	0.00%	0.1	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Distillate Oil	627.1	0.45%	0.40%	635.4	1.81%	1.29%	32.8	0.49%	0.43%	28.7	0.51%	0.44%
		Lignite	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Liquid Waste	20.5	0.01%	0.01%	0.0	0.00%	0.00%	4.0	0.06%	0.05%	0.0	0.00%	0.00%
		LPG	0.8	0.00%	0.00%	2.6	0.01%	0.01%	0.1	0.00%	0.00%	0.1	0.00%	0.00%
		Natural Gas	111.1	0.08%	0.07%	159.9	0.45%	0.32%	6.4	0.10%	0.08%	6.6	0.12%	0.10%
		Petroleum Coke	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	3.7	0.00%	0.00%	57.2	0.16%	0.12%	0.1	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	578.9	0.42%	0.37%	404.8	1.15%	0.82%	48.0	0.71%	0.63%	25.2	0.44%	0.39%
		Solid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood/Bark Waste	144.9	0.10%	0.09%	65.2	0.19%	0.13%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*7,137.0*	*5.1%*	*4.5%*	*3,545*	*10.1%*	*7.2%*	*539*	*8.0%*	*7.1%*	*1,068*	*18.8%*	*16.4%*

Table A-2 (Cont.) 2002 $PM_{2.5}$ Inventory, New York State and Select NYS Urban Areas.

SECTOR	Type	Source	PRIMARY PM2.5											
			NYS			NYC METRO PM2.5 NONATTAIN AREA (1)			OTHER NYS URBAN AREAS					
			All Counties						ERIE County (Buffalo)			MONROE County (Rochester)		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
EFC TOTAL		TOTAL COAL	16,523.0	11.9%	10.5%	6,593.5	35.3%	18.7%	2,509.8	58.3%	37.4%	2,673.7	71.0%	47.1%
		TOTAL BAGASSE	0.0	0.0%	0.0%	0.1	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL DISTILLATE OIL	1,912.9	1.4%	1.2%	2,064.7	11.0%	5.9%	37.1	0.9%	0.6%	35.5	0.9%	0.6%
		TOTAL LIGNITE	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL LIQUID WASTE	22.2	0.0%	0.0%	0.2	0.0%	0.0%	4.7	0.1%	0.1%	0.0	0.0%	0.0%
		TOTAL LPG	2.8	0.0%	0.0%	5.0	0.0%	0.0%	0.1	0.0%	0.0%	0.1	0.0%	0.0%
		TOTAL NATURAL GAS	956.3	0.7%	0.6%	1,077.3	5.8%	3.1%	14.0	0.3%	0.2%	11.7	0.3%	0.2%
		TOTAL PETROLEUM COKE	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL PROCESS GAS	3.7	0.0%	0.0%	57.5	0.3%	0.2%	0.1	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL RESIDUAL OIL	2,376.5	1.7%	1.5%	2,226.8	11.9%	6.3%	65.8	1.5%	1.0%	25.2	0.7%	0.4%
		TOTAL SOLID WASTE	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL WOOD/BARK WASTE	60,728.4	43.7%	38.7%	6,675.3	35.7%	19.0%	1,675.3	38.9%	24.9%	1,018.8	27.1%	17.9%
		TOTAL EXT. COMBUSTION	82,526.0	59%	53%	18,700.4	100%	38%	4,306.9	100%	57%	3,765.1	100%	58%
OTHER FUEL COMBUSTION	Internal Combustion	Diesel	21.6	0.02%	0.01%	26.3	0.07%	0.05%	0.0	0.00%	0.00%	0.1	0.00%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Kerosene	37.4	0.03%	0.02%	80.7	0.23%	0.16%	1.9	0.03%	0.02%	1.6	0.03%	0.02%
		LFG	6.9	0.00%	0.00%	6.9	0.02%	0.01%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.1	0.00%	0.00%	0.0	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	7.9	0.01%	0.01%	16.6	0.05%	0.03%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	11.6	0.01%	0.01%	15.8	0.04%	0.03%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.0	0.00%	0.00%	5.0	0.07%	0.07%	4.1	0.07%	0.06%
		Sub-Total	*86.4*	*0.1%*	*0.06%*	*146.3*	*0.4%*	*0.30%*	*7.0*	*0.1%*	*0.09%*	*5.8*	*0.1%*	*0.09%*
	Turbine	Diesel	25.8	0.02%	0.02%	42.3	0.12%	0.09%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Kerosene	28.3	0.02%	0.02%	29.6	0.08%	0.06%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LFG	5.5	0.00%	0.00%	30.5	0.09%	0.06%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	37.7	0.03%	0.02%	160.2	0.46%	0.32%	0.1	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*97.4*	*0.1%*	*0.06%*	*262.7*	*0.7%*	*0.53%*	*0.1*	*0.0%*	*0.00%*	*0.0*	*0.0%*	*0.00%*
	Industrial Space Heating	Diesel	18.0	0.01%	0.01%	17.7	0.05%	0.04%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Gasoline	0.1	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Kerosene	5.9	0.00%	0.00%	8.1	0.02%	0.02%	0.2	0.00%	0.00%	0.0	0.00%	0.00%
		LFG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	61.6	0.04%	0.04%	13.2	0.04%	0.03%	6.9	0.10%	0.09%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	6.7	0.02%	0.01%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	2.8	0.01%	0.01%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Waste Oil	0.0	0.0	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*85.6*	*0.1%*	*0.06%*	*48.4*	*0.1%*	*0.10%*	*7.1*	*0.1%*	*0.09%*	*0.0*	*0.0%*	*0.00%*

Table A-2 (Cont.) 2002 PM$_{2.5}$ Inventory, New York State and Select NYS Urban Areas.

SECTOR	Type	Source	PRIMARY PM2.5											
			NYS			NYC METRO PM2.5 NONATTAIN AREA (1)			OTHER NYS URBAN AREAS					
			All Counties						ERIE County (Buffalo)			MONROE County (Rochester)		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
OFC TOTAL		TOTAL DIESEL	65.4	0.0%	0.0%	86	18.9%	0.2%	0.0	0.0%	0.0%	0.1	0.0%	0.0%
		TOTAL GASOLINE	0.1	0.0%	0.0%	0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL KEROSENE	71.5	0.1%	0.0%	118	25.9%	0.3%	2.1	0.0%	0.0%	1.6	0.0%	0.0%
		TOTAL LFG	12.5	0.0%	0.0%	37	8.2%	0.1%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL LPG	0.0	0.0%	0.0%	0	0.0%	0.0%	0.1	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL METHANOL	0.0	0.0%	0.0%	0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTOAL NATURAL GAS	107.2	0.1%	0.1%	190	41.5%	0.5%	7.0	0.1%	0.1%	0.0	0.0%	0.0%
		TOTAL PROCESS GAS	11.6	0.0%	0.0%	23	4.9%	0.1%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL RESIDUAL OIL	0.0	0.0%	0.0%	3	0.6%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL WASTE OIL	0.0	0.0%	0.0%	0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL WOOD	0.0	0.0%	0.0%	0	0.0%	0.0%	5.0	0.1%	0.1%	4.1	0.1%	0.1%
		TOTAL OTHER COMBUST.	268.4	0%	0.2%	457	100%	1.3%	14.2	0%	0.2%	5.8	0%	0.1%
OTHER STATIONARY	Food and Kindred Products: SIC20	Conveyorized Charbroiling	482.8	0.35%	0.31%	350.7	1.00%	0.71%	24.1	0.36%	0.32%	18.2	0.32%	0.28%
		Under-fired Charbroiling	3,616.1	2.60%	2.30%	4,247.2	12.07%	8.61%	163.7	2.44%	2.15%	125.6	2.21%	1.93%
		Flat Griddle Frying	745.0	0.54%	0.47%	564.8	1.61%	1.14%	34.4	0.51%	0.45%	26.9	0.47%	0.41%
		Clamshell Griddle Frying	50.5	0.04%	0.03%	38.1	0.11%	0.08%	2.4	0.04%	0.03%	1.7	0.03%	0.03%
		Meat Products	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Other	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*4,894.3*	*3.5%*	*3.1%*	*5,200.7*	*14.8%*	*10.5%*	*224.6*	*3.3%*	*3.0%*	*172.4*	*3.0%*	*2.6%*
	All Other Sources	Miscellaneous	51,343.3	36.93%	32.72%	10,820.7	30.76%	21.93%	2,170.6	32.32%	28.55%	1,733.1	30.53%	26.59%
		TOTAL OTHER STAT. SOURCES	56,238	40%	36%	16,021	46%	32%	2,395	36%	32%	1,906	34%	29%
TOTAL STATIONARY INVENTORY			139,032	100%	89%	35,179	100%	71%	6,716	100%	88%	5,676	100%	87%
TOTAL STATIONARY & MOBILE INVENTORY			156,895		100%	49,331		100%	7,602		100%	6,518		100%

Table A-3. 2002 $PM_{2.5}$ Inventory, Select Rural Areas in New York State.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5								
			TYPICAL NYS RURAL COUNTIES								
			GREENE COUNTY			LEWIS COUNTY			CATTARAUGUS COUNTY		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
ONROAD VEHICLES	Light Duty Gasoline Car (LDGC)	Exhaust	2.7	3.7%	0.2%	0.7	1.2%	0.0%	2.9	2.5%	0.1%
		Brake Wear	2.0	2.7%	0.1%	0.5	0.9%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.8	1.0%	0.1%	0.2	0.3%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*5.5*	*7.4%*	*0.4%*	*1.5*	*2.4%*	*0.1%*	*2.9*	*2.5%*	*0.1%*
	Light Duty Gasoline Truck (LDGT)	Exhaust	3.9	5.3%	0.3%	1.5	2.4%	0.1%	4.0	3.5%	0.1%
		Brake Wear	2.0	2.7%	0.1%	0.8	1.3%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.8	1.1%	0.1%	0.3	0.5%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*6.7*	*9.1%*	*0.4%*	*2.6*	*4.1%*	*0.1%*	*4.0*	*3.5%*	*0.1%*
	Heavy Duty Gasoline Truck (HDGV)	Exhaust	3.1	4.2%	0.2%	0.9	1.4%	0.0%	4.8	4.1%	0.2%
		Brake Wear	0.2	0.3%	0.0%	0.1	0.1%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.1	0.2%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*3.4*	*4.7%*	*0.2%*	*0.9*	*1.5%*	*0.0%*	*4.8*	*4.1%*	*0.2%*
	Motorcycle (MC)	Exhaust	0.1	0.1%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Brake Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*0.1*	*0.1%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*
	Light Duty Diesel Car (LDDV)	Exhaust	0.2	0.2%	0.0%	0.1	0.1%	0.0%	0.2	0.2%	0.0%
		Brake Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*0.2*	*0.2%*	*0.0%*	*0.1*	*0.1%*	*0.0%*	*0.2*	*0.2%*	*0.0%*
	Light Duty Diesel Truck (LDDT)	Exhaust	1.1	1.5%	0.1%	0.5	0.7%	0.0%	1.3	1.1%	0.0%
		Brake Wear	0.0	0.1%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*1.2*	*1.6%*	*0.1%*	*0.5*	*0.8%*	*0.0%*	*1.3*	*1.1%*	*0.0%*
	Heavy Duty Diesel Truck (HDDT)	Exhaust	15.9	21.6%	1.1%	4.7	7.6%	0.2%	21.6	18.6%	0.7%
		Brake Wear	0.1	0.2%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.2	0.3%	0.0%	0.1	0.1%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*16.2*	*22.1%*	*1.1%*	*4.8*	*7.7%*	*0.2%*	*21.6*	*18.6%*	*0.7%*
	Heavy Duty Diesel Bus (HDDB)	Exhaust	7.5	10.2%	0.5%	2.8	4.6%	0.1%	15.7	13.5%	0.5%
		Brake Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Tire Wear	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*7.5*	*10.3%*	*0.5%*	*2.9*	*4.6%*	*0.1%*	*15.7*	*13.5%*	*0.5%*
		TOTAL EXHAUST	**34.4**	*46.8%*	*2.3%*	**11.2**	*18.0%*	*0.5%*	**50.5**	*43.4%*	*1.7%*
		TOTAL BRAKE	**4.4**	*6.0%*	*0.3%*	**1.4**	*2.3%*	*0.1%*	**0.0**	*0.0%*	*0.0%*
		TOTAL TIRE	**1.9**	*2.6%*	*0.1%*	**0.6**	*0.9%*	*0.0%*	**0.0**	*0.0%*	*0.0%*
		TOTAL ON-ROAD	**40.7**	***55%***	***2.7%***	**13.2**	***21%***	***0.5%***	**50.5**	***43%***	***1.7%***

Table A-3 (Cont.) 2002 $PM_{2.5}$ Inventory, Select Rural Areas in New York State.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5								
			TYPICAL NYS RURAL COUNTIES								
			GREENE COUNTY			LEWIS COUNTY			CATTARAUGUS COUNTY		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
NONROAD EQUIPMENT	Aircraft	Jet Fuel	0.1	0.1%	0.0%	0.0	0.0%	0.0%	1.1	0.9%	0.0%
		Sub-Total	*0.1*	*0.1%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*1.1*	*0.9%*	*0.0%*
	Airport GSE	Gasoline	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*0.0*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*
	Agricultural	Gasoline	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	3.4	4.7%	0.2%	16.3	26.2%	0.7%	15.1	13.0%	0.5%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*3.4*	*4.7%*	*0.2%*	*16.3*	*26.3%*	*0.7%*	*15.1*	*13.0%*	*0.5%*
	Commercial	Gasoline	0.2	0.3%	0.0%	0.1	0.2%	0.0%	0.5	0.4%	0.0%
		Diesel	0.9	1.2%	0.1%	0.5	0.8%	0.0%	2.1	1.8%	0.1%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*1.1*	*1.5%*	*0.1%*	*0.7*	*1.1%*	*0.0%*	*2.6*	*2.2%*	*0.1%*
	Construction & Mining	Gasoline	0.2	0.3%	0.0%	0.3	0.5%	0.0%	0.9	0.7%	0.0%
		Diesel	7.3	9.9%	0.5%	9.2	14.9%	0.4%	28.6	24.6%	1.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*7.5*	*10.2%*	*0.5%*	*9.5*	*15.4%*	*0.4%*	*29.5*	*25.3%*	*1.0%*
	Industrial	Gasoline	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.1	0.1%	0.0%
		Diesel	1.1	1.5%	0.1%	1.0	1.7%	0.0%	4.1	3.5%	0.1%
		LPG	0.1	0.1%	0.0%	0.1	0.2%	0.0%	0.4	0.3%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*1.1*	*1.6%*	*0.1%*	*1.2*	*1.9%*	*0.0%*	*4.6*	*3.9%*	*0.2%*
	Lawn & Garden	Gasoline	1.3	1.7%	0.1%	0.6	1.0%	0.0%	2.7	2.4%	0.1%
		Diesel	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.3	0.3%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*1.3*	*1.7%*	*0.1%*	*0.6*	*1.0%*	*0.0%*	*3.0*	*2.6%*	*0.1%*

Table A-3 (Cont.) 2002 $PM_{2.5}$ Inventory, Select Rural Areas in New York State.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5								
			TYPICAL NYS RURAL COUNTIES								
			GREENE COUNTY			LEWIS COUNTY			CATTARAUGUS COUNTY		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
NONROAD EQUIPMENT (cont)	Logging	Gasoline	0.0	0.0%	0.0%	0.7	1.2%	0.0%	0.0	0.0%	0.0%
		Diesel	0.0	0.0%	0.0%	1.8	2.9%	0.1%	0.0	0.0%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*0.0*	*0.0%*	*0.0%*	*2.5*	*4.1%*	*0.1%*	*0.0*	*0.0%*	*0.0%*
	Marine	Gasoline	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	2.7	3.7%	0.2%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*2.7*	*3.7%*	*0.2%*	*0.0*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*
	Rail	Gasoline	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Diesel	4.1	5.5%	0.3%	0.1	0.1%	0.0%	1.6	1.3%	0.1%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*4.1*	*5.5%*	*0.3%*	*0.1*	*0.1%*	*0.0%*	*1.6*	*1.3%*	*0.1%*
	Recreational	Gasoline	11.1	15.0%	0.7%	17.6	28.5%	0.7%	8.0	6.9%	0.3%
		Diesel	0.4	0.6%	0.0%	0.3	0.5%	0.0%	0.4	0.3%	0.0%
		LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		Sub-Total	*11.5*	*15.6%*	*0.8%*	*18.0*	*29.0%*	*0.7%*	*8.4*	*7.2%*	*0.3%*
		TOTAL JET FUEL	**0.1**	***0.1%***	***0.0%***	**0.0**	***0.0%***	***0.0%***	**1.1**	***0.9%***	***0.0%***
		TOTAL GASOLINE	**12.8**	***17.4%***	***0.9%***	**19.5**	***31.4%***	***0.8%***	**12.2**	***10.5%***	***0.4%***
		TOTAL DIESEL	**19.9**	***27.0%***	***1.3%***	**29.2**	***47.2%***	***1.2%***	**52.0**	***44.8%***	***1.7%***
		TOTAL LPG	**0.1**	***0.1%***	***0.0%***	**0.1**	***0.2%***	***0.0%***	**0.4**	***0.3%***	***0.0%***
		TOTAL CNG	**0.0**	***0.0%***	***0.0%***	**0.0**	***0.0%***	***0.0%***	**0.1**	***0.0%***	***0.0%***
		TOTAL NON-ROAD	**32.8**	***45%***	**2.2%**	**48.8**	***79%***	**2.0%**	**65.8**	***57%***	**2.2%**

Table A-3 (Cont.) 2002 $PM_{2.5}$ Inventory, Select Rural Areas in New York State.

SECTOR	Type	Source	PRIMARY PM2.5								
			TYPICAL NYS RURAL COUNTIES								
			GREENE COUNTY			LEWIS COUNTY			CATTARAUGUS COUNTY		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
EXTERNAL FUEL COMBUSTION	Residential	Coal	0.3	0.02%	0.02%	0.1	0.00%	0.00%	0.4	0.01%	0.01%
		Bagasse	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Distillate Oil	7.0	0.49%	0.47%	3.2	0.14%	0.13%	2.3	0.08%	0.08%
		Lignite	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Liquid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.4	0.01%	0.01%
		Petroleum Coke	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Solid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood/Bark Waste	685.9	48.29%	45.91%	1,880.5	79.65%	77.61%	2,128.4	73.94%	71.07%
		Sub-Total	*693*	*48.8%*	*46.4%*	*1,884*	*79.8%*	*77.7%*	*2,131*	*74.1%*	*71.2%*
	Electric Generation	Coal	2.7	0.19%	0.18%	1.5	0.06%	0.06%	4.6	0.16%	0.15%
		Bagasse	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Distillate Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Lignite	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Liquid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.1	0.00%	0.00%
		Petroleum Coke	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.9	0.06%	0.06%	0.5	0.02%	0.02%	1.6	0.05%	0.05%
		Solid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood/Bark Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*4*	*0.3%*	*0.2%*	*2*	*0.1%*	*0.1%*	*6*	*0.2%*	*0.2%*
	Industrial / Commercial / Institutional	Coal	7.1	0.50%	0.47%	9.4	0.40%	0.39%	43.5	1.51%	1.45%
		Bagasse	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Distillate Oil	0.9	0.06%	0.06%	0.5	0.02%	0.02%	2.3	0.08%	0.08%
		Lignite	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Liquid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	0.1	0.01%	0.01%	0.2	0.01%	0.01%	0.4	0.02%	0.01%
		Petroleum Coke	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.5	0.04%	0.03%	0.3	0.01%	0.01%	1.4	0.05%	0.05%
		Solid Waste	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood/Bark Waste	0.1	0.01%	0.01%	0.7	0.03%	0.03%	0.3	0.01%	0.01%
		Sub-Total	*9*	*0.6%*	*0.6%*	*11*	*0.5%*	*0.5%*	*48*	*1.7%*	*1.6%*

Table A-3 (Cont.) 2002 PM$_{2.5}$ Inventory, Select Rural Areas in New York State.

SECTOR	Type	Source	PRIMARY PM2.5								
			TYPICAL NYS RURAL COUNTIES								
			GREENE COUNTY			LEWIS COUNTY			CATTARAUGUS COUNTY		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
EFC TOTAL		**TOTAL COAL**	**10.1**	1.4%	0.7%	**11.0**	0.6%	0.5%	**48.4**	2.2%	1.7%
		TOTAL BAGASSE	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL DISTILLATE OIL	**7.9**	1.1%	0.6%	**3.7**	0.2%	0.2%	**4.6**	0.2%	0.2%
		TOTAL LIGNITE	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL LIQUID WASTE	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL LPG	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL NATURAL GAS	**0.1**	0.0%	0.0%	**0.2**	0.0%	0.0%	**0.9**	0.0%	0.0%
		TOTAL PETROLEUM COKE	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL PROCESS GAS	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL RESIDUAL OIL	**1.4**	0.2%	0.1%	**0.8**	0.0%	0.0%	**3.0**	0.1%	0.1%
		TOTAL SOLID WASTE	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%	**0.0**	0.0%	0.0%
		TOTAL WOOD/BARK WASTE	**686.1**	97.2%	48.3%	**1,881.2**	99.2%	79.7%	**2,128.7**	97.4%	74.0%
		TOTAL EXT. COMBUSTION	**705.6**	**100%**	**47%**	**1,896.8**	**100%**	**78%**	**2,185.6**	**100%**	**73%**
OTHER FUEL COMBUSTION	Internal Combustion	Diesel	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Kerosene	0.1	0.00%	0.00%	0.0	0.00%	0.00%	0.1	0.00%	0.00%
		LFG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*0.1*	*0.0%*	*0.00%*	*0.0*	*0.0%*	*0.00%*	*0.1*	*0.0%*	*0.00%*
	Turbine	Diesel	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Kerosene	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LFG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*0.0*	*0.0%*	*0.00%*	*0.0*	*0.0%*	*0.00%*	*0.0*	*0.0%*	*0.00%*
	Industrial Space Heating	Diesel	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Kerosene	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LFG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Natural Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*0.0*	*0.0%*	*0.00%*	*0.0*	*0.0%*	*0.00%*	*0.0*	*0.0%*	*0.00%*

Table A-3 (Cont.) 2002 $PM_{2.5}$ Inventory, Select Rural Areas in New York State.

SECTOR	Type	Source	PRIMARY PM2.5								
			TYPICAL NYS RURAL COUNTIES								
			GREENE COUNTY			LEWIS COUNTY			CATTARAUGUS COUNTY		
			Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total	Tons/yr	% of Sector	% of Total
OFC TOTAL		TOTAL DIESEL	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL GASOLINE	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL KEROSENE	0.1	0.0%	0.0%	0.0	0.0%	0.0%	0.1	0.0%	0.0%
		TOTAL LFG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL LPG	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL METHANOL	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTOAL NATURAL GAS	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL PROCESS GAS	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL RESIDUAL OIL	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL WASTE OIL	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL WOOD	0.0	0.0%	0.0%	0.0	0.0%	0.0%	0.0	0.0%	0.0%
		TOTAL OTHER COMBUST.	0.1	0%	0.0%	0.0	0%	0.0%	0.1	0%	0.0%
OTHER STATIONARY	Food and Kindred Products: SIC20	Conveyorized Charbroiling	0.9	0.06%	0.06%	0.6	0.02%	0.02%	3.0	0.10%	0.10%
		Under-fired Charbroiling	9.0	0.63%	0.60%	3.8	0.16%	0.16%	20.1	0.70%	0.67%
		Flat Griddle Frying	1.8	0.13%	0.12%	0.9	0.04%	0.04%	4.3	0.15%	0.14%
		Clamshell Griddle Frying	0.1	0.01%	0.01%	0.0	0.00%	0.00%	0.3	0.01%	0.01%
		Meat Products	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Other	0.0	0.00%	0.00%	0.0	0.00%	0.00%	0.0	0.00%	0.00%
		Sub-Total	*11.8*	*0.8%*	*0.8%*	*5.3*	*0.2%*	*0.2%*	*27.7*	*1.0%*	*0.9%*
	All Other Sources	Miscellaneous	703.1	49.50%	47.06%	459.1	19.44%	18.94%	665.0	23.10%	22.21%
		TOTAL OTHER STAT. SOURCES	715	50%	48%	464	20%	19%	693	24%	23%
TOTAL STATIONARY INVENTORY			1,421	100%	95%	2,361	100%	97%	2,878	100%	96%
TOTAL STATIONARY & MOBILE INVENTORY			1,494		100%	2,423		100%	2,995		100%

Table A-4. 2002 $PM_{2.5}$ Inventory, New York City Metropolitan $PM_{2.5}$ Nonattainment Area.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5											
			NYC Metro PM 2.5 Nonattainment Area (NAA)											
			NY COUNTIES (1)				NJ COUNTIES (1)				CT COUNTIES (1)			
			Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA
ON ROAD VEHICLES	Light Duty Gasoline Car (LDGC)	Exhaust	274.7	3.7%	1.0%	54.5%	171.6	3.6%	1.2%	34.0%	58.1	2.9%	0.8%	11.5%
		Brake Wear	238.7	3.2%	0.9%	53.4%	156.0	3.3%	1.1%	34.9%	52.7	2.7%	0.7%	11.8%
		Tire Wear	90.1	1.2%	0.3%	53.5%	58.6	1.2%	0.4%	34.8%	19.8	1.0%	0.3%	11.7%
		Sub-Total	603.4	8.1%	2.2%	53.9%	386.2	8.1%	2.6%	34.5%	130.6	6.6%	1.8%	11.7%
	Light Duty Gasoline Truck (LDGT)	Exhaust	198.2	2.7%	0.7%	53.1%	134.6	2.8%	0.9%	36.1%	40.2	2.0%	0.6%	10.8%
		Brake Wear	139.5	1.9%	0.5%	51.5%	104.7	2.2%	0.7%	38.7%	26.7	1.3%	0.4%	9.8%
		Tire Wear	52.6	0.7%	0.2%	51.7%	39.2	0.8%	0.3%	38.5%	10.0	0.5%	0.1%	9.8%
		Sub-Total	390.3	5.3%	1.4%	52.3%	278.5	5.9%	1.9%	37.3%	76.9	3.9%	1.1%	10.3%
	Heavy Duty Gasoline Truck (HDGV)	Exhaust	150.9	2.0%	0.5%	65.9%	49.7	1.0%	0.3%	21.7%	28.6	1.4%	0.4%	12.5%
		Brake Wear	11.6	0.2%	0.0%	63.5%	4.3	0.1%	0.0%	23.6%	2.4	0.1%	0.0%	12.9%
		Tire Wear	6.8	0.1%	0.0%	71.2%	1.8	0.0%	0.0%	18.5%	1.0	0.0%	0.0%	10.3%
		Sub-Total	169.3	2.3%	0.6%	65.9%	55.7	1.2%	0.4%	21.7%	31.9	1.6%	0.4%	12.4%
	Motorcycle (MC)	Exhaust	3.3	0.0%	0.0%	43.6%	3.5	0.1%	0.0%	46.4%	0.8	0.0%	0.0%	10.1%
		Brake Wear	1.0	0.0%	0.0%	41.7%	1.1	0.0%	0.0%	47.0%	0.3	0.0%	0.0%	11.3%
		Tire Wear	0.1	0.0%	0.0%	30.8%	0.2	0.0%	0.0%	55.9%	0.1	0.0%	0.0%	13.3%
		Sub-Total	4.4	0.1%	0.0%	42.6%	4.9	0.1%	0.0%	46.9%	1.1	0.1%	0.0%	10.5%
	Light Duty Diesel Car (LDDV)	Exhaust	34.7	0.5%	0.1%	60.3%	19.3	0.4%	0.1%	33.4%	3.6	0.2%	0.1%	6.3%
		Brake Wear	0.7	0.0%	0.0%	54.9%	0.5	0.0%	0.0%	38.6%	0.1	0.0%	0.0%	6.6%
		Tire Wear	0.2	0.0%	0.0%	50.1%	0.2	0.0%	0.0%	42.6%	0.0	0.0%	0.0%	7.3%
		Sub-Total	35.7	0.5%	0.1%	60.1%	20.0	0.4%	0.1%	33.6%	3.8	0.2%	0.1%	6.3%
	Light Duty Diesel Truck (LDDT)	Exhaust	66.0	0.9%	0.2%	88.9%	4.5	0.1%	0.0%	6.0%	3.8	0.2%	0.1%	5.1%
		Brake Wear	2.6	0.0%	0.0%	92.2%	0.1	0.0%	0.0%	3.8%	0.1	0.0%	0.0%	3.9%
		Tire Wear	0.9	0.0%	0.0%	92.0%	0.0	0.0%	0.0%	3.9%	0.0	0.0%	0.0%	4.0%
		Sub-Total	69.4	0.9%	0.3%	89.0%	4.6	0.1%	0.0%	5.9%	3.9	0.2%	0.1%	5.1%
	Heavy Duty Diesel Truck (HDDT)	Exhaust	642.4	8.7%	2.3%	36.9%	739.6	15.5%	5.1%	42.5%	357.4	18.1%	5.0%	20.5%
		Brake Wear	9.0	0.1%	0.0%	36.9%	10.3	0.2%	0.1%	42.0%	5.2	0.3%	0.1%	21.1%
		Tire Wear	9.9	0.1%	0.0%	34.9%	12.3	0.3%	0.1%	43.2%	6.2	0.3%	0.1%	21.9%
		Sub-Total	661.4	8.9%	2.4%	36.9%	762.1	16.0%	5.2%	42.5%	368.8	18.6%	5.2%	20.6%
	Heavy Duty Diesel Bus (HDDB)	Exhaust	557.6	7.5%	2.0%	89.7%	42.6	0.9%	0.3%	6.8%	21.8	1.1%	0.3%	3.5%
		Brake Wear	2.4	0.0%	0.0%	84.3%	0.3	0.0%	0.0%	11.0%	0.1	0.0%	0.0%	4.7%
		Tire Wear	1.3	0.0%	0.0%	84.4%	0.2	0.0%	0.0%	11.0%	0.1	0.0%	0.0%	4.6%
		Sub-Total	561.3	7.6%	2.0%	89.6%	43.1	0.9%	0.3%	6.9%	22.0	1.1%	0.3%	3.5%
		TOTAL EXHAUST	**1,927.8**	26.0%	7.0%	53.4%	**1,165.2**	24.5%	8.0%	32.3%	**514.2**	26.0%	7.2%	14.3%
		TOTAL BRAKE	**405.5**	5.5%	1.5%	52.6%	**277.4**	5.8%	1.9%	36.0%	**87.5**	4.4%	1.2%	11.4%
		TOTAL TIRE	**162.0**	2.2%	0.6%	52.0%	**112.5**	2.4%	0.8%	36.1%	**37.2**	1.9%	0.5%	11.9%
		TOTAL ON-ROAD	**2,495.3**	34%	9%	53.2%	**1,555.1**	33%	11%	33.2%	**638.9**	32%	9%	13.6%

Table A-4 (Cont.) 2002 $PM_{2.5}$ Inventory, NYC Metropolitan $PM_{2.5}$ Nonattainment Area.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5											
			NYC Metro PM 2.5 Nonattainment Area (NAA)											
			NY COUNTIES (1)				NJ COUNTIES (1)				CT COUNTIES (1)			
			Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA
NONROAD EQUIPMENT	Aircraft	Jet Fuel	212.5	2.9%	0.8%	66.0%	92.7	1.9%	0.6%	28.8%	16.9	0.9%	0.2%	5.3%
		Sub-Total	*212.5*	*2.9%*	*0.8%*	*66.0%*	*92.7*	*1.9%*	*0.6%*	*28.8%*	*16.9*	*0.9%*	*0.2%*	*5.3%*
	Airport GSE	Gasoline	0.2	0.0%	0.0%	0.1%	230.7	4.8%	1.6%	99.9%	0.0	0.0%	0.0%	0.0%
		Diesel	51.9	0.7%	0.2%	39.3%	79.5	1.7%	0.5%	60.2%	0.6	0.0%	0.0%	0.4%
		LPG	0.1	0.0%	0.0%	57.1%	0.1	0.0%	0.0%	42.4%	0.0	0.0%	0.0%	0.5%
		CNG	0.0	0.0%	0.0%	0.0%	0.1	0.0%	0.0%	100.0%	0.0	0.0%	0.0%	0.0%
		Sub-Total	*52.1*	*0.7%*	*0.2%*	*14.4%*	*310.3*	*6.5%*	*2.1%*	*85.5%*	*0.6*	*0.0%*	*0.0%*	*0.2%*
	Agricultural	Gasoline	0.0	0.0%	0.0%	39.0%	0.0	0.0%	0.0%	53.6%	0.0	0.0%	0.0%	7.4%
		Diesel	15.7	0.2%	0.1%	37.9%	22.7	0.5%	0.2%	54.6%	3.1	0.2%	0.0%	7.5%
		LPG	0.0	0.0%	0.0%	100.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	47.8%	0.0	0.0%	0.0%	44.0%	0.0	0.0%	0.0%	8.1%
		Sub-Total	*15.7*	*0.2%*	*0.1%*	*37.9%*	*22.7*	*0.5%*	*0.2%*	*54.6%*	*3.1*	*0.2%*	*0.0%*	*7.5%*
	Commercial	Gasoline	131.5	1.8%	0.5%	62.3%	64.6	1.4%	0.4%	30.6%	14.8	0.7%	0.2%	7.0%
		Diesel	565.1	7.6%	2.0%	62.0%	282.4	5.9%	1.9%	31.0%	64.5	3.3%	0.9%	7.1%
		LPG	4.0	0.1%	0.0%	62.3%	2.0	0.0%	0.0%	30.7%	0.5	0.0%	0.0%	7.0%
		CNG	2.8	0.0%	0.0%	62.6%	1.4	0.0%	0.0%	30.4%	0.3	0.0%	0.0%	7.0%
		Sub-Total	*703.4*	*9.5%*	*2.6%*	*62.0%*	*350.4*	*7.4%*	*2.4%*	*30.9%*	*80.1*	*4.0%*	*1.1%*	*7.1%*
	Construction & Mining	Gasoline	71.1	1.0%	0.3%	60.1%	33.5	0.7%	0.2%	28.4%	13.6	0.7%	0.2%	11.5%
		Diesel	2,333.3	31.5%	8.5%	59.7%	1,128.1	23.7%	7.7%	28.8%	449.1	22.7%	6.3%	11.5%
		LPG	0.5	0.0%	0.0%	60.0%	0.2	0.0%	0.0%	28.6%	0.1	0.0%	0.0%	11.5%
		CNG	0.0	0.0%	0.0%	59.2%	0.0	0.0%	0.0%	32.0%	0.0	0.0%	0.0%	8.8%
		Sub-Total	*2,404.9*	*32.4%*	*8.7%*	*59.7%*	*1,161.9*	*24.4%*	*8.0%*	*28.8%*	*462.8*	*23.4%*	*6.5%*	*11.5%*
	Industrial	Gasoline	3.5	0.0%	0.0%	43.6%	3.3	0.1%	0.0%	41.9%	1.1	0.1%	0.0%	14.4%
		Diesel	334.6	4.5%	1.2%	50.6%	245.7	5.2%	1.7%	37.2%	80.5	4.1%	1.1%	12.2%
		LPG	27.4	0.4%	0.1%	43.5%	26.4	0.6%	0.2%	42.0%	9.1	0.5%	0.1%	14.5%
		CNG	2.0	0.0%	0.0%	43.5%	1.9	0.0%	0.0%	42.0%	0.7	0.0%	0.0%	14.5%
		Sub-Total	*367.4*	*5.0%*	*1.3%*	*49.9%*	*277.3*	*5.8%*	*1.9%*	*37.7%*	*91.4*	*4.6%*	*1.3%*	*12.4%*
	Lawn & Garden	Gasoline	495.3	6.7%	1.8%	44.3%	472.8	9.9%	3.2%	42.3%	150.7	7.6%	2.1%	13.5%
		Diesel	112.6	1.5%	0.4%	43.8%	119.6	2.5%	0.8%	46.5%	25.1	1.3%	0.4%	9.8%
		LPG	0.2	0.0%	0.0%	42.2%	0.3	0.0%	0.0%	43.9%	0.1	0.0%	0.0%	13.9%
		CNG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Sub-Total	*608.2*	*8.2%*	*2.2%*	*44.2%*	*592.6*	*12.5%*	*4.1%*	*43.0%*	*175.8*	*8.9%*	*2.5%*	*12.8%*

Table A-4 (Cont.) 2002 $PM_{2.5}$ Inventory, NYC Metropolitan $PM_{2.5}$ Nonattainment Area.

SECTOR	Vehicle Type	Source	PRIMARY PM2.5											
			NYC Metro PM 2.5 Nonattainment Area (NAA)											
			NY COUNTIES (1)				NJ COUNTIES (1)				CT COUNTIES (1)			
			Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA
NONROAD EQUIPMENT (cont)	Logging	Gasoline	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Diesel	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		LPG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Sub-Total	*0.0*	*0.0%*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*0.0%*	*0.0*	*0.0%*	*0.0%*	*0.0%*
	Marine	Gasoline	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Diesel	64.6	0.9%	0.2%	23.6%	30.3	0.6%	0.2%	11.1%	178.2	9.0%	2.5%	65.3%
		LPG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		CNG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Sub-Total	*64.6*	*0.9%*	*0.2%*	*23.6%*	*30.3*	*0.6%*	*0.2%*	*11.1%*	*178.2*	*9.0%*	*2.5%*	*65.3%*
	Rail	Gasoline	0.1	0.0%	0.0%	61.5%	0.0	0.0%	0.0%	29.8%	0.0	0.0%	0.0%	8.7%
		Diesel	33.1	0.4%	0.1%	22.0%	114.9	2.4%	0.8%	76.2%	2.7	0.1%	0.0%	1.8%
		LPG	0.0	0.0%	0.0%	61.6%	0.0	0.0%	0.0%	29.8%	0.0	0.0%	0.0%	8.6%
		CNG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Sub-Total	*33.2*	*0.4%*	*0.1%*	*22.0%*	*114.9*	*2.4%*	*0.8%*	*76.2%*	*2.7*	*0.1%*	*0.0%*	*1.8%*
	Recreational	Gasoline	442.5	6.0%	1.6%	44.0%	245.3	5.2%	1.7%	24.4%	318.6	16.1%	4.5%	31.7%
		Diesel	13.0	0.2%	0.0%	44.1%	6.0	0.1%	0.0%	20.4%	10.5	0.5%	0.1%	35.5%
		LPG	0.0	0.0%	0.0%	66.2%	0.0	0.0%	0.0%	27.3%	0.0	0.0%	0.0%	6.5%
		CNG	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%	0.0	0.0%	0.0%	0.0%
		Sub-Total	*455.5*	*6.1%*	*1.7%*	*44.0%*	*251.3*	*5.3%*	*1.7%*	*24.3%*	*329.1*	*16.6%*	*4.6%*	*31.8%*
		TOTAL JET FUEL	**212.5**	***2.9%***	***0.8%***	***66.0%***	**92.7**	***1.9%***	***0.6%***	***28.8%***	**16.9**	***0.9%***	***0.2%***	***5.3%***
		TOTAL GASOLINE	**1,144.0**	***15.4%***	***4.1%***	***42.5%***	**1,050.4**	***22.1%***	***7.2%***	***39.0%***	**498.8**	***25.2%***	***7.0%***	***18.5%***
		TOTAL DIESEL	**3,524.0**	***47.5%***	***12.8%***	***55.3%***	**2,029.2**	***42.6%***	***13.9%***	***31.9%***	**814.4**	***41.1%***	***11.4%***	***12.8%***
		TOTAL LPG	**32.2**	***0.4%***	***0.1%***	***45.5%***	**28.9**	***0.6%***	***0.2%***	***40.8%***	**9.7**	***0.5%***	***0.1%***	***13.7%***
		TOTAL CNG	**4.8**	***0.1%***	***0.0%***	***52.6%***	**3.3**	***0.1%***	***0.0%***	***36.6%***	**1.0**	***0.0%***	***0.0%***	***10.8%***
		TOTAL NON-ROAD	**4,917.6**	***66%***	***18%***	***52.0%***	**3,204.5**	***67%***	***22%***	***33.9%***	**1,340.8**	***68%***	***19%***	***14.2%***

Table A-4 (Cont.) 2002 $PM_{2.5}$ Inventory, NYC Metropolitan $PM_{2.5}$ Nonattainment Area.

SECTOR	Type	Source	PRIMARY PM2.5											
			NYC Metro NAA PM 2.5											
			NY COUNTIES (1)				NJ COUNTIES (1)				CT COUNTIES (1)			
			Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA
EXTERNAL FUEL COMBUSTION	Residential	Coal	16.1	0.08%	0.06%	0.05%	0.5	0.01%	0.00%	53.00%	0.7	0.01%	0.01%	82.68%
		Bagasse	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Distillate Oil	962.0	4.77%	3.49%	2.73%	182.1	1.85%	1.25%	0.52%	282.2	5.46%	3.95%	0.80%
		Lignite	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Liquid Waste	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		LPG	1.0	0.00%	0.00%	0.00%	1.0	0.01%	0.01%	0.00%	0.4	0.01%	0.01%	0.00%
		Natural Gas	48.1	0.24%	0.17%	0.14%	33.9	0.34%	0.23%	0.10%	4.9	0.10%	0.07%	0.01%
		Petroleum Coke	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Solid Waste	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Wood/Bark Waste	1,554.5	7.71%	5.64%	4.42%	2,632.3	26.73%	18.02%	7.48%	2,423.3	46.91%	33.91%	6.89%
		Sub-Total	*2,582*	*12.80%*	*9.36%*	*7.34%*	*2,850*	*28.94%*	*19.51%*	*8.10%*	*2,712*	*52.49%*	*37.95%*	*7.71%*
	Electric Generation	Coal	1,705.7	8.46%	6.18%	4.85%	2,492.6	25.32%	17.07%	7.09%	158.5	3.07%	2.22%	0.45%
		Bagasse	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Distillate Oil	0.7	0.00%	0.00%	0.00%	2.3	0.02%	0.02%	0.01%	0.1	0.00%	0.00%	0.00%
		Lignite	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Liquid Waste	0.2	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Natural Gas	762.3	3.78%	2.76%	2.17%	68.0	0.69%	0.47%	0.19%	0.1	0.00%	0.00%	0.00%
		Petroleum Coke	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.00%	0.3	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Residual Oil	1,639.2	8.13%	5.94%	4.66%	51.1	0.52%	0.35%	0.15%	131.7	2.55%	1.84%	0.37%
		Solid Waste	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Wood/Bark Waste	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Sub-Total	*4,108*	*20.37%*	*14.90%*	*11.68%*	*2,614*	*26.55%*	*17.90%*	*7.43%*	*290*	*5.62%*	*4.06%*	*0.83%*
	Industrial / Commercial / Institutional	Coal	851.4	4.22%	3.09%	2.42%	3.4	0.03%	0.02%	0.01%	97.8	1.89%	1.37%	0.28%
		Bagasse	0.0	0.00%	0.00%	0.00%	0.1	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Distillate Oil	404.8	2.01%	1.47%	1.15%	131.0	1.33%	0.90%	0.37%	377.7	7.31%	5.29%	1.07%
		Lignite	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Liquid Waste	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		LPG	0.4	0.00%	0.00%	0.00%	2.1	0.02%	0.01%	0.01%	0.1	0.00%	0.00%	0.00%
		Natural Gas	60.7	0.30%	0.22%	0.17%	91.1	0.93%	0.62%	0.26%	8.0	0.16%	0.11%	0.02%
		Petroleum Coke	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Process Gas	2.5	0.01%	0.01%	0.01%	54.8	0.56%	0.37%	0.16%	0.0	0.00%	0.00%	0.00%
		Residual Oil	284.7	1.41%	1.03%	0.81%	64.5	0.66%	0.44%	0.18%	55.6	1.08%	0.78%	0.16%
		Solid Waste	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Wood/Bark Waste	65.1	0.32%	0.24%	0.18%	0.1	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Sub-Total	*1,670*	*8.28%*	*6.05%*	*4.75%*	*347*	*3.53%*	*2.38%*	*0.99%*	*539*	*10.44%*	*7.55%*	*1.53%*

Table A-4 (Cont.) 2002 $PM_{2.5}$ Inventory, NYC Metropolitan $PM_{2.5}$ Nonattainment Area.

SECTOR	Type	Source	PRIMARY PM2.5											
			NYC Metro NAA PM 2.5											
			NY COUNTIES (1)				NJ COUNTIES (1)				CT COUNTIES (1)			
			Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA
EFC TOTAL		TOTAL COAL	2,573	30.8%	12.8%	7.31%	2,497	43.0%	25.4%	7.10%	257	7.3%	5.0%	0.73%
		TOTAL BAGASSE	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%
		TOTAL DISTILLATE OIL	1,367	16.4%	6.8%	3.89%	315	5.4%	3.2%	0.90%	660	18.6%	12.8%	1.88%
		TOTAL LIGNITE	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%
		TOTAL LIQUID WASTE	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%
		TOTAL LPG	1	0.0%	0.0%	0.00%	3	0.1%	0.0%	0.01%	1	0.0%	0.0%	0.00%
		TOTAL NATURAL GAS	871	10.4%	4.3%	2.48%	193	3.3%	2.0%	0.55%	13	0.4%	0.3%	0.04%
		TOTAL PETROLEUM COKE	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%
		TOTAL PROCESS GAS	2	0.0%	0.0%	0.01%	55	0.9%	0.6%	0.16%	0	0.0%	0.0%	0.00%
		TOTAL RESIDUAL OIL	1,924	23.0%	9.6%	5.47%	116	2.0%	1.2%	0.33%	187	5.3%	3.6%	0.53%
		TOTAL SOLID WASTE	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%	0	0.0%	0.0%	0.00%
		TOTAL WOOD/BARK WASTE	1,620	19.4%	8.0%	4.60%	2,632	45.3%	26.7%	7.48%	2,423	68.4%	46.9%	6.89%
		TOTAL EXT. COMBUSTION	8,359	100%	30%	23.76%	5,811	100%	40%	16.52%	3,541	100%	50%	10.07%
OTHER FUEL COMBUSTION	Internal Combustion	Diesel	21.4	0.11%	0.08%	0.06%	3.8	0.04%	0.03%	0.01%	1.1	0.02%	0.02%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Kerosene	0.0	0.00%	0.00%	0.00%	47.5	0.48%	0.32%	0.13%	0.0	0.00%	0.00%	0.00%
		LFG	6.9	0.03%	0.03%	0.02%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Natural Gas	6.9	0.03%	0.03%	0.02%	9.4	0.10%	0.06%	0.03%	0.2	0.00%	0.00%	0.00%
		Process Gas	11.6	0.06%	0.04%	0.03%	4.2	0.04%	0.03%	0.01%	0.0	0.00%	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Sub-Total	46.9	0.23%	0.17%	0.13%	64.8	0.66%	0.44%	0.18%	1.3	0.03%	0.02%	0.00%
	Turbine	Diesel	24.2	0.12%	0.09%	0.07%	15.7	0.16%	0.11%	0.04%	2.4	0.05%	0.03%	0.01%
		Gasoline	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Kerosene	28.3	0.14%	0.10%	0.08%	0.2	0.00%	0.00%	0.00%	1.2	0.02%	0.02%	0.00%
		LFG	5.6	0.03%	0.02%	0.02%	26.0	0.26%	0.17%	0.07%	0.0	0.00%	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Natural Gas	36.3	0.18%	0.13%	0.10%	60.6	0.62%	0.42%	0.17%	63.2	1.22%	0.89%	0.18%
		Process Gas	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Sub-Total	94.3	0.47%	0.34%	0.27%	101.5	1.03%	0.69%	0.29%	66.8	1.29%	0.94%	0.19%
	Industrial Space Heating	Diesel	13.0	0.06%	0.05%	0.04%	4.5	0.05%	0.03%	0.01%	0.2	0.00%	0.00%	0.00%
		Gasoline	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Kerosene	31.2	0.15%	0.11%	0.09%	0.0	0.00%	0.00%	0.00%	10.1	0.19%	0.14%	0.03%
		LFG	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		LPG	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Methanol	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Natural Gas	3.9	0.02%	0.01%	0.01%	9.3	0.09%	0.06%	0.03%	0.0	0.00%	0.00%	0.00%
		Process Gas	0.0	0.00%	0.00%	0.00%	6.7	0.07%	0.05%	0.02%	0.0	0.00%	0.00%	0.00%
		Residual Oil	0.0	0.00%	0.00%	0.00%	2.8	0.03%	0.02%	0.01%	0.0	0.00%	0.00%	0.00%
		Waste Oil	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Wood	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Sub-Total	48.1	0.24%	0.17%	0.14%	23.3	0.24%	0.16%	0.07%	10.2	0.20%	0.14%	0.03%

Table A-4 (Cont.) 2002 $PM_{2.5}$ Inventory, NYC Metropolitan $PM_{2.5}$ Nonattainment Area.

SECTOR	Type	Source	PRIMARY PM2.5											
			NYC Metro NAA PM 2.5											
			NY COUNTIES (1)				NJ COUNTIES (1)				CT COUNTIES (1)			
			Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA	Tons/yr	% of Sector	% of Total	% of NAA
OFC TOTAL		TOTAL DIESEL	58.6	0.3%	0.2%	0.17%	24.1	0.2%	0.2%	0.07%	3.6	0.1%	0.1%	0.01%
		TOTAL GASOLINE	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%
		TOTAL KEROSENE	59.5	0.3%	0.2%	0.17%	47.6	0.5%	0.3%	0.14%	11.3	0.2%	0.2%	0.03%
		TOTAL LFG	12.5	0.1%	0.0%	0.04%	25.0	0.3%	0.2%	0.07%	0.0	0.0%	0.0%	0.00%
		TOTAL LPG	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%
		TOTAL METHANOL	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%
		TOTOAL NATURAL GAS	47.2	0.2%	0.2%	0.13%	79.3	0.8%	0.5%	0.23%	63.5	1.2%	0.9%	0.18%
		TOTAL PROCESS GAS	11.6	0.1%	0.0%	0.03%	10.9	0.1%	0.1%	0.03%	0.0	0.0%	0.0%	0.00%
		TOTAL RESIDUAL OIL	0.0	0.0%	0.0%	0.00%	2.8	0.0%	0.0%	0.01%	0.0	0.0%	0.0%	0.00%
		TOTAL WASTE OIL	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%
		TOTAL WOOD	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%	0.0	0.0%	0.0%	0.00%
		TOTAL OTHER COMBUST.	189.4	1%	1%	0.54%	189.6	2%	1%	0.54%	78.4	2%	1%	0.22%
OTHER STATIONARY	Food and Kindred Products: SIC20	Conveyorized Charbroiling	301.3	1.49%	1.09%	0.86%	0.0	0.00%	0.00%	0.00%	49.4	0.96%	0.69%	0.14%
		Under-fired Charbroiling	2,408.0	11.94%	8.73%	6.85%	1,478.7	15.02%	10.12%	4.20%	360.5	6.98%	5.04%	1.02%
		Flat Griddle Frying	491.1	2.44%	1.78%	1.40%	0.0	0.00%	0.00%	0.00%	73.7	1.43%	1.03%	0.21%
		Clamshell Griddle Frying	32.8	0.16%	0.12%	0.09%	0.0	0.00%	0.00%	0.00%	5.3	0.10%	0.07%	0.02%
		Meat Products	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Other	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%	0.0	0.00%	0.00%	0.00%
		Sub-Total	3,233.2	*16.0%*	*11.7%*	*9.19%*	1,478.7	*15.0%*	*10.1%*	*4.20%*	488.8	*9.5%*	*6.8%*	*1.39%*
	All Other Sources	Miscellaneous	8,385.5	41.58%	30.40%	23.84%	2,366.6	24.04%	16.20%	6.73%	1,057.3	20.47%	14.80%	3.01%
		TOTAL OTHER STAT. SOURCES	11,619	58%	42%	33%	3,845	39%	26%	11%	1,546	30%	22%	4%
TOTAL STATIONARY INVENTORY			20,167	100%	73%	57%	9,846	100%	67%	28%	5,166	100%	72%	15%
TOTAL STATIONARY & MOBILE INVENTORY			27,580		100%		14,606		100%		7,145		100%	

Appendix B

DIESEL ENGINE TECHNOLOGIES

INTRODUCTION

Diesel engine technology has changed dramatically in the last 20 years. Environmental Protection Agency (EPA) regulations have reduced the allowable nitrogen oxides (NO_X) and particulate matter (PM) emissions from new onroad diesel truck and bus engines by 80% and 90%, respectively, since 1988.
To meet increasingly stringent emissions standards, engine manufacturers have made fundamental changes to engines and have added exhaust "after-treatment" devices. Some of the changes that have been made to onroad diesel engines over the last 20 years include the change from a two-stroke to four-stroke cycle, improved cylinder designs, improved fuel injectors and higher injection pressures, electronic fuel control, the addition of turbochargers, and the addition of exhaust gas recirculation. These changes have contributed to both improved fuel economy and lower emissions. To a lesser extent, these technology advances have also been applied to nonroad engines.

Stricter standards for onroad diesel engines that take effect between 2007 and 2010 will reduce allowable NO_X and PM emissions by a further 90% (EPA 2001) and are driving additional changes—primarily the use of even more effective after-treatment technologies. Regulations for the diesel engines used in nonroad equipment such as ships, trains, and agricultural and construction tractors have lagged—emissions standards for these engines will not achieve regulatory parity with those for onroad engines until 2015 or later (EPA 2004c; EPA 2004d). However, because many of the nonroad engines are very similar to onroad engines, the same technologies being developed for the onroad market can in many cases be used to reduce their emissions in advance of regulatory mandates.

While great progress has been achieved in reducing emissions from new diesel engines, there is a significant legacy fleet of older diesels still in use. Many of these engines were made before diesel emissions standards went into effect; diesel trucks and buses can stay in service for 20 years, while some nonroad equipment can last for more than 40 years. Many of the same strategies that can be applied to new diesel engines to meet stricter EPA standards can also be applied to existing engines to reduce their emissions.

REDUCING EMISSIONS FROM EXISTING VEHICLES

There are four ways that exhaust emissions from existing diesel vehicles and fleets can be reduced—most of which take advantage of the progress that has been made in the development of new diesel engines. These approaches are:

- *Replace/Repower/Rebuild:* Retire vehicles or engines "early" and replace with new, cleaner engines, or with engines rebuilt to incorporate cleaner technologies.
- *Retrofit:* Add an "after-treatment" device to the tailpipe.

- *Refuel:* Use a “cleaner” alternative diesel fuel.
- *Reduce Idling:* Reduce unproductive engine idling, thereby reducing emissions and saving fuel.

Each of these approaches is discussed further below. A brief description of the technologies, cost estimates, implementation issues, and emissions benefits is included for retrofits, alternative diesel fuels, and idle reduction.

Replace/Repower/Rebuild

For many fleets of older diesel vehicles, particularly those built prior to 1988, dramatic reductions in PM emissions can be achieved by retiring older vehicles and replacing them with new vehicles equipped with new engines that meet much stricter emissions standards. In many cases, these new engines will also achieve better fuel economy, providing some economic benefit to the owner.

Another approach is to keep the vehicle but retire the engine by “repowering” with a new engine. While some fleet operators have taken this approach with onroad vehicles,[45] it is much more likely to be cost-effective for nonroad fleets, including some marine vessels, locomotives, and large specialty construction equipment. These types of equipment typically have much longer life spans than onroad trucks and buses; in addition, the engines generally represent a smaller percentage of the value of the entire vehicle. For some nonroad equipment, it may be possible to repower with an onroad engine certified to lower emissions standards than new nonroad engines, and thereby produce even greater emissions reductions.

Engines that are properly maintained and tuned perform better and typically emit less pollution than poorly maintained engines, regardless of how old they are. Proper maintenance includes the periodic overhaul/rebuilding of the entire engine to evaluate and replace major components that wear out, such as cylinder liners. Proper maintenance and rebuilding of severely worn engines may also increase fuel economy and extend engine life. Most diesel engines, particularly the very large nonroad engines used in marine vessels, locomotives, and some construction equipment, typically go through one or two major overhauls during their lifetimes. Some of the changes that have been applied to new engine designs can be applied as an upgrade to an existing engine during an overhaul, resulting in lower emissions. As with repowering, this approach is generally more practical and cost-effective for the largest and longest-lived nonroad engines.

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[45] Between 2000 and 2001, MTA New York City Transit repowered 600 model year 1990 and 1993 urban transit buses with new engines.

Retrofit

"After-treatment" refers to a device or technology installed in a vehicle's exhaust system to reduce emissions. Unlike in-engine and fuel technologies, these devices do not reduce the emissions produced by the engine; rather, they act to reduce exhaust pollutants after they have left the engine but before they enter the atmosphere. With some exceptions and limitations, as noted below, after-treatment devices can be retrofitted onto existing diesel vehicles to reduce their emissions.

There are a number of different approaches to diesel exhaust after-treatment that reduce PM emissions, but most include a catalyst that promotes chemical reactions in the exhaust, oxidizing the hydrocarbons (HC) and PM to carbon dioxide (CO_2) and water. The five most common diesel PM retrofit technologies are described in detail below; these technologies are all potentially applicable to a wide variety of diesel engines. Some are commercially available now for a significant number of applications, others for a smaller number of specific engines.

In order to provide end-users with standardized information on the effectiveness of various retrofit and alternative diesel fuel technologies, both the EPA and the California Air Resources Board (CARB) operate technology verification programs. Manufacturers who want to receive verification must submit test results from standard emissions tests, along with information on the limitations and special requirements of the technology (e.g., minimum fuel sulfur level). CARB also requires manufacturers to provide minimum warranties for their products.

The EPA and CARB programs have slightly different requirements, but each organization has agreed to provide reciprocity to products verified under the other's program. The EPA program requires manufacturers to verify a product or technology separately for each of a number of different engine families and also to report the actual emissions reductions achieved for NO_X, PM, HC, and carbon monoxide (CO) (EPA 2006).

The CARB program also provides verification for specific engines and engine families. However, it separates technologies into three levels, based on emissions reduction effectiveness for PM, and only reports the level that a product falls into rather than the actual PM reductions achieved. The CARB categories include Level 1 (greater than 25% PM reduction), Level 2 (greater than 50% PM reduction), and Level 3 (greater than 85% PM reduction). In the case of technologies that also reduce NO_X, the actual NO_X control-effectiveness is reported (CARB 2006).

Either EPA or CARB, or both, has verified at least one commercial product for each of the retrofit technologies and alternative diesel fuels discussed below. A number of commercial products have been verified that combine several after-treatment technologies, either to improve their effectiveness at reducing

PM, or to reduce both PM and NO_X simultaneously. In addition, several commercial products have been verified that combine engine modifications to reduce NO_X with after-treatment to reduce PM. The latest information on verified technologies appears on the EPA and CARB Web sites (see references).

Refuel

Various fuel parameters can affect diesel PM emissions, regardless of engine type and after-treatment technology. One approach to reducing emissions is to use a non-standard diesel fuel formulation. Most alternative "clean" diesel fuels are based on blending a small amount (up to 20% by volume) of another substance with commercial petroleum diesel. Various approaches are used, including blending with water (emulsified diesel), alcohol (oxy-diesel), converted vegetable oil (bio-diesel), or metal-based powdered catalysts in a liquid suspension (fuel-borne catalysts). In the future, synthetic diesel fuel made from natural gas may also be available.

In addition, reducing sulfur levels in diesel fuel can itself reduce PM emissions, while also improving the effectiveness of some retrofit PM control devices, as discussed below. EPA rules will require significant diesel sulfur reductions beginning in late 2006 (EPA 2001; EPA 2004d). In some parts of the country, ultra-low sulfur diesel fuel with less than 30 parts per million (ppm) sulfur has been available for several years, ahead of EPA deadlines. Some fleets have already begun to use this fuel in order to reduce emissions.

The implementation dates for mandatory sulfur reductions in nonroad and marine fuels lag behind those for onroad fuel by several years. Between now and 2012, the use of reduced sulfur onroad diesel fuel in nonroad construction, locomotive, and marine engines could complement other emissions reduction strategies.

Reduce Idling

The primary role of a diesel engine in most vehicles is to provide propulsion power to move the vehicle. However, most onroad and nonroad diesel equipment has auxiliary loads (pumps, air compressors, heating/air conditioning, etc.) that are also supplied by the engine, both when the equipment is moving and when it is stationary.

In certain situations, idling of the vehicle's very large diesel engine in order to supply a fairly small load results in excess pollution. Common examples include long-haul truck drivers who leave their trucks running all night in order to provide lights and heating or cooling in the truck cabin while they are sleeping, and charter bus operators who idle their buses while waiting for their passengers to return from an outing—again to maintain comfort in the bus cabin.

Opportunities exist to reduce vehicle emissions by finding other ways of supplying these auxiliary loads, thus allowing the operator to shut off the main engine. Approaches include adding equipment to the vehicle (e.g., a battery system or a small auxiliary engine), or installing infrastructure where specific types of vehicles congregate (e.g., electrical connections at truck stops or waiting facilities for bus drivers at major tourist attractions).

Besides onroad trucks and buses, certain types of nonroad diesel equipment may also be a good target for idle-reduction efforts: in particular, switchyard locomotives and some passenger and container ships that have high onboard electrical loads while in port. These loads, which are usually supplied by onboard diesel engines, could be supplied by the land-side electrical grid, resulting in significant net emissions reductions.

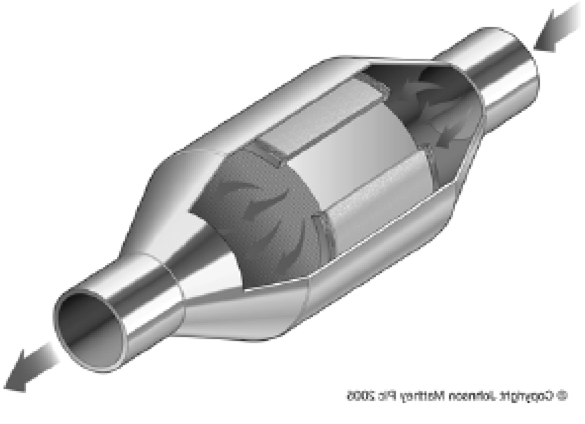
Depending on how much idling is eliminated, idle reduction technologies can reduce total fuel consumption by up to 15% on trucks and 25% on locomotives (ANL 2005), with coincident reductions in PM emissions. In most cases, the required idle reduction equipment will pay for itself in a relatively short time through fuel savings (see Appendix C).

PM RETROFIT TECHNOLOGIES

Diesel Oxidation Catalyst

A diesel oxidation catalyst (DOC) is a flow-through metal or ceramic substrate coated with a precious metal catalyst (e.g., platinum) and packaged into a metal container similar to an exhaust muffler/resonator. The DOC sits in the exhaust stream of a vehicle and promotes the oxidation of unburned PM, HC, and CO in the exhaust passing through it, producing CO_2 and water.

Figure B-1. Typical Diesel Oxidation Catalyst.
Courtesy of Johnson Matthey, Inc.



DOCs reduce HC and CO emissions by 20–50% and reduce PM emissions by 10–30% .[46] PM reductions are primarily the “wet” organic carbon portion of PM, rather than the elemental carbon portion. When applied to older onroad trucks and buses, DOCs can reduce total PM emissions by 2–20 pounds per vehicle per year depending on duty cycle. These reductions can be achieved for $30,000–$450,000/ton; $/ton costs will be higher for newer vehicles and vehicles that are used less. When applied to older nonroad equipment such as construction vehicles, DOCs can reduce total

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[46] For all technologies, the reported range of emissions reductions is from test results reported under the EPA and CARB Technology Verification Programs (EPA 2006; CARB 2006).

annual PM by 11–70 pounds per vehicle, and these reductions can be achieved for $15,000–$125,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles because they have much lower baseline emissions levels.[47] Annual reductions from locomotives can be as high as 500 pounds per vehicle due to their larger engine size, higher baseline emissions, and higher usage rates. The cost effectiveness for locomotives is similar to that for nonroad construction equipment.

The following implementation issues[48] and costs[49] apply to the installation and operation of DOCs:

- If properly sized and created with an appropriate catalyst formulation, DOCs can be used on virtually any diesel engine, but they may not be appropriate for very old two-stroke engines with high oil consumption because excess PM can plug the flow channels in the DOC.
- Retrofit installation is generally straightforward and takes only a few hours.
- Fuel should not contain more than 500 ppm sulfur. With higher sulfur fuels, excessive sulfate $PM_{2.5}$ is created across the catalyst.
- DOC purchase cost is $1,000–$2,000 for a typical truck or bus engine (250–350 horsepower (hp)); larger engines generally require larger, more expensive devices and cost is roughly proportional to engine horsepower. Costs for nonroad engines will be higher than costs for onroad engines of the same horsepower because of their higher baseline emissions levels.
- DOCs can incur a fuel economy penalty of 1–2%.

Passive Diesel Particulate Filter

A passive diesel particulate filter (DPF) combines a DOC with a porous ceramic, metal mesh, or silicon carbide filter in a metal container similar to an exhaust muffler/resonator. There are several variations on the design: some DPFs have a separate flow-through catalyst section in series with an uncatalyzed filter, while others use a filter with the catalyst applied directly to it. The DPF sits in the exhaust stream of the vehicle like a typical muffler/resonator.

Inside the device, the gaseous components of the exhaust pass through the porous walls of the filter, while the solid PM particles are physically trapped on and in the filter walls. Carbon will typically oxidize only at temperatures greater than 600 degrees Centigrade (^{O}C); the catalyst promotes oxidation of the trapped PM (carbon and HC) at the temperatures more typical of diesel exhaust (200–400 ^{O}C), which then exits the

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[47] See Appendix C for a complete description of the methodology used to calculate the annual reductions and cost-effectiveness for each technology.

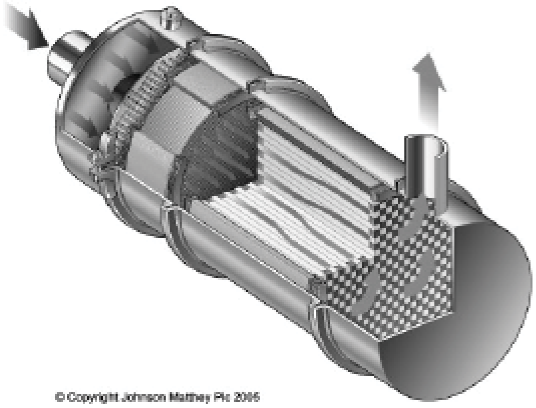
[48] For all technologies, information on implementation issues is taken from technology summaries on the EPA/OTAQ Web site (EPA 2004g), discussions with manufacturers, and the author's project experience.

[49] Typical costs for all technologies are from technology summaries available on the EPA/OTAQ Web site (EPA 2004g) and from an independent cost survey by the Manufacturers of Emissions Controls Association (MECA) (MECA 2000).

filter as gaseous CO_2 and water. The catalyst also oxidizes gaseous HC and CO in the exhaust like a typical DOC.

Figure B-2. Typical Passive DPF.

Courtesy of Johnson Matthey, Inc.



Passive DPFs reduce HC and CO emissions by 60–90% and reduce PM emissions by 80–90%. PM reductions include both the organic carbon and elemental carbon portions of PM. When applied to older onroad trucks and buses DPFs can reduce totalPM by 13–60 pounds per vehicle per year depending on duty cycle. These reductions can be achieved for $60,000-$350,000/ton; $/ton costs will be higher for newer vehicles and vehicles that are used less. When applied to older nonroad equipment such as construction vehicles, DPFs can reduce total annual PM by 90–200 pounds per vehicle, and these reductions can be achieved for $35,000–$100,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles because they have much lower baseline emissions levels. Annual reductions from locomotives can be as high as 1,500 pounds per vehicle due to their larger engine size, higher baseline emissions, and higher usage rates. The cost-effectiveness for locomotives is similar to that for nonroad construction equipment.

The following implementation issues and costs apply to the installation and operation of DPFs:

- DPFs put a restriction in the exhaust and increase “back pressure” on the engine pistons; they must be carefully designed to minimize this impact, since high back pressure can lead to progressive engine damage.
- Over time, noncombustible components of lube oil collect as ash in the filter. Approximately once per year, DPFs must be removed from the vehicle and cleaned to remove the ash.
- As ash and/or excess carbon builds up in the filter, engine back pressure rises. DPFs should always be used with a back pressure monitoring system that triggers a maintenance light once the back pressure rises above a set threshold.
- Fuel cannot have more than 50 ppm sulfur. Higher levels of sulfur reduce the oxidation efficiency of collected PM and can result in filter plugging. Onroad diesel fuel will be required to have less than 15 ppm sulfur beginning in late 2006. This ultra-low sulfur fuel is available in some parts of the country today and can be used with DPFs. Nonroad diesel fuel with sulfur levels of 500 ppm or more will continue to be available through 2012. DPF retrofits will not be possible on all nonroad equipment until after 2012 unless onroad fuel is used.

- DPFs will not work on all engines. The more PM the engine produces, the larger the filter and catalyst must be to work continuously without plugging. For some very old, very dirty engines, it may not be practical to design a DPF that will work consistently, due to cost issues and space constraints.
- The DPF requires a minimum exhaust temperature of 240–300°C for 35–40% of the time to oxidize the collected PM in the filter. This is easily achievable for many vehicles, but certain engines and certain duty cycles may not have sufficient exhaust temperature to use a DPF effectively. In general, DPFs can be used for duty cycles in which the diesel engine operates for a majority of the time under high loads. Lightly loaded duty cycles may not be appropriate for DPFs. Evaluation of the temperature profile for the engine/duty cycle is highly recommended before DPFs are retrofitted on a vehicle type for the first time.
- Installation is relatively straightforward and usually takes less than five hours per vehicle.
- Purchase cost is \$6,500–\$10,000 for a typical truck or bus engine (250–350 hp); larger engines generally require larger, more expensive devices, and cost is roughly proportional to engine horsepower. Costs for nonroad engines will be higher than costs for onroad engines of the same horsepower because of their higher baseline emissions levels.
- DPFs incur a fuel economy penalty of 2–4%.

Active Diesel Particulate Filter

An active diesel particulate filter (ADPF) system uses the same porous filter as a passive DPF to remove PM from diesel exhaust. Like passive DPFs, these systems may also employ a catalyst to lower the temperature at which the collected PM will oxidize off of the filter. However, in order to accommodate a wider range of duty cycles, they also incorporate some kind of active system to raise the temperature inside the filter.

The most common method used to raise the temperature in the filter is to inject additional diesel fuel into the exhaust stream across a small catalyst, downstream of the engine but in front of the filter. Oxidation of this fuel raises the temperature.

Virtually all new onroad diesel trucks and buses are expected to use ADPF systems beginning in 2007 to comply with the stricter EPA PM standards that will take effect at that time. While there are currently very few ADPF systems commercially available for retrofits, more are expected to become available after 2007.

ADPFs reduce HC and CO emissions by 60–90% and reduce PM emissions by 80 –90%. PM reductions include both the organic carbon and elemental carbon portions of PM. When applied to older onroad trucks and buses, ADPFs can reduce total annual PM by 13–60 pounds per vehicle depending on duty cycle. These reductions can be achieved for \$65,000 - \$350,000/ton; \$/ton costs will be higher for newer vehicles and vehicles that are used less. When applied to older nonroad equipment such as construction vehicles, ADPFs can reduce total annual PM by 90–200 pounds per vehicle, and these reductions can be achieved for $35,000 -$100,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles because they have much lower baseline emissions levels. Annual reductions from locomotives can be as high as 1,500 pounds per vehicle due to their larger engine size, higher baseline emissions, and higher usage rates. The cost-effectiveness for locomotives is similar to that for nonroad construction equipment.

The following implementation issues and costs apply to the installation and operation of ADPFs:

- ADPFs increase back pressure on the engine pistons and must be carefully designed to minimize this impact because high back pressure can lead to progressive engine damage.
- Over time, noncombustible components of lube oil collect as ash in the filter. Approximately once per year, ADPFs must be removed from the vehicle and cleaned to remove the ash.
- ADPFs will work on a wider range of engines and duty cycles than passive DPFs because they are not limited by the inherent exhaust temperature profile.
- ADPF systems are significantly more complicated than passive DPFs. In addition to the filter element, an ADPF system will typically contain a fuel pump, a fuel injector into the exhaust, a separate catalyst, back pressure and temperature monitors, and an Electronic Control Module (ECM).
- Depending on whether or not a catalyst is used to help oxidize collected PM out of the filter, ADPF systems may be usable with fuel that has up to 500 ppm sulfur. Fully active systems that raise the exhaust temperature above 600°C to oxidize collected carbon can operate on the higher sulfur fuels. However, systems that incorporate a catalyst will be limited to operation on fuel with less than 50 ppm sulfur in the same way that DPFs are. Most systems designed for use with onroad vehicles are expected to include a catalyst and will require the lower sulfur fuels that are due to become readily available for onroad vehicles beginning in late 2006.
- Purchase cost for early retrofit devices is expected to be comparable to current DPF prices: $6,500–$10,000 for a typical truck or bus engine (250–350 hp); larger engines generally require larger, more expensive devices, and cost is roughly proportional to engine horsepower. Costs for nonroad engines will be higher than costs for onroad engines of the same horsepower because of their higher baseline emissions levels. As manufacturing volume increases to accommodate the installation of ADPFs on all new trucks after 2007, this price is expected to fall.
- ADPFs incur a fuel economy penalty of 4–7%.

Flow-Through Filter

A flow-through filter (FTF) is very similar to a DOC, but it uses a different type of substrate to hold the catalyst material. Different manufacturers use wire mesh, wire fleece, or sintered metal, all coated with a precious metal catalyst and packaged into metal containers similar to those used to package a DOC.

Figure B-3. Typical Flow-Through Filter.

Courtesy of Fleetguard Emission Solutions

As in a DOC, the catalyst promotes the oxidation of unburned carbon, HC, and CO in the exhaust, producing CO_2 and water. Because of the substrate formation, individual PM particles typically have greater opportunity to contact a catalyst site than in a typical DOC, so that an FTF eliminates a greater percentage of PM than a typical DOC, including a larger percentage of elemental carbon.

FTFs reduce HC and CO emissions by 50-89% and reduce PM emissions by approximately 50%. PM reductions include both the organic and elemental carbon portions. When applied to older onroad trucks and buses, FTFs can reduce total annual PM b*y* 8-33 pounds per vehicle depending on duty cycle. These reductions can be achieved for $40,000–*$*200,000/ton; $/ton costs will be higher for newer vehicles and vehicles that are used less. When applied to older nonroad equipment such as construction vehicles, FTFs can reduce total annual PM by 55–110 pounds per vehicle, and these reductions can be achieved for $20,000–$60,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles because they have much lower baseline emissions levels. Annual reductions from locomotives can be as high as 850 pounds per vehicle due to their larger engine size, higher baseline emissions, and greater annual usage. The cost-effectiveness for locomotives is similar to that for nonroad construction equipment.

The following implementation issues and costs apply to the installation and operation of FTFs:

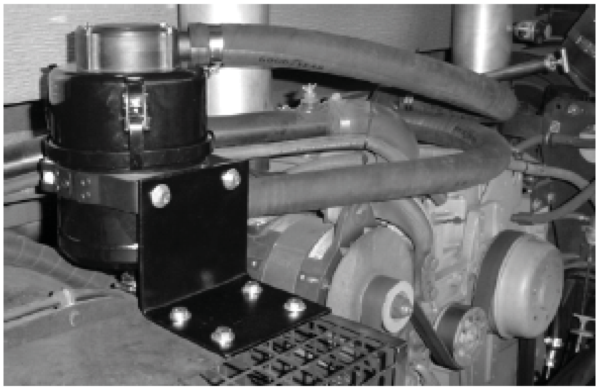
- FTFs, if properly sized and made with an appropriate catalyst formulation, can be used on a greater range of diesel engines than passive DPFs.
- Like passive DPFs, FTFs require a minimum exhaust temperature to work consistently, and therefore cannot be used in all duty cycles.
- Retrofit installation is generally straightforward and takes only a few hours.
- Fuel should not have more than 500 ppm sulfur. With higher sulfur fuels, excessive sulfate PM is created across the catalyst.
- Purchase cost is $3,000–$4,000 for a typical truck or bus engine; larger engines generally require larger, more expensive devices, and cost is roughly proportional to engine horsepower. Costs for nonroad engines will be higher than costs for onroad engines of the same horsepower because of their higher baseline emissions levels.
- FTFs can incur a fuel economy penalty of 1-2%.

Closed Crankcase Filter System

Most current diesel engines have "open" crankcases, which are vented to the atmosphere to relieve any pressure that builds up as combustion gases leak from the cylinders into the crankcase. While the flow from these vents is generally much smaller than exhaust flow from the tailpipe, crankcase vent emissions include combustion gases, unburned fuel, various HC vapors, diesel PM, and engine oil droplets.

Figure B-4. Closed Crankcase Filter.

Courtesy of Donaldson Company, Inc.



Under current EPA standards and test procedures, these crankcase vent gases are not included in exhaust emissions limits. Nonetheless, they can be significant. Testing conducted under the EPA Verification Program on a 1998 onroad diesel engine showed that PM emissions from the crankcase vent totaled 16–23% of the PM emissions coming from the tailpipe (RTI/SWRI 2003).

A closed crankcase system routes gases from the crankcase vent to the engine's air intake manifold, effectively "closing" the crank case. These systems generally include a pressure-regulating valve and an oil filter to keep oil in the vent gases from fouling the engine's turbocharger.

By themselves, closed crankcase filter systems do not reduce vehicle emissions. However, when used in conjunction with other devices that treat exhaust coming from the tailpipe (DOC, DPF, FTF), they can reduce PM emissions by an additional 5-8%. They have also been shown to have an even greater positive effect on the PM exposure of vehicle passengers in some cases, particularly on school buses (Hill, Zimmerman, and Gooch 2005). When applied to older onroad trucks and buses, closed crankcase filter systems can reduce total annual PM by 1–5 pounds per vehicle depending on duty cycle. These reductions can be achieved for $14,000– $200,000/ton; $/ton costs will be higher for newer vehicles and vehicles that are used less. When applied to older nonroad equipment such as construction vehicles, these systems can reduce total annual PM by 6–18 pounds per vehicle, and these reductions can be achieved for $8,000–$60,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles because they have much lower baseline emissions levels. Annual reductions from locomotives can be as high as 140 pounds per vehicle due to their larger engine size, higher baseline emissions, and greater usage. The cost- effectiveness for locomotives is similar to that for nonroad construction equipment.

The following implementation issues and costs apply to the installation and operation of a closed crankcase filter system:

- Closed crankcase systems can be installed on virtually any diesel engine.
- Retrofit installation is generally straightforward and takes only a few hours.
- Removing crankcase vent emissions generally results in a cleaner vehicle engine compartment, and can reduce the PM exposure of vehicle passengers.
- Purchase cost is $200–$500 for a typical truck or bus engine.
- Closed crankcase systems have no effect on fuel economy.

ALTERNATIVE DIESEL FUELS

Emulsified Diesel Fuel

Emulsified diesel fuel (ED) is commercial diesel fuel blended with up to 20% water and a small amount of a proprietary hydrocarbon additive. The additive creates a stable emulsion that will not separate, in which the water molecules are completely enclosed by fuel molecules. This prevents the water from coming into contact with the engine and fuel system components, thereby preventing corrosion and maintaining lubricity. During combustion, evaporation of the water contained in the fuel decreases peak combustion temperatures; this lowers NO_X formation and also results in greater atomization of the fuel, which reduces $PM_{2.5}$ emissions.

ED reduces PM emissions by 16–50%, but may increase CO and HC emissions by as much as 35%. ED can also decrease NO_X emissions by 5–30%. When used in older onroad trucks and buses, ED can reduce total annual PM by 3–33 pounds per vehicle, depending on duty cycle. These reductions can be achieved for $95,000– $290,000/ton; $/ton costs will be higher for newer vehicles due to their lower baseline emissions. When used in older nonroad equipment such as construction vehicles, ED can reduce total annual PM by 18– 110 pounds per vehicle, and these reductions can be achieved for $20,000– $65,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles. Annual reductions from locomotives can be as high as 430 pounds per vehicle due to their larger engine size, higher baseline emissions, and greater use. However, unlike the situation for retrofit devices, the use of ED (and other alternative fuels) has much higher $/ton costs for locomotives than for nonroad construction equipment ($160,000– $260,000/ton). This is due to the much greater use of locomotives, which makes the retrofit devices more cost-effective than the fuel options.

The following implementation issues and costs apply to ED:

- ED can be used in virtually any diesel engine without modification.
- Creation of the emulsion requires specialized blending equipment. Blending can be done at the fuel supplier’s terminal or at a customer site.
- Emulsified fuel is stable, but it can stratify if left in a tank for an extended period without use.

- Because of its water content, emulsified fuel has 10–30% lower effective energy content per gallon than standard diesel, resulting in lower fuel economy (miles per gallon).
- Since fuel is typically metered into an engine's cylinder by volume, the use of emulsified fuel will result in a 10–30% reduction in power output at the same throttle setting.
- ED typically costs $0.01 per gallon more than the base diesel fuel. In addition, typical vehicles will use 10–20% more fuel to do the same amount of work.

Biodiesel Fuel

Biodiesel is a renewable fuel with high oxygen content and low sulfur content. It is composed of mono-alkyl esters and long-chain fatty acids derived from vegetable oils or animal fat. Biodiesel has very similar properties to petroleum diesel.

Oxygenates in biodiesel lower PM emissions by supplying additional combustion oxygen. This can also increase peak combustion temperatures, resulting in additional NO_X formation. Biodiesel can be used neat, but is typically used as a blend with petroleum diesel. The most common form is a B20 blend of 20% biodiesel and 80% petroleum diesel.

Biodiesel fuel reduces HC and CO emissions by 0–50% and PM emissions by 0–50%, depending on the percentage of biodiesel in the fuel blend, but higher blend levels may increase NO_X emissions by as much as 10%. A B20 blend can reduce PM emissions by 10–20%. As a renewable fuel, net fuel cycle CO_2 emissions from biodiesel are also lower than those from petroleum diesel. When used in older onroad trucks and buses B20 can reduce total annual PM by 2–13 pounds per vehicle depending on duty cycle. At current commercial pricing in the Northeast, these reductions can be achieved for $160,000–$300,000/ton; $/ton costs will be higher for newer vehicles due to their lower baseline emissions. When used in older nonroad equipment such as construction vehicles, B20 can reduce total annual PM by 11–44 pounds per vehicle, and these reductions can be achieved for $35,000–$75,000/ton in many cases. As with onroad vehicles, total reductions will be lower and $/ton costs will be higher for newer vehicles. Annual reductions from locomotives can be as high as 172 pounds per vehicle due to their larger engine size, higher baseline emissions, and greater use. However, unlike the situation for retrofit devices, the use of B20 (and other alternative fuels) has much higher $/ton costs for locomotives than for nonroad construction equipment ($200,000–$300,000/ton). This is due to the much greater use of locomotives, which makes the retrofit devices more cost-effective than the fuel options.

The following implementation issues and costs apply to the use of biodiesel fuel:

- Biodiesel can be used in virtually any diesel engine with no or minimal modifications.
- Biodiesel is a robust solvent that can loosen accumulated deposits in a vehicle fuel system. When switching to biodiesel, additional fuel filter changes are usually required.

- While biodiesel can be used neat (B100), it is generally recommended that biodiesel not constitute more than 20% of the total fuel mix. This is because a B20 blend achieves much of the potential PM reduction benefit while minimizing potential NO_X emissions increases.
- Recently, some engine manufacturers have indicated that the use of biodiesel blends with greater than 5% biodiesel will affect their engine warranties, due to concerns over the effect of biodiesel on fuel injectors.
- According to the latest Department of Energy alternative fuel cost survey, in September 2005 biodiesel (B20) cost on average \$3.07 per gallon in the Central Atlantic states, which includes New York (DOE 2005). During the cost survey, standard petroleum diesel sold at the same or nearby stations cost, on average, \$2.83 per gallon. A more extensive survey of commercial diesel fuel prices in the Central Atlantic states by the Energy Information Administration for the same time period determined that standard diesel fuel cost, on average, \$2.88 per gallon. Therefore, in September 2005 in the mid-Atlantic region biodiesel cost \$0.19–\$0.24 per gallon more than standard diesel.

Oxygenated Diesel Fuel

Oxygenated diesel fuel is a blend of standard petroleum diesel fuel with a small amount of an alcohol (up to 10%) and a proprietary hydrocarbon co-solvent additive that keeps the alcohol from separating out of the diesel. In a diesel engine, the alcohol provides increased combustion oxygen—similar to biodiesel and with similar results. Ethanol is low in reactivity and high in oxygen content, making it the preferred oxygenate, but methanol can also be used.

Oxygenated diesel fuel oxy-diesel reduces HC and CO emissions by 0–50% and PM emissions by 10–20%, but it may increase NO_X emissions by as much as 10%. If ethanol (a renewable fuel) is used as the oxygenate, net fuel cycle CO_2 emissions from oxygenated diesel will be marginally lower than those from petroleum diesel. When used in older onroad trucks and buses, oxy-diesel can reduce total annual PM by 2–13 pounds per vehicle depending on duty cycle. These reductions can be achieved for \$100,000–\$200,000/ton; \$/ton costs will be higher for newer vehicles due to their lower baseline emissions. When used in older nonroad equipment such as construction vehicles, oxy-diesel can reduce total annual PM by 11–44 pounds per vehicle, and these reductions can be achieved for \$24,000–\$48,000/ton in many cases. As with onroad vehicles, total reductions will be lower and \$/ton costs will be higher for newer vehicles. Annual reductions from locomotives can be as high as 172 pounds per vehicle due to their larger engine size, higher baseline emissions, and greater use. However, unlike the retrofit devices, the use of oxy-diesel (and other alternative fuels) has much higher \$/ton costs for locomotives than for nonroad construction equipment (\$140,000–\$190,000/ton). This is due to the much higher annual usage of locomotives, which makes the retrofit devices more cost-effective than the fuel options.

The following implementation issues and costs apply to the use of oxygenated diesel fuel:

- Oxygenated diesel can be used in virtually any diesel engine with no modifications.
- The alcohol increases the vapor pressure inside a closed fuel tank above that of typical diesel fuel, to levels similar to gasoline.
- Oxygenated diesel typically costs \$0.05–\$0.15 per gallon more than the base diesel fuel. While the per gallon cost of the oxygenate is similar to the cost of diesel fuel, there is always an incremental cost for handling and delivery associated with the use of specialty fuels, especially if they are used in small volumes for pilot or demonstration programs.

Fuel-Borne Catalysts

When metal-based powdered catalysts in a liquid suspension are added to diesel fuel in very low concentrations, they can promote more complete combustion in a diesel engine, thus potentially reducing emissions of both NO_X and PM. Various companies sell proprietary catalyst packages, which may include small amounts of platinum, cerium, other precious metals, or iron compounds.

At present, neither EPA nor CARB has verified a fuel-borne catalyst as a stand-alone product, though several are included as part of verified systems when used in conjunction with a DOC or FTF. Based on the verification results, the use of a fuel-borne catalyst appears to provide additional benefits beyond those achieved by the retrofit devices alone.

Fuel treated with a fuel-borne catalyst reduces HC and CO emissions by 0-50% and PM emissions by 10–30%. It can also decrease NO_X emissions by as much as 10%. When used in older onroad trucks and buses, fuel-borne catalysts can reduce total annual PM by 2–20 pounds per vehicle depending on duty cycle. These reductions can be achieved for \$18,000–\$53,000/ton; \$/ton costs will be higher for newer vehicles due to their lower baseline emissions. When used in older nonroad equipment such as construction vehicles, fuel-borne catalysts can reduce total annual PM by 11–66 pounds per vehicle, and these reductions can be achieved for \$4,000–\$13,000/ton in many cases. As with onroad vehicles, total reductions will be lower and \$/ton costs will be higher for newer vehicles. Annual reductions from locomotives can be as high as 260 pounds per vehicle due to their larger engine size, higher baseline emissions, and greater use. However, unlike the situation for retrofit devices, the use of fuel-borne catalysts (and other alternative fuels) has much higher \$/ton costs for locomotives than for nonroad construction equipment (\$35,000–\$50,000/ton). This is due to the much greater use of locomotives.

The following implementation issues and costs apply to the use of fuel-borne catalysts:

- Fuel-borne catalysts can be used in virtually any diesel engine with no modifications.

- Some catalyst metals such as cerium pose inhalation hazards; therefore, because metals are increasingly implicated in PM health effects, fuel-borne catalysts that contain these metals should only be used in conjunction with a high efficiency DPF.
- While catalyst formulations can theoretically be added by the operator directly to the fuel tank on individual pieces of equipment, this is not recommended. In order to better monitor and control dosage, on-board or at-the-pump dosing systems should be used, or the fuel supplier can add catalyst to bulk fuel.
- Fuel-borne catalysts typically add \$0.05–\$0.06 per gallon to fuel costs.
- Some manufacturers claim that fuel-borne catalysts can increase fuel economy by up to 5%, but these claims have not been confirmed through EPA verification testing.

IDLE REDUCTION TECHNOLOGIES

Engine Idle Management

The electronic control module (ECM) in an electronically controlled engine can perform a number of functions, including automatically shutting down the engine after it has been idling for a certain period of time. While this does not address idling required to supply auxiliary loads, automatic shutdown can complement efforts to educate vehicle operators to eliminate truly unnecessary idling. This capability is standard on virtually all modern, electronically controlled engines, but it must be turned on by a maintenance technician using software provided by the engine manufacturer. Some engine manufacturers sell a slightly more sophisticated system that can also restart the vehicle as required to maintain engine coolant and/or cab temperatures.

Some systems are available for retrofit on locomotive engines without electronic control. These systems have their own ECM as well as various sensors to monitor the engine and other vehicle systems. The systems are designed to shut down the engine automatically and to restart it as required to maintain appropriate engine coolant temperature, brake air pressure, battery voltage, and other needs.

For Class 8 trucks and buses, every hour of eliminated idling can save on average 0.8 gallons of fuel and avoid 3.7 grams of PM emissions (EPA 2002; EPA 2004e). Engine shutdown/restart systems for onroad trucks and buses can reduce total annual PM by approximately 3 pounds per vehicle. These reductions can be achieved at a net cost savings of over \$800 per year per vehicle based on reduced fuel use, after a 1 to 2 year payback period for the additional equipment required.

For switchyard locomotives, every hour of eliminated idling can save on average 3 to 11 gallons of fuel, depending on the ambient temperature, and can avoid 26 to 32 grams of PM emissions (EPA 2004f; EPA 2005). Engine shutdown/restart systems for switcher yard locomotives can reduce total annual PM by approximately 33 pounds per year per locomotive. These reductions can be achieved at a net cost savings of over $7,000 per year per locomotive based on reduced fuel use, after a 0.7 to 1.5 year payback period for the additional equipment required.

The following implementation issues and costs apply to the use of engine idle management systems:

- Automatic shutdown systems will not cause vehicles to stall in traffic because they are programmed not to shut down the engine with the vehicle in gear.
- Automatic shutdown on electronically controlled engines does not require any new equipment to be added to the vehicle. Basic automatic idle shutdown is available on electronically controlled engines at no extra cost.
- Shutdown/restart systems for nonelectronically controlled engines require installation of an ECM and sensors.
- Shutdown/restart systems cost $1,000–$1,500 for onroad Class 8 trucks and $5,000–$10,000 for locomotives.[50]

Electrically Driven Auxiliary Systems

Some vehicle auxiliary loads are driven directly by the engine (e.g., cab heating), while others are supplied electrically but are indirectly powered by the engine through the engine-driven alternator (e.g., air conditioning). Various companies sell equipment that will allow engine-driven loads to be driven electrically and/or allow electrical loads to be supplied even with the engine off. Some of these technologies rely on the engine's existing starter battery to supply the required electrical power, while others include additional supplementary batteries for longer service time.

Electrically driven auxiliary systems for onroad trucks with sleeper cabs can reduce total annual PM by approximately 11 pounds per vehicle. These reductions can be achieved at a net cost savings of almost $3,000 per year per vehicle based on reduced fuel use, after a 0.2–1.4 year payback period for the additional equipment required.

The following implementation issues and costs apply to the use of electrically driven auxiliary systems:

- In general, equipment must be added to the vehicle. Space requirements and installation time vary significantly by system.
- Costs are $500–$4,000 for onroad Class 8 trucks, depending on whether heating or air conditioning loads are covered and whether a supplementary battery is provided.

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[50] Information on costs for all idling reduction technologies is from the EPA Web site: www.epa.gov/otaq/smartway/idlingtechnologies.htm.

Auxiliary Power Units

The most versatile way to power various vehicle auxiliary loads with the main engine off is to use an auxiliary power unit. These units combine a small diesel engine and generator to provide electrical power for heating, air conditioning, pumps, lighting, and other electrical loads. The engine is sized to match the required load so that using these systems is much more efficient than idling the vehicle's much larger main engine.

In some cases, auxiliary power devices are stand-alone units that provide heated/cooled air directly to the vehicle cab, as well as a place to plug in standard electrical equipment. In other cases, there may be a high degree of integration with existing vehicle heating, air conditioning, lighting, and other systems.

Figure B-5. Auxiliary Power Unit.

Courtesy Pony Pack, Inc.

For Class 8 trucks, the use of an auxiliary power unit instead of the main engine to power vehicle accessories reduces total annual PM emissions by approximately 15 pounds per vehicle. These reductions can be achieved at a net cost savings of over $3,500 per year per vehicle based on reduced fuel use, after a 1.5–2.5 payback period for the additional equipment.

Auxiliary power units for switchyard locomotives can reduce total annual PM by approximately 90 pounds per year per locomotive. These reductions can be achieved at a net cost savings of almost $20,000 per year per locomotive based on reduced fuel use, after a 1.5–2 year payback period for the additional equipment.

The following implementation issues and costs apply to the use of auxiliary power units:

- Equipment must be added to the vehicle. Space requirements, installation time, and degree of integration with existing vehicle systems vary significantly.
- Costs are $6,000–$9,000 for onroad Class 8 trucks and $30,000–$40,000 for locomotives.

Supplementary Diesel Heating Systems

Cab heat and/or heat to maintain engine coolant temperature for easy starting can be supplied by a diesel fuel-fired heater (similar in design to those used for residential heating, but much smaller). The vehicle's battery provides power for the heater's fuel pump. Heated air can be supplied from the heater directly to the cab, or the heater can be used to heat the engine coolant, and cab heat can be supplied through the vehicle's normal heating system. Some systems include timers for automatic starting as well as thermostats to maintain specific cab temperatures.

For Class 8 trucks, the use of a supplementary diesel heating system in lieu of idling the main engine to keep the engine and cab warm can reduce total annual PM by approximately five pounds per vehicle. These reductions can be achieved at a net cost savings of over $1,200 per year per vehicle based on reduced fuel use, after a 0.75–2.5 year payback period for the additional equipment.

Supplementary diesel heating systems for switcher yard locomotives can reduce total annual PM emissions by approximately 30 pounds per year per locomotive. These reductions can be achieved at a net cost savings of over $7,000 per year per locomotive based on reduced fuel use, after a 2–3 year payback period for the additional equipment.

The following implementation issues and costs apply to the use of supplementary diesel heating systems:

- Installation is generally straightforward, and required fuel is supplied from the vehicle's existing fuel tank.
- Costs are $900–$3,000 for onroad Class 8 trucks, and $15,000–$20,000 for locomotives.

Grid Power Systems

The electrical power required for on-vehicle auxiliary loads can be supplied from the electrical grid rather than being generated onboard. However, this always requires that infrastructure be installed at locations where vehicles congregate (e.g., trucks stops/rest stops). Some systems require only this fixed infrastructure to provide heated/cooled air to the truck cab, along with lighting and a place to plug in electrical equipment. These services are usually bundled with other amenities, such as telephone and cable TV service.

Figure B-6. Shore Power Idle Reduction System.

Courtesy IdleAire Technologies, Inc.

Another approach is to supply electrical power to the vehicle from fixed infrastructure as a means of providing heating/cooling using the truck's existing on-board systems. In this case, modifications must also be made to the truck, so that its existing electrical systems can accept grid-supplied power.

Power plug-in units can also be designed for locomotives; these generally require both fixed infrastructure and modifications to the vehicle itself. This is an approach that may be well suited to commuter trains and yard switchers, but it is unlikely to be effective for many long-haul locomotives.

The use of grid power systems for onroad trucks with sleeper cabs can reduce total annual PM emissions by approximately 8 pounds per year per vehicle. These reductions can be achieved at a net cost savings of over $2,000 per year per vehicle based on reduced fuel use after less than a one-year payback period for the additional on-truck equipment. At current diesel fuel prices, vehicle operators who purchase heat/electricity and other services from operators who have outfitted special parking spaces at truck stops rather than idling their engines can save $0.50–$1.50 per hour, or up to $3,000 per year.

The following implementation issues and costs apply to the use of grid power systems:

- In all cases, fixed infrastructure must be installed at locations where truckers are likely to stop to rest, or at the parking tracks for locomotives.
- Depending on the system, modifications may be necessary to enable the truck to accept and use power from the grid.
- Truck modifications (if required) cost $100–$2,000 per vehicle.
- The cost of installing fixed infrastructure can be $5,000 or more per parking space. Generally, this cost is recovered through the charges levied on truckers for use of the system. Depending on what services are provided, these charges can range from $0.50–$1.50 per hour, compared to $2.00 per hour in fuel costs to idle the truck's main engine.

Hybrid Electric Locomotives

Typical locomotives are propelled by electric motors. Electricity to power these propulsion motors is produced onboard using a very large diesel engine coupled to a generator. The diesel engine is sized to provide the peak power needed to start the locomotive from a stop and to pull large loads up an incline.

Unlike long-haul locomotives, whose diesel engines work at or close to peak load for much of the time, switcher yard locomotives operate in short bursts. Their diesel engines spend much of the time at idle or working at very low loads. This type of duty cycle is appropriate for a "hybrid electric" drive system, in which the diesel engine is sized for the average load (and can therefore be much smaller), while peak loads are provided by a battery pack.

At least one company has developed such a hybrid switcher locomotive. It uses a 125 horsepower diesel engine (smaller than the engines used in Class 8 trucks) and 60,000 pounds of lead-acid batteries. The propulsion motors are powered by the batteries, and the diesel engine continuously charges the batteries as they become depleted. This configuration is more efficient than using a large diesel engine and completely eliminates unproductive engine idling.

The use of a hybrid electric switchyard locomotive can reduce total annual PM emissions by approximately 110 pounds per year compared to a standard locomotive. These reductions can be achieved at a net cost savings of almost $25,000 per year per locomotive based on reduced fuel use, after an eight-year payback period.

The following implementation issues and costs apply to the use of hybrid electric locomotives:

- Batteries are expected to last over 10 years, but this has not been demonstrated.
- Existing locomotives could be converted in lieu of an engine rebuild.
- Conversion of an old locomotive costs approximately $200,000, which is generally competitive with engine rebuilds, assuming that acceptable battery life can be achieved (Argonne 2005).

Dock Electrification

Some ships, while docked in port, have large electrical loads that are supplied by onboard diesel-driven auxiliary generators. If this power were provided instead from the land-side electrical grid, net PM, SO_X, and NO_X emissions would be reduced significantly in most cases. The U.S. Navy pioneered this approach, which is often referred to as “cold ironing.”

Dock electrification requires both land-side infrastructure and expensive modifications to the ships themselves. This approach is not practical for all ships, but it can be cost-effective for individual vessels with a large number of long-duration port calls in the same location and high dock-side electrical loads. The ship types most likely to be good candidates for dock electrification include cruise ships, refrigerated container vessels, and some large tankers, which can have dockside auxiliary loads of 600 to 7,000 kilowatts (Port of Long Beach 2004).

An analysis of 12 ships that called at the Port of Long Beach in 2002 showed that their annual dockside PM emissions ranged from 0.1–9.7 tons/year per ship. The use of shore power in lieu of onboard electricity generation can reduce these PM emissions by 83–97%. In the case of the highest emitter in the Port of Long Beach study, dock electrification could have reduced annual PM emissions by more than nine tons from a single ship. The cost of these reductions would have been approximately $200,000/ton over a 10-year period if all costs were attributed to the PM reductions. However, dock electrification would have also reduced NO_X emissions by 85 tons/year and SO_X emissions by 80 tons/year from this ship. The average cost of all emissions reductions would have been approximately $11,000/ton (Port of Long Beach 2004).

The following implementation issues and costs apply to dock electrification:

- Installation of shore-side equipment can cost $5 million or more, and ship retrofit costs can average approximately $500,000 per vessel (Port of Long Beach 2004).
- Given the low cost of marine distillate fuels, the typical cost of onboard electricity generation is lower than the cost of electricity supplied from the land-side grid.
- The Port of Los Angeles has developed specifications for both the ship-side and shore-side equipment necessary for cold ironing, which it calls Alternative Marine Power (AMP).

Appendix C

COST-EFFECTIVENESS ANALYSIS: RETROFIT, FUEL, AND IDLE REDUCTION OPTIONS FOR VEHICLES

INTRODUCTION

Relatively little comparative data has been published on the cost-effectiveness of various retrofit and fuel technologies to reduce fine particulate emissions from diesel vehicles. Moreover, comparing results from different published sources can be difficult because of the varying assumptions used for the calculations. In order to evaluate the relative merits of various emissions reduction strategies, the authors developed a spreadsheet model to calculate the costs and benefits of five retrofit technologies and four fuel options, as applied to three different diesel vehicle types. In addition, the costs and benefits of six different idle reduction technologies were evaluated, as applied to two different diesel vehicle types.

This Appendix documents the results of the analysis. These results are illustrative only. The cost-benefit estimates rely on a significant number of assumptions about the reduction effectiveness and cost of each technology, as well as baseline emissions and annual usage factors for the different vehicle types. Given that each device can have a range of costs and reduction effectiveness, and that different vehicle types can have a range of baseline emissions, the resulting range of values for cost-effectiveness ($/ton reduction) is in many cases quite broad. The results of this analysis are best used to evaluate general trends, rather than the specific cost-effectiveness of any particular technology or vehicle type.

METHODOLOGY

Technologies

The analysis covers five retrofit technologies, four fuel options, and six idle reduction technologies, as shown in Table C-1. Each of these technologies is described more fully in Appendix B.

Table C-1. Technologies Included in the Analysis.

Fuel Options	Retrofit Technologies	Idle Reduction Technologies
Emulsified Diesel Fuel	Diesel Oxidation Catalyst (DOC)	Engine Shut-down/Restart System
Biodiesel Fuel	Flow-Through Filter (FTF)	Auxiliary Power Unit
Oxygenated Diesel Fuel	Diesel Particulate Filter (DPF)	Supplementary Diesel Heating System
Fuel-borne Catalyst	Active Diesel Particulate Filter (ADPF)	Electrically Driven Accessories
	Closed Crank Case Filter (CCCF)	Grid Power System
		Hybrid Electric Locomotive

Vehicles and Duty Cycles

Each of the retrofit and fuel options was evaluated as applied to three different diesel vehicle types: onroad trucks, nonroad construction equipment, and locomotives. Because emissions standards have become more stringent over time, onroad trucks and nonroad construction equipment were analyzed in two sets: (1) older vehicles with relatively higher baseline emissions levels and (2) newer vehicles with lower baseline emissions. The standards for locomotives have not changed as quickly and, therefore, only one set was used.

Because vehicle usage factors can affect baseline emissions and technology costs significantly, the effects of the technologies were analyzed separately for two duty cycles for each vehicle type. For onroad trucks, a distinction was made between trucks primarily used in urban areas and those primarily operated on highways. Compared to urban driving, highway driving results in different baseline emissions levels (grams per mile), higher total annual vehicle use (miles per year), and higher total annual fuel use (gallons per year).

For nonroad construction equipment, “average usage” and “high usage” cases were analyzed. These were distinguished by differences in the number of hours of annual operation, which affects both total emissions and total fuel use.

Locomotives were analyzed for each of the two main operating cycles seen in typical service: line-haul operation and switchyard operation. Line-haul operation is characterized by significant time at high speed and high engine load, while switchyard operation is characterized by very low speeds and a significant amount of time at idle and at low engine loads.

The different cases analyzed for each technology option are summarized in Table C-2.

Table C-2. Vehicle/Duty Cycle/Age Combinations.

Vehicle Type	Duty Cycle	Age/Emission Standards
Onroad Truck	Highway	1990
		1998
	Urban	1990
		1998
Construction Equipment	High Usage	Tier 0
		Tier 2
	Average Usage	Tier 0
		Tier 2
Locomotive	Line-Haul	Tier 0
	Switchyard	

Note that not every technology analyzed will necessarily be applicable to all vehicles of every type. Many of the assumptions about reduction effectiveness and cost for each technology are based on testing and demonstrations on onroad vehicles. While included in this analysis for nonroad vehicles, very few of these technologies have been successfully demonstrated on nonroad construction equipment and locomotives, and some may never be. See Appendix B for a more complete description of each technology, including constraints and restrictions on use.

For the idle reduction technologies, only two specific vehicle/duty cycle combinations were analyzed: sleeper-cab equipped onroad trucks and switchyard locomotives. These technologies are primarily designed to reduce long-duration idling associated with necessary operation of engine-driven vehicle accessories while a vehicle is parked. Both sleeper-cab trucks and switchyard locomotives operate in this mode for a significant amount of time annually, while other vehicles typically do not.

Cost and Benefit Calculations

For each technology/vehicle/duty cycle/age combination, the potential annual PM reductions (pounds/year) were calculated by applying a percentage reduction factor for the technology to baseline emissions levels for the vehicle. The reduction factors for each technology were taken from technology summaries on the Web site of the EPA Office of Transportation and Air Quality (OTAQ). These factors were derived from test results compiled under the EPA's retrofit Technology Verification Program, and typically they include a range of reductions for each technology. Most of this testing has been conducted on onroad engines and vehicles. With a few exceptions, these same assumptions about reduction effectiveness have also been applied to the nonroad vehicles analyzed.

Baseline annual emissions levels were calculated using a PM emissions factor (grams/mile, grams/gallon, or grams/hour) specific to the vehicle, and relevant annual usage factors (miles/year, gallons/year, or hours/year). PM emissions factors were taken from EPA emissions models, as were many of the assumptions about annual usage.

For each retrofit technology option, the costs attributable to the technology include purchase and installation cost, incremental annual maintenance costs, and incremental annual fuel costs. For the fuel options, the costs attributable to the option include only incremental annual fuel costs. For each option, the costs for onroad vehicles were calculated based on information from technology summaries on the OTAQ Web site and other published sources. These costs were adjusted for application to the nonroad equipment based on larger engine sizes and higher baseline emissions. All of the data sources for every emissions, usage, and cost assumption used in the analysis are noted below in Table C-3 to Table C-15.

For the technology options, the cost analysis was based on an assumed seven-year useful life for each device. To calculate the cost-effectiveness of each technology ($/ton PM reduction), annual PM reductions were summed over the seven-year useful life of the device, and compared to the net present value of capital and incremental operating costs over the same seven-year period.

All of the idle reduction technologies provide financial benefits to the user attributable to reduced fuel use. Therefore, for these technologies a net annual cost savings was calculated, along with the payback period (years) of the initial purchase cost.

RESULTS

The detailed results of the analysis are shown in Table C-3 to Table C-15. These results and general trends are also discussed below; select results are summarized in Figure C-1 to Figure C-11, which illustrate relevant trends. Please note that all of the values for annual reductions and cost-effectiveness are illustrative only, and actual results could vary depending on vehicle baseline emissions, usage, and actual technology cost. This analysis is intended to determine general trends rather than to establish specific numerical values for annual reductions and cost-effectiveness.

Retrofit Technologies and Fuel Options

The range of potential annual PM reductions available from each technology is shown in Table C-3. The cost-effectiveness of these reductions ($/ton PM reduction) is shown in Table C-4.

As shown, for each technology the range of available reductions varies significantly depending on vehicle type, vehicle age, and duty cycle. This is illustrated graphically in Figure C-1 and Figure C-2, which show the potential reductions available from each vehicle that is retrofitted with a diesel oxidation catalyst (DOC). As shown, the largest reductions, up to approximately 500 lb/yr of PM emissions, are available from retrofit of line-haul locomotives, followed by switchyard locomotives, older (Tier 0) construction equipment, and older (1990) highway trucks. The smallest reductions (less than a pound per year) come from newer onroad trucks—particularly those used in an urban duty cycle.

The relative ranking of available reductions by vehicle type, age, and duty cycle is similar for all technologies and fuel options. In all cases, per-vehicle reductions are 2 to 10 times larger from construction equipment than from onroad trucks of similar age. Reductions from locomotives are 5 to 50 times larger than reductions from even the oldest pieces of construction equipment, and 10 to 500 times larger than from onroad trucks. The higher reductions available from locomotives are based on their much larger engine size, higher baseline emissions levels, and higher annual usage. The reductions available from urban onroad trucks are lower than from highway trucks due to much lower annual usage.

For each vehicle type, available reductions also vary significantly by technology. This is illustrated in Figure C-3, which shows the available reductions from each technology, as applied to a piece of Tier 0 construction equipment with high annual usage. As shown, the greatest reductions are available from retrofit with a diesel particulate filter (DPF), followed by retrofit with a flow-through filter (FTF), and possibly the use of emulsified diesel fuel. The other fuel options provide reductions similar to those associated with the use of a DOC. The smallest reductions come from the use of a closed crankcase filter (CCCF).

Figure C-1. Annual PM Reductions per Vehicle from Retrofit with Diesel Oxidation Catalyst (DOC).

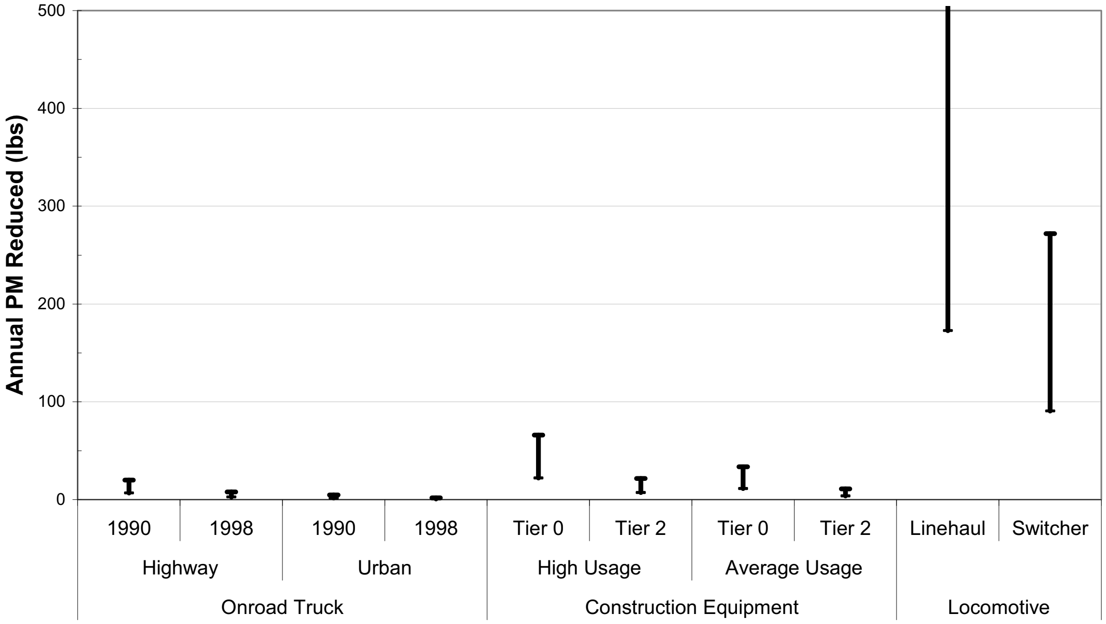


Figure C-2. Annual PM Reductions per Vehicle from Retrofit with Diesel Oxidation Catalyst (DOC).

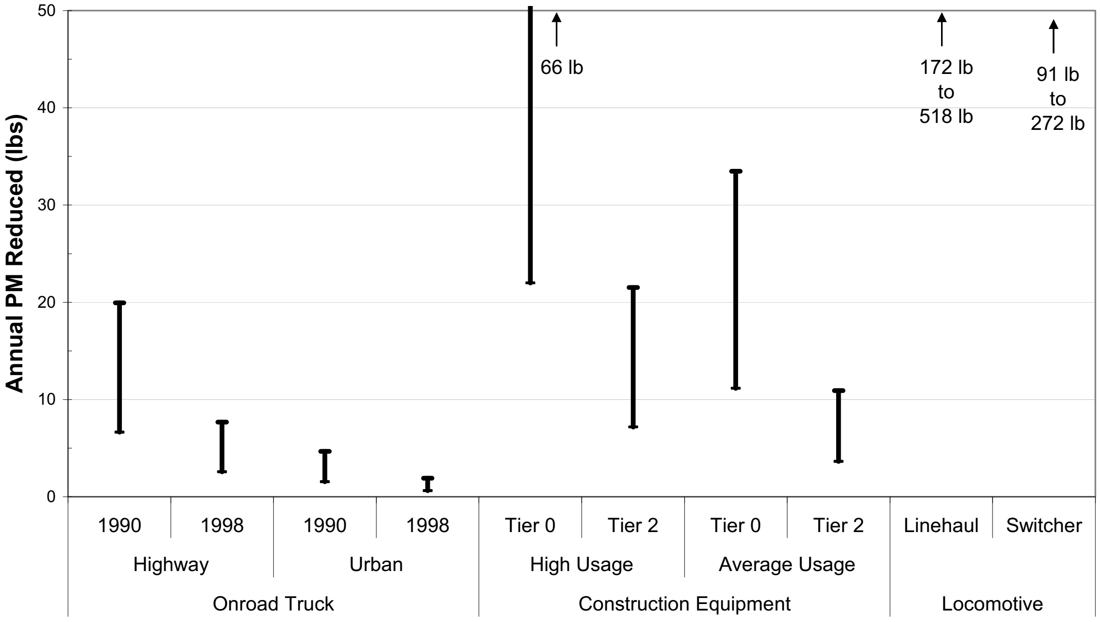
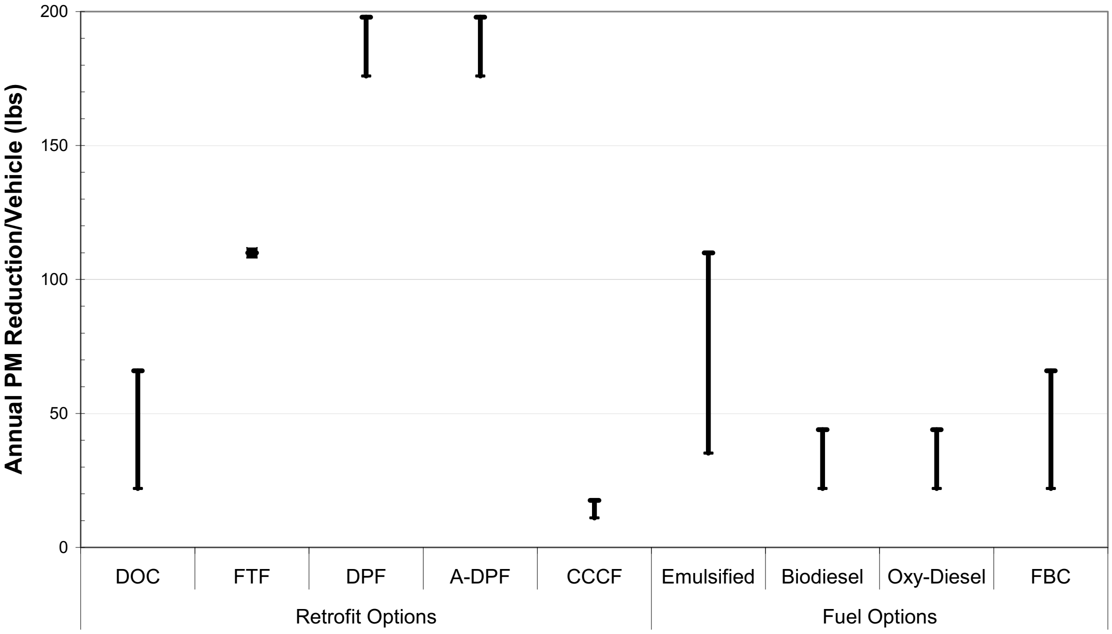


Figure C-3. Annual PM Reductions from Different Technologies Applied to Tier 1 Construction Equipment with High Usage.



The cost-effectiveness of a DOC, DPF, and emulsified fuel as applied to each vehicle/age/duty cycle combination is summarized in Figure C-4 to Figure C-6. As shown, the retrofit technologies (DOC, DPF) have about the same cost-effectiveness when applied to locomotives, older highway trucks, and older construction equipment ($25,000–$125,000/ton for DOCs and $50,000–$100,000/ton for DPFs). The cost-effectiveness is slightly worse when applied to new construction equipment with high usage, and significantly worse when applied to newer onroad vehicles, particularly those operating on an urban duty cycle ($250,000–$1,170,000/ton).

The results are somewhat different for emulsified diesel fuel. This option is most cost-effective for older construction equipment ($20,000–$65,000/ton), and slightly less so for new construction equipment and older onroad trucks. As with the retrofit technologies, it is least cost-effective for newer onroad trucks. As with all of the fuel options, the cost-effectiveness of emulsified diesel is less sensitive to duty cycle than the retrofit options are. Because retrofit options have a significant up-front purchase and installation cost, they are more cost-effective when applied to vehicles that have high annual usage. The fuel options are more of a "pay-as-you-go" emissions reduction solution. The incremental cost of the emissions reduction is paid on each gallon of fuel purchased, and higher annual vehicle use does not significantly improve cost-effectiveness.

Figure C-4. Cost Effectiveness of Diesel Oxidation Catalyst (DOC) Retrofit.

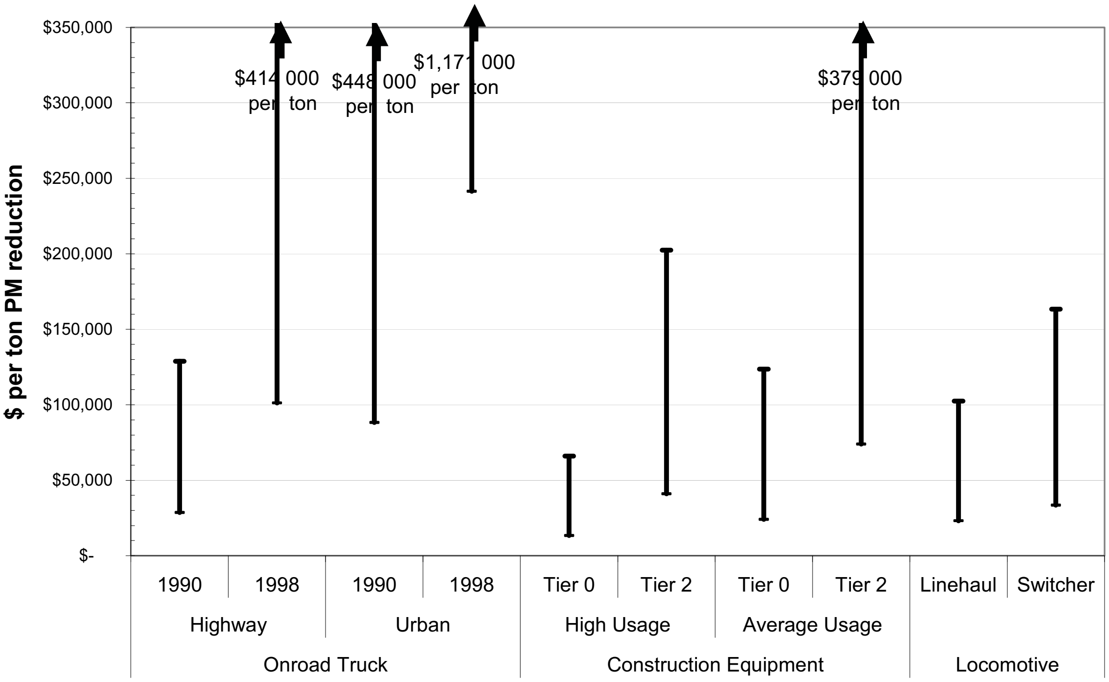


Figure C-5. Cost Effectiveness of Diesel Particulate Filter (DPF) Retrofit.

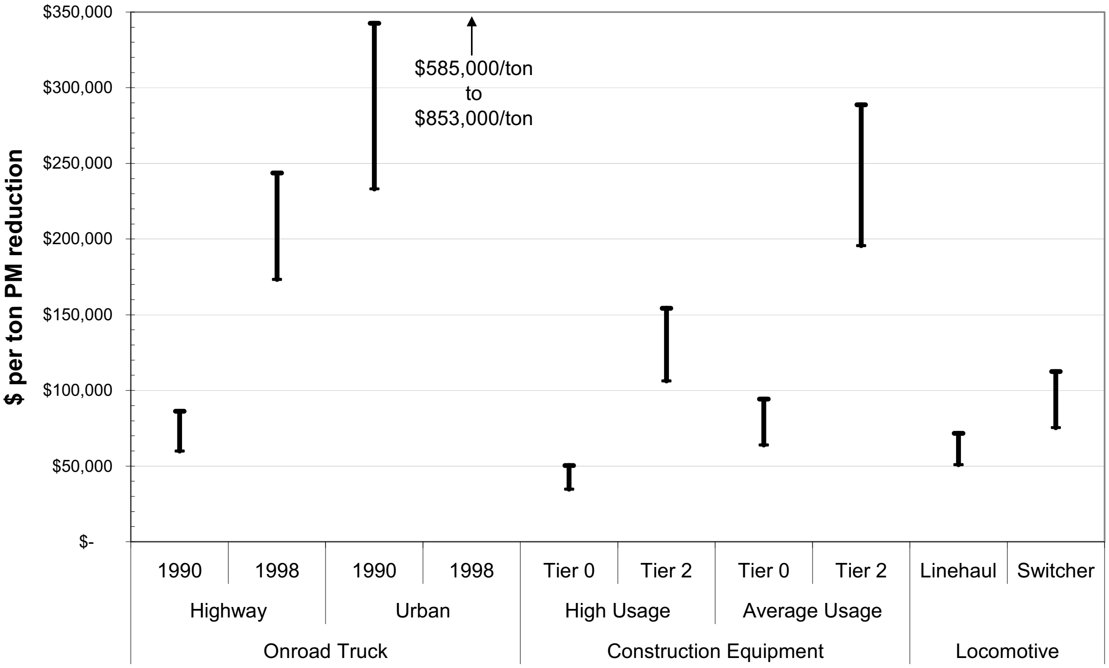
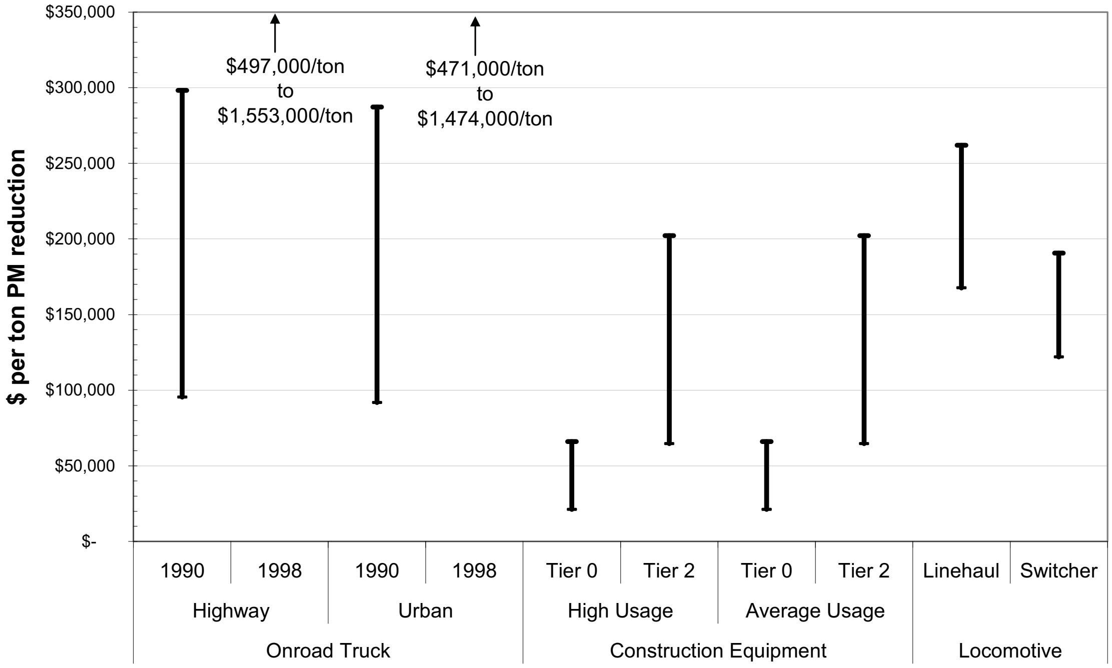


Figure C-6. Cost Effectiveness of Emulsified Diesel Fuel.



The relative cost-effectiveness of each retrofit and fuel option applied to the same vehicle is illustrated in Figure C-7. This chart shows the cost-effectiveness of each technology applied to a piece of Tier 0 construction equipment with high annual usage. As shown, all of the retrofit devices have minimum cost-effectiveness of less than $35,000/ton and significant overlap in their ranges of cost-effectiveness. It is hard to evaluate which of these devices is the "most cost-effective"; in fact, they are all similar.

At current expected pricing, fuel-borne catalysts are quite cost-effective when applied to older construction equipment, with $/ton costs equivalent to, or slightly lower than, the low values for retrofit devices. The other fuel options (emulsified diesel, oxy-diesel, and biodiesel) are somewhat less cost-effective, but all have a range of cost-effectiveness similar to the retrofit options.

Figure C-8 shows the cost-effectiveness of the different technologies applied to line-haul locomotives. For locomotives, the retrofit options are much more cost-effective than the fuel options, for two reasons. First, these locomotives have very high annual utilization to offset the capital cost of the retrofit device. Second, a large portion of PM emissions from locomotives comes from the burning of lube oil. This lube oil PM can be reduced by catalytic retrofit devices (DOC, DPF, ADPF), but it is unaffected by alternative fuels.

Figure C-7. Cost-Effectiveness of Different Technologies of Tier 1 Construction Equipment with High Usage.

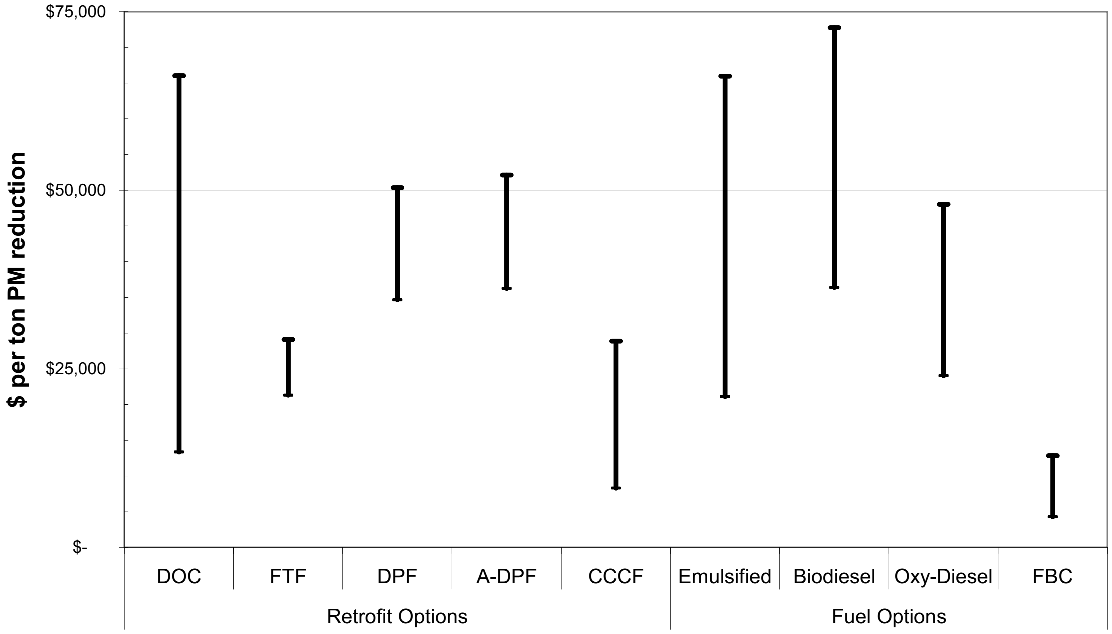
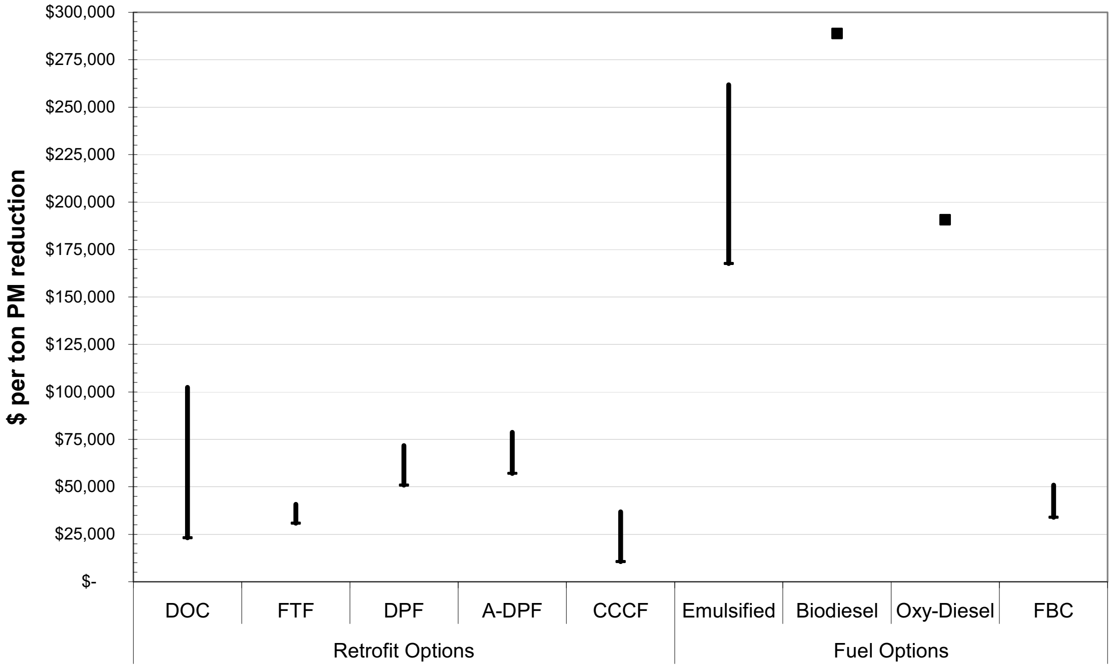


Figure C-8. Cost Effectiveness of Different Technologies on Linehaul Locomotive.



Idle Reduction Technologies

The annual PM reductions available from the use of idle reduction devices applied to switcher locomotives and sleeper-cab equipped onroad trucks are shown in Table C-14. Reductions are shown graphically in Figure C-9. As one might expect, the per-vehicle PM reductions available from switcher locomotives are almost 10 times greater than those from onroad trucks. For each vehicle type, the scale of these annual reductions is similar to those that can be achieved by retrofit of the engine with a DOC.

Because reducing idling reduces total fuel use, these technologies provide a financial payback to the user. These potential cost savings are shown in Figure C-10. As shown, the use of idle reduction devices can save the owner of a sleeper-cab equipped truck \$800–\$2,900 per year depending on the device. The level of savings is directly proportional to the amount of idling reduced. Similarly, the use of idle reduction devices can save the owner of a switcher locomotive \$7,000–\$24,000 per year.

The payback period for these idle reduction devices is shown in Figure C-11. As shown, virtually all of these devices have a payback period of less than three years, with many potentially paying for themselves through fuel savings in significantly less than one year. The only technology with a longer payback period is a hybrid-electric locomotive, which at current pricing has a payback period of eight years. However, the cost of retrofitting this technology is generally comparable to overhaul and rebuilding of traditional diesel locomotive engines. If implemented as part of a normal locomotive overhaul cycle, conversion to hybrid-electric drive may be significantly more cost-effective than shown.

Figure C-9. Annual PM Reductions per Vehicle from Idle Reduction Technologies.

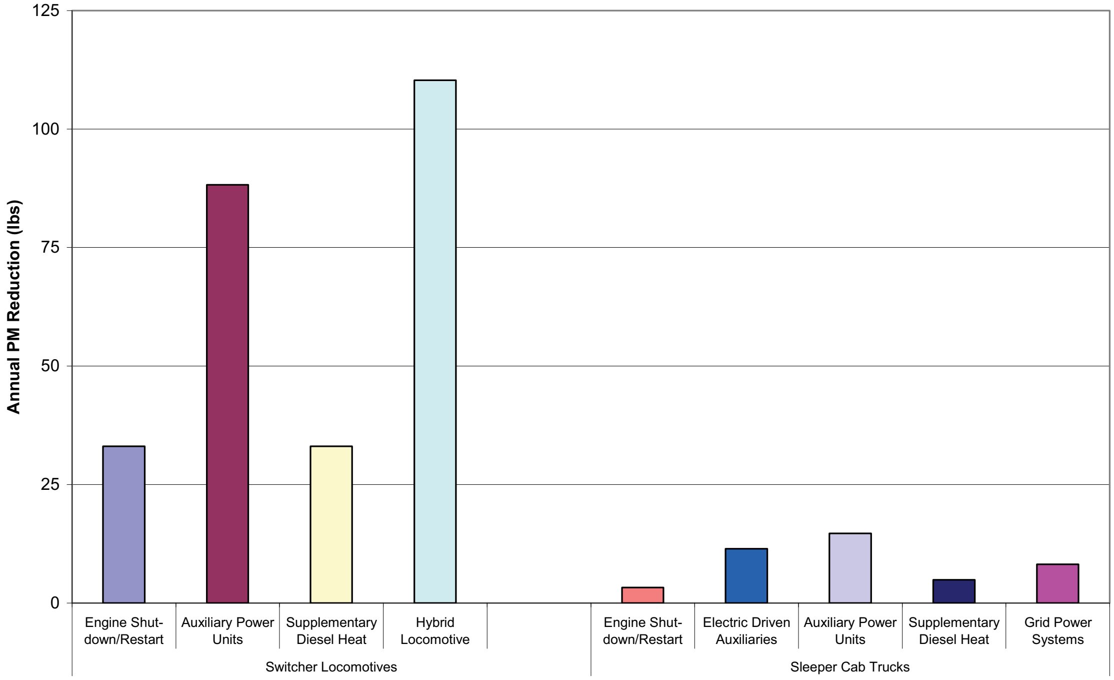


Figure C-10. Annual Fuel Cost Savings per Vehicle from Idle Reduction Technologies.

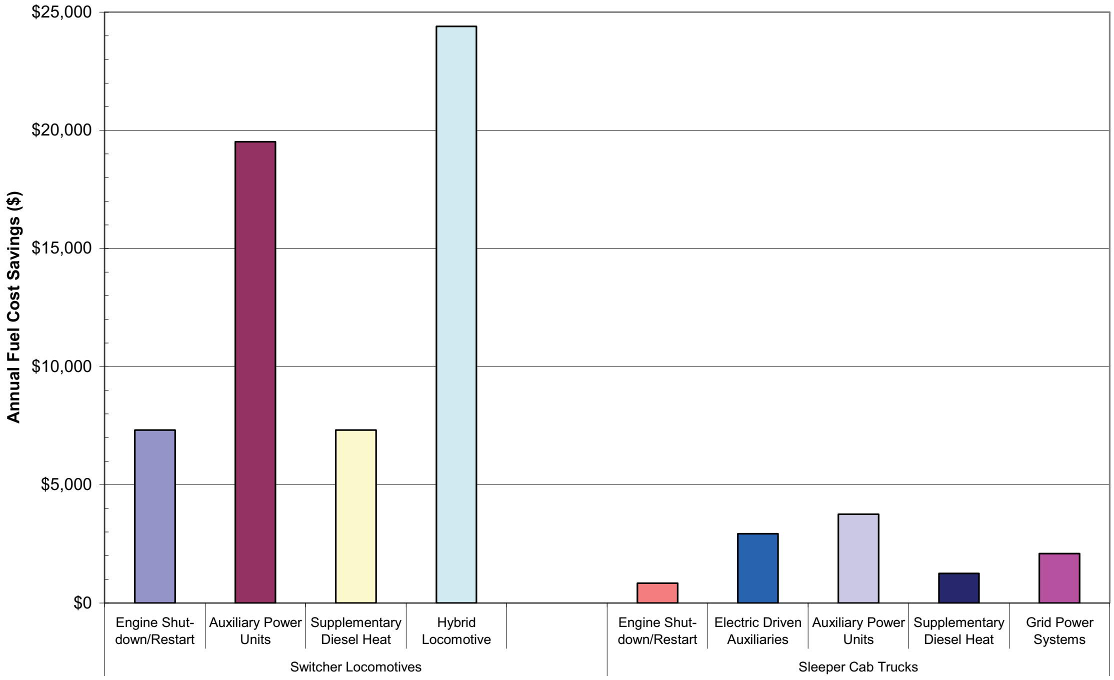


Figure C-11. Payback Period for Idle Reduction Technologies.

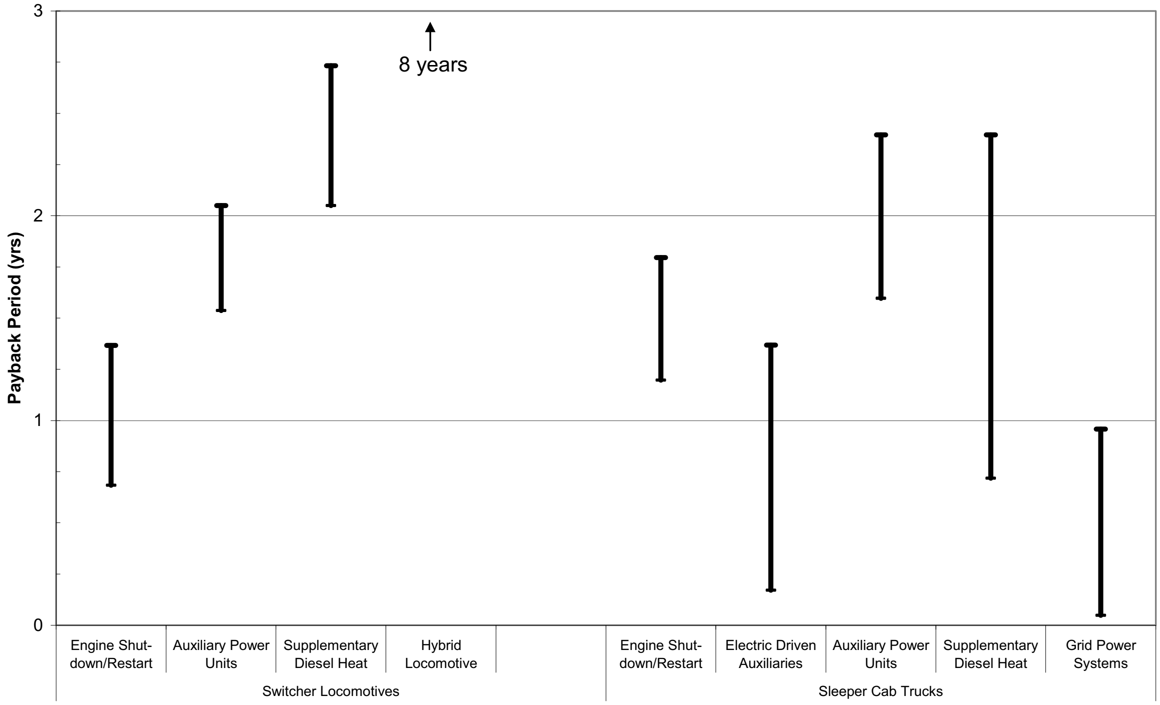
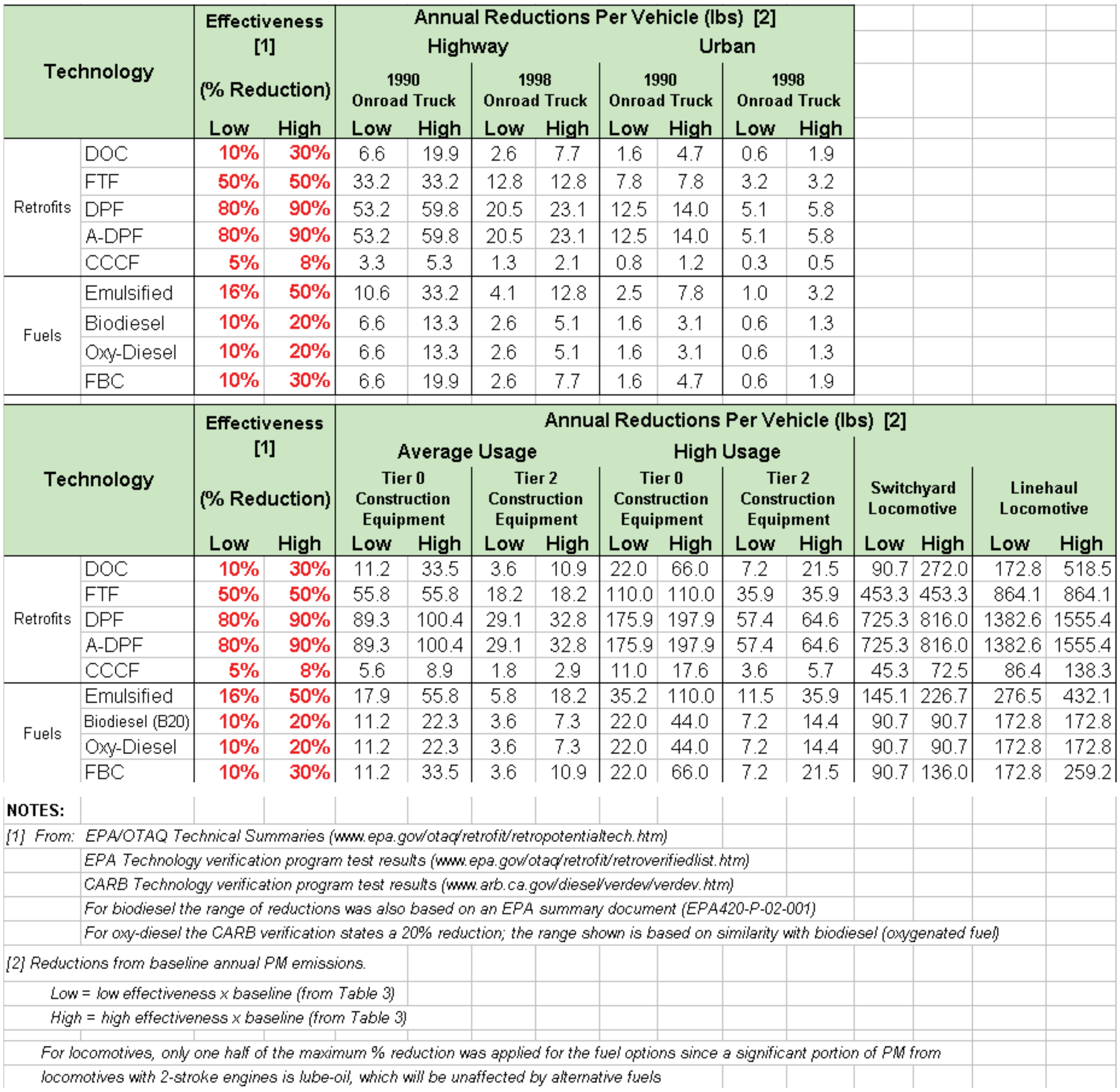


Table C-3. Potential Emissions Reductions – Retrofits & Fuel Options.



Technology		Effectiveness [1] (% Reduction)		Annual Reductions Per Vehicle (lbs) [2]							
				Highway				Urban			
				1990 Onroad Truck		1998 Onroad Truck		1990 Onroad Truck		1998 Onroad Truck	
		Low	High	Low	High	Low	High	Low	High	Low	High
Retrofits	DOC	10%	30%	6.6	19.9	2.6	7.7	1.6	4.7	0.6	1.9
	FTF	50%	50%	33.2	33.2	12.8	12.8	7.8	7.8	3.2	3.2
	DPF	80%	90%	53.2	59.8	20.5	23.1	12.5	14.0	5.1	5.8
	A-DPF	80%	90%	53.2	59.8	20.5	23.1	12.5	14.0	5.1	5.8
	CCCF	5%	8%	3.3	5.3	1.3	2.1	0.8	1.2	0.3	0.5
Fuels	Emulsified	16%	50%	10.6	33.2	4.1	12.8	2.5	7.8	1.0	3.2
	Biodiesel	10%	20%	6.6	13.3	2.6	5.1	1.6	3.1	0.6	1.3
	Oxy-Diesel	10%	20%	6.6	13.3	2.6	5.1	1.6	3.1	0.6	1.3
	FBC	10%	30%	6.6	19.9	2.6	7.7	1.6	4.7	0.6	1.9

Technology		Effectiveness [1] (% Reduction)		Annual Reductions Per Vehicle (lbs) [2]											
				Average Usage				High Usage							
				Tier 0 Construction Equipment		Tier 2 Construction Equipment		Tier 0 Construction Equipment		Tier 2 Construction Equipment		Switchyard Locomotive		Linehaul Locomotive	
		Low	High	Low	High	Low	High	Low	High	Low	High	Low	High	Low	High
Retrofits	DOC	10%	30%	11.2	33.5	3.6	10.9	22.0	66.0	7.2	21.5	90.7	272.0	172.8	518.5
	FTF	50%	50%	55.8	55.8	18.2	18.2	110.0	110.0	35.9	35.9	453.3	453.3	864.1	864.1
	DPF	80%	90%	89.3	100.4	29.1	32.8	175.9	197.9	57.4	64.6	725.3	816.0	1382.6	1555.4
	A-DPF	80%	90%	89.3	100.4	29.1	32.8	175.9	197.9	57.4	64.6	725.3	816.0	1382.6	1555.4
	CCCF	5%	8%	5.6	8.9	1.8	2.9	11.0	17.6	3.6	5.7	45.3	72.5	86.4	138.3
Fuels	Emulsified	16%	50%	17.9	55.8	5.8	18.2	35.2	110.0	11.5	35.9	145.1	226.7	276.5	432.1
	Biodiesel (B20)	10%	20%	11.2	22.3	3.6	7.3	22.0	44.0	7.2	14.4	90.7	90.7	172.8	172.8
	Oxy-Diesel	10%	20%	11.2	22.3	3.6	7.3	22.0	44.0	7.2	14.4	90.7	90.7	172.8	172.8
	FBC	10%	30%	11.2	33.5	3.6	10.9	22.0	66.0	7.2	21.5	90.7	136.0	172.8	259.2

NOTES:

[1] From: EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm)

EPA Technology verification program test results (www.epa.gov/otaq/retrofit/retroverifiedlist.htm)

CARB Technology verification program test results (www.arb.ca.gov/diesel/verdev/verdev.htm)

For biodiesel the range of reductions was also based on an EPA summary document (EPA420-P-02-001)

For oxy-diesel the CARB verification states a 20% reduction; the range shown is based on similarity with biodiesel (oxygenated fuel)

[2] Reductions from baseline annual PM emissions.

Low = low effectiveness x baseline (from Table 3)

High = high effectiveness x baseline (from Table 3)

For locomotives, only one half of the maximum % reduction was applied for the fuel options since a significant portion of PM from locomotives with 2-stroke engines is lube-oil, which will be unaffected by alternative fuels

Table C-4. Cost Effectiveness of Retrofits and Fuel Options.

Technology		Cost Effectiveness of PM Reductions ($/ton) [1]							
		Onroad Truck, Highway				Onroad Truck, Urban			
		1990 Truck		1998 Truck		1990 Truck		1998 Truck	
		Low	High	Low	High	Low	High	Low	High
Retrofits	DOC	$ 28,626	$ 128,858	$ 101,158	$ 414,887	$ 88,181	$ 448,100	$ 241,409	$ 1,170,981
	FTF	$ 39,166	$ 52,060	$ 117,698	$ 151,122	$ 146,825	$ 201,892	$ 373,426	$ 507,452
	DPF	$ 59,902	$ 86,193	$ 173,246	$ 243,644	$ 233,109	$ 342,554	$ 585,218	$ 853,825
	A-DPF	$ 66,769	$ 93,919	$ 209,016	$ 283,885	$ 239,721	$ 349,992	$ 619,168	$ 892,019
	CCCF	$ 13,744	$ 47,779	$ 35,627	$ 123,850	$ 58,698	$ 204,050	$ 142,863	$ 496,633
Fuels	Emulsified	$ 95,429	$ 298,214	$ 497,084	$ 1,553,387	$ 91,884	$ 287,138	$ 471,791	$ 1,474,346
	Biodiesel	$ 158,949	$ 317,898	$ 827,959	$ 1,655,919	$ 153,045	$ 306,091	$ 785,830	$ 1,571,660
	Oxy-Diesel	$ 105,137	$ 210,274	$ 547,656	$ 1,095,312	$ 101,232	$ 202,464	$ 519,789	$ 1,039,579
	FBC	$ 18,411	$ 55,233	$ 95,901	$ 287,704	$ 17,727	$ 53,181	$ 91,022	$ 273,065

Technology		Cost Effectiveness of PM Reductions ($/ton) [1]							
		Constuction Equipment, High Usage				Construction Equipment, Average Usage			
		Tier 0		Tier 2		Tier 0		Tier 2	
		Low	High	Low	High	Low	High	Low	High
Retrofits	DOC	$ 13,355	$ 66,048	$ 40,925	$ 202,406	$ 24,113	$ 123,558	$ 73,896	$ 378,644
	FTF	$ 21,308	$ 29,103	$ 65,297	$ 89,186	$ 40,673	$ 56,039	$ 124,644	$ 171,731
	DPF	$ 34,648	$ 50,347	$ 106,179	$ 154,290	$ 63,828	$ 94,214	$ 195,601	$ 288,720
	A-DPF	$ 36,225	$ 52,122	$ 111,013	$ 159,728	$ 65,405	$ 95,989	$ 200,436	$ 294,159
	CCCF	$ 8,309	$ 28,885	$ 25,464	$ 88,520	$ 16,378	$ 56,936	$ 50,192	$ 174,481
Fuels	Emulsified	$ 21,105	$ 65,955	$ 64,678	$ 202,119	$ 21,105	$ 65,955	$ 64,678	$ 202,119
	Biodiesel	$ 36,377	$ 72,754	$ 111,478	$ 222,956	$ 36,377	$ 72,754	$ 111,478	$ 222,956
	Oxy-Diesel	$ 24,016	$ 48,031	$ 73,596	$ 147,193	$ 24,016	$ 48,031	$ 73,596	$ 147,193
	FBC	$ 4,275	$ 12,824	$ 13,100	$ 39,299	$ 4,275	$ 12,824	$ 13,100	$ 39,299

Technology		Cost Effectiveness of PM Reductions ($/ton) [1]			
		Locomotive			
		Line Haul		Switcher	
		Low	High	Low	High
Retrofits	DOC	$ 23,132	$ 102,461	$ 33,453	$ 163,385
	FTF	$ 30,797	$ 40,716	$ 52,319	$ 71,228
	DPF	$ 50,857	$ 71,680	$ 75,433	$ 112,437
	A-DPF	$ 57,120	$ 78,725	$ 79,994	$ 117,568
	CCCF	$ 10,573	$ 36,756	$ 20,155	$ 70,064
Fuels	Emulsified	$ 167,584	$ 261,850	$ 122,045	$ 190,695
	Biodiesel	$ 288,844	$ 288,844	$ 210,354	$ 210,354
	Oxy-Diesel	$ 190,691	$ 190,691	$ 138,873	$ 138,873
	FBC	$ 33,941	$ 50,912	$ 24,718	$ 37,077

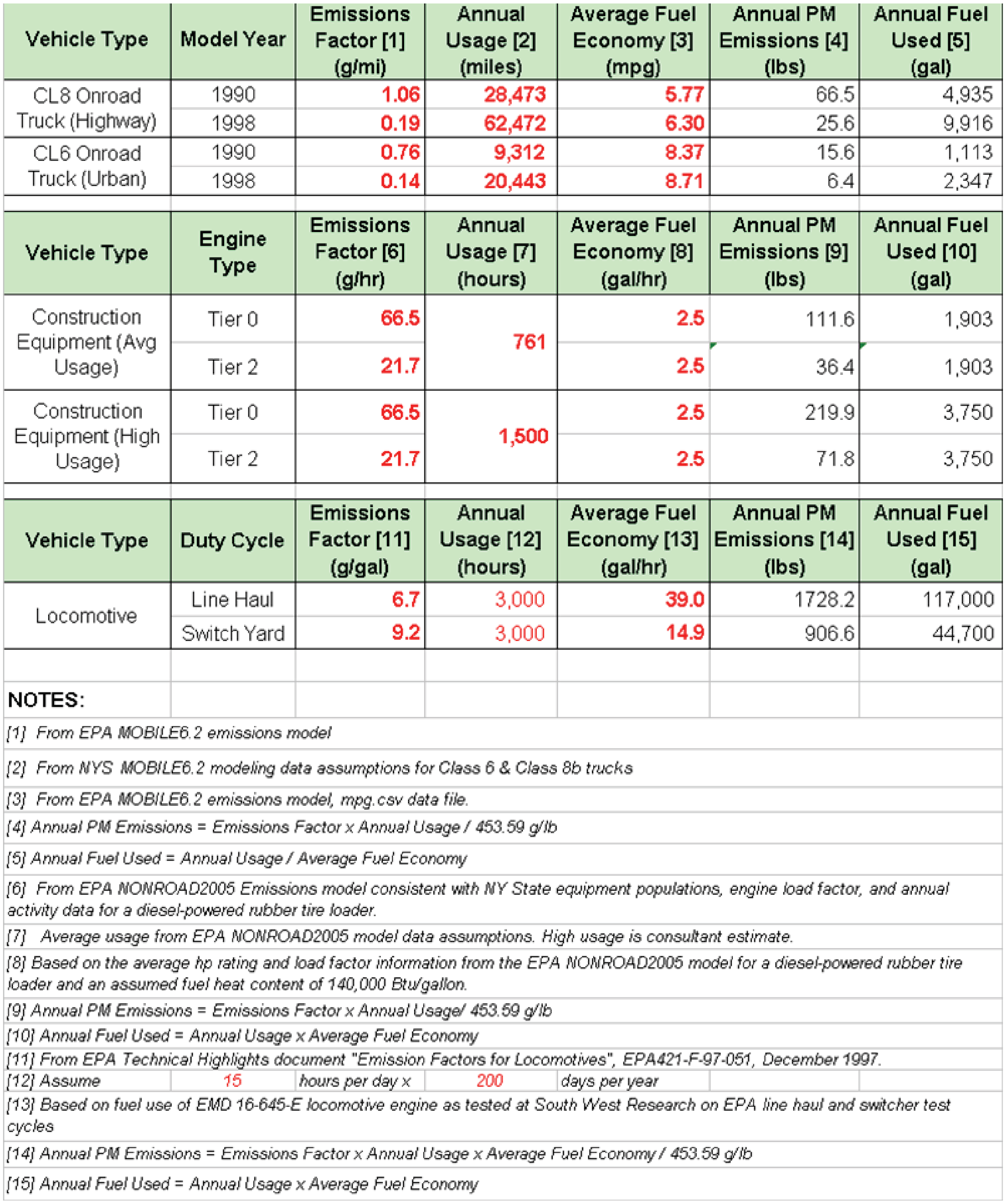
[1] Cost Effectiveness ($/ton) = NPV of Total Costs / (Annual PM Reductions (lb)/2000 (lb/ton) x 7)

Low = Low NPV / High Annual Reductions
High = High NPV / Low Annual Reductions

Annual reductions for each vehicle type and technology are shown in Table 1
NPV of total costs for each vehicle type and technology are shown in Tables 4 - 11

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this full useful life.

Table C-5. Vehicle Operations Baseline Calculations.



Vehicle Type	Model Year	Emissions Factor [1] (g/mi)	Annual Usage [2] (miles)	Average Fuel Economy [3] (mpg)	Annual PM Emissions [4] (lbs)	Annual Fuel Used [5] (gal)
CL8 Onroad Truck (Highway)	1990	1.06	28,473	5.77	66.5	4,935
	1998	0.19	62,472	6.30	25.6	9,916
CL6 Onroad Truck (Urban)	1990	0.76	9,312	8.37	15.6	1,113
	1998	0.14	20,443	8.71	6.4	2,347

Vehicle Type	Engine Type	Emissions Factor [6] (g/hr)	Annual Usage [7] (hours)	Average Fuel Economy [8] (gal/hr)	Annual PM Emissions [9] (lbs)	Annual Fuel Used [10] (gal)
Construction Equipment (Avg Usage)	Tier 0	66.5	761	2.5	111.6	1,903
	Tier 2	21.7		2.5	36.4	1,903
Construction Equipment (High Usage)	Tier 0	66.5	1,500	2.5	219.9	3,750
	Tier 2	21.7		2.5	71.8	3,750

Vehicle Type	Duty Cycle	Emissions Factor [11] (g/gal)	Annual Usage [12] (hours)	Average Fuel Economy [13] (gal/hr)	Annual PM Emissions [14] (lbs)	Annual Fuel Used [15] (gal)
Locomotive	Line Haul	6.7	3,000	39.0	1728.2	117,000
	Switch Yard	9.2	3,000	14.9	906.6	44,700

NOTES:

[1] From EPA MOBILE6.2 emissions model

[2] From NYS MOBILE6.2 modeling data assumptions for Class 6 & Class 8b trucks

[3] From EPA MOBILE6.2 emissions model, mpg.csv data file.

[4] Annual PM Emissions = Emissions Factor x Annual Usage / 453.59 g/lb

[5] Annual Fuel Used = Annual Usage / Average Fuel Economy

[6] From EPA NONROAD2005 Emissions model consistent with NY State equipment populations, engine load factor, and annual activity data for a diesel-powered rubber tire loader.

[7] Average usage from EPA NONROAD2005 model data assumptions. High usage is consultant estimate.

[8] Based on the average hp rating and load factor information from the EPA NONROAD2005 model for a diesel-powered rubber tire loader and an assumed fuel heat content of 140,000 Btu/gallon.

[9] Annual PM Emissions = Emissions Factor x Annual Usage/ 453.59 g/lb

[10] Annual Fuel Used = Annual Usage x Average Fuel Economy

[11] From EPA Technical Highlights document "Emission Factors for Locomotives", EPA421-F-97-051, December 1997.

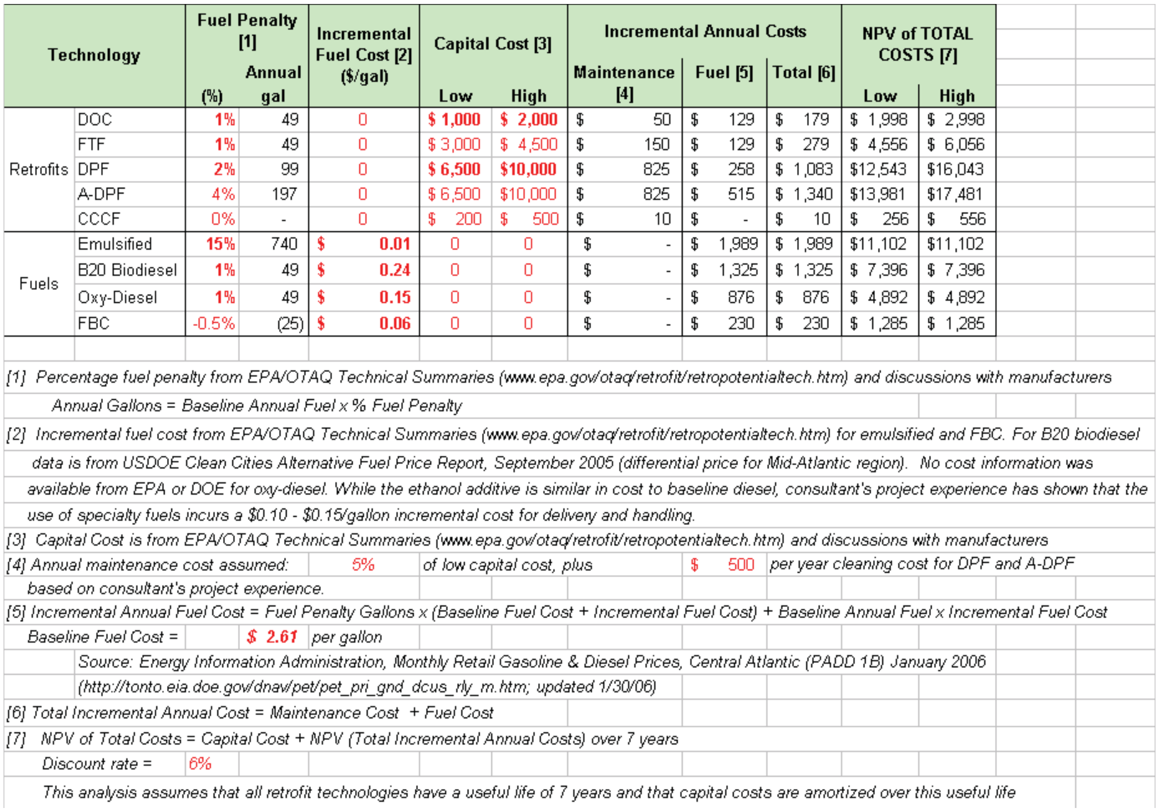
[12] Assume 15 hours per day x 200 days per year

[13] Based on fuel use of EMD 16-645-E locomotive engine as tested at South West Research on EPA line haul and switcher test cycles

[14] Annual PM Emissions = Emissions Factor x Annual Usage x Average Fuel Economy / 453.59 g/lb

[15] Annual Fuel Used = Annual Usage x Average Fuel Economy

Table C-6. Retrofit & Fuel Costs – 1990 Onroad Trucks Highway.



Technology		Fuel Penalty [1] (%)	Annual gal	Incremental Fuel Cost [2] ($/gal)	Capital Cost [3] Low	High	Incremental Annual Costs: Maintenance [4]	Fuel [5]	Total [6]	NPV of TOTAL COSTS [7] Low	High
Retrofits	DOC	1%	49	0	$ 1,000	$ 2,000	$ 50	$ 129	$ 179	$ 1,998	$ 2,998
	FTF	1%	49	0	$ 3,000	$ 4,500	$ 150	$ 129	$ 279	$ 4,556	$ 6,056
	DPF	2%	99	0	$ 6,500	$10,000	$ 825	$ 258	$ 1,083	$12,543	$16,043
	A-DPF	4%	197	0	$ 6,500	$10,000	$ 825	$ 515	$ 1,340	$13,981	$17,481
	CCCF	0%	-	0	$ 200	$ 500	$ 10	$ -	$ 10	$ 256	$ 556
Fuels	Emulsified	15%	740	$ 0.01	0	0	$ -	$ 1,989	$ 1,989	$11,102	$11,102
	B20 Biodiesel	1%	49	$ 0.24	0	0	$ -	$ 1,325	$ 1,325	$ 7,396	$ 7,396
	Oxy-Diesel	1%	49	$ 0.15	0	0	$ -	$ 876	$ 876	$ 4,892	$ 4,892
	FBC	-0.5%	(25)	$ 0.06	0	0	$ -	$ 230	$ 230	$ 1,285	$ 1,285

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

[3] Capital Cost is from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

[4] Annual maintenance cost assumed: 5% of low capital cost, plus $ 500 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = **$ 2.61** *per gallon*

Source: Energy Information Administration, Monthly Retail Gasoline & Diesel Prices, Central Atlantic (PADD 1B) January 2006
(http://tonto.eia.doe.gov/dnav/pet/pet_pri_gnd_dcus_rly_m.htm; updated 1/30/06)

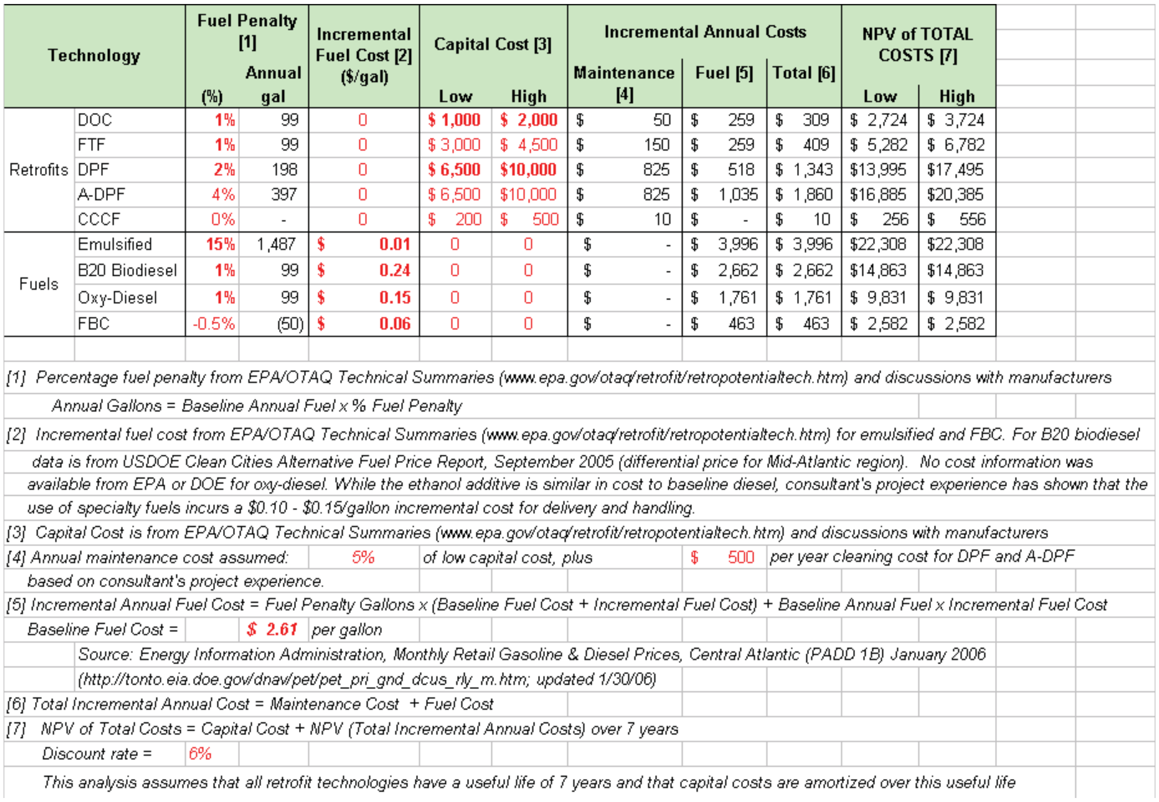
[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this useful life

Table C-7. Retrofit & Fuel Costs – 1998 Onroad Trucks Highway.



Technology		Fuel Penalty [1]		Incremental Fuel Cost [2] ($/gal)	Capital Cost [3]		Incremental Annual Costs			NPV of TOTAL COSTS [7]	
		(%)	Annual gal		Low	High	Maintenance [4]	Fuel [5]	Total [6]	Low	High
Retrofits	DOC	1%	99	0	$ 1,000	$ 2,000	$ 50	$ 259	$ 309	$ 2,724	$ 3,724
	FTF	1%	99	0	$ 3,000	$ 4,500	$ 150	$ 259	$ 409	$ 5,282	$ 6,782
	DPF	2%	198	0	$ 6,500	$10,000	$ 825	$ 518	$ 1,343	$13,995	$17,495
	A-DPF	4%	397	0	$ 6,500	$10,000	$ 825	$ 1,035	$ 1,860	$16,885	$20,385
	CCCF	0%	-	0	$ 200	$ 500	$ 10	$ -	$ 10	$ 256	$ 556
Fuels	Emulsified	15%	1,487	$ 0.01	0	0	$ -	$ 3,996	$ 3,996	$22,308	$22,308
	B20 Biodiesel	1%	99	$ 0.24	0	0	$ -	$ 2,662	$ 2,662	$14,863	$14,863
	Oxy-Diesel	1%	99	$ 0.15	0	0	$ -	$ 1,761	$ 1,761	$ 9,831	$ 9,831
	FBC	-0.5%	(50)	$ 0.06	0	0	$ -	$ 463	$ 463	$ 2,582	$ 2,582

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

[3] Capital Cost is from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

[4] Annual maintenance cost assumed: 5% of low capital cost, plus $ 500 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = $ 2.61 per gallon

Source: Energy Information Administration, Monthly Retail Gasoline & Diesel Prices, Central Atlantic (PADD 1B) January 2006 (http://tonto.eia.doe.gov/dnav/pet/pet_pri_gnd_dcus_rly_m.htm; updated 1/30/06)

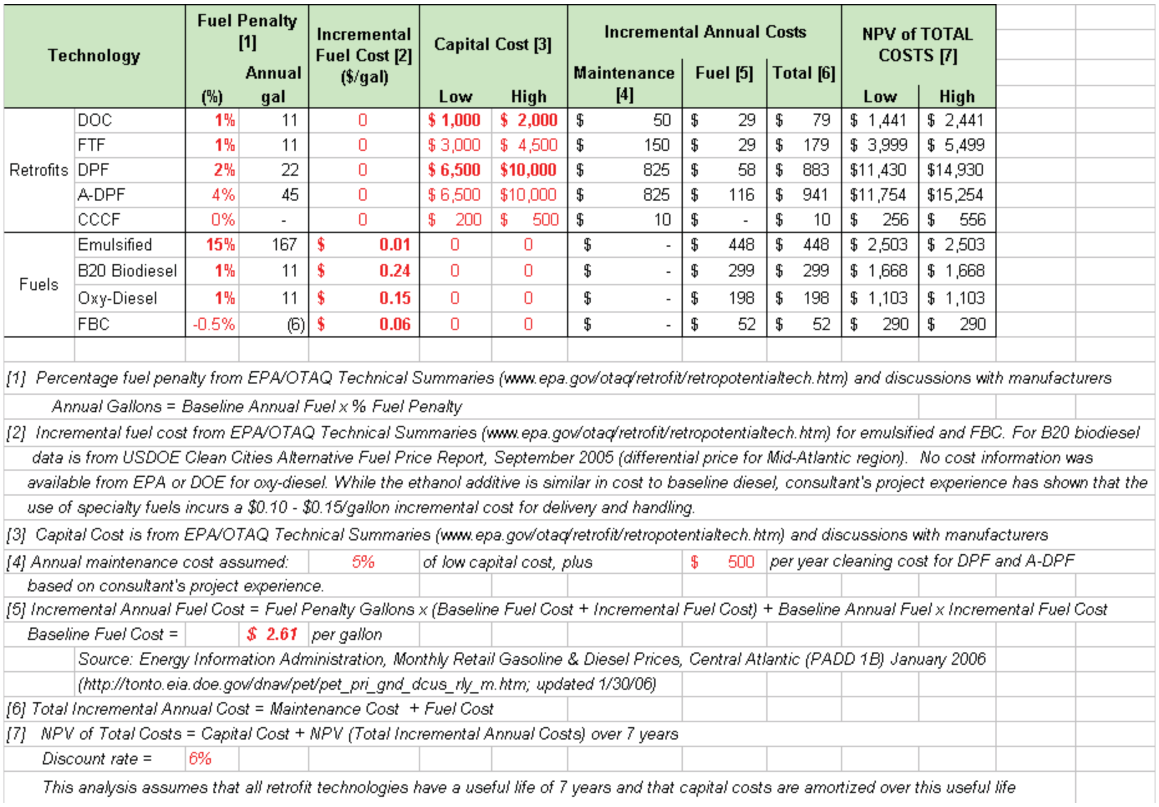
[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this useful life

Table C-8. Retrofit & Fuel Costs – 1990 Onroad Trucks Urban.



Technology		Fuel Penalty [1]		Incremental Fuel Cost [2] ($/gal)	Capital Cost [3]		Incremental Annual Costs			NPV of TOTAL COSTS [7]	
		(%)	Annual gal		Low	High	Maintenance [4]	Fuel [5]	Total [6]	Low	High
Retrofits	DOC	1%	11	0	$ 1,000	$ 2,000	$ 50	$ 29	$ 79	$ 1,441	$ 2,441
	FTF	1%	11	0	$ 3,000	$ 4,500	$ 150	$ 29	$ 179	$ 3,999	$ 5,499
	DPF	2%	22	0	$ 6,500	$10,000	$ 825	$ 58	$ 883	$11,430	$14,930
	A-DPF	4%	45	0	$ 6,500	$10,000	$ 825	$ 116	$ 941	$11,754	$15,254
	CCCF	0%	-	0	$ 200	$ 500	$ 10	$ -	$ 10	$ 256	$ 556
Fuels	Emulsified	15%	167	$ 0.01	0	0	$ -	$ 448	$ 448	$ 2,503	$ 2,503
	B20 Biodiesel	1%	11	$ 0.24	0	0	$ -	$ 299	$ 299	$ 1,668	$ 1,668
	Oxy-Diesel	1%	11	$ 0.15	0	0	$ -	$ 198	$ 198	$ 1,103	$ 1,103
	FBC	-0.5%	(6)	$ 0.06	0	0	$ -	$ 52	$ 52	$ 290	$ 290

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

[3] Capital Cost is from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

[4] Annual maintenance cost assumed: 5% of low capital cost, plus $ 500 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = $ 2.61 per gallon

Source: Energy Information Administration, Monthly Retail Gasoline & Diesel Prices, Central Atlantic (PADD 1B) January 2006 (http://tonto.eia.doe.gov/dnav/pet/pet_pri_gnd_dcus_rly_m.htm; updated 1/30/06)

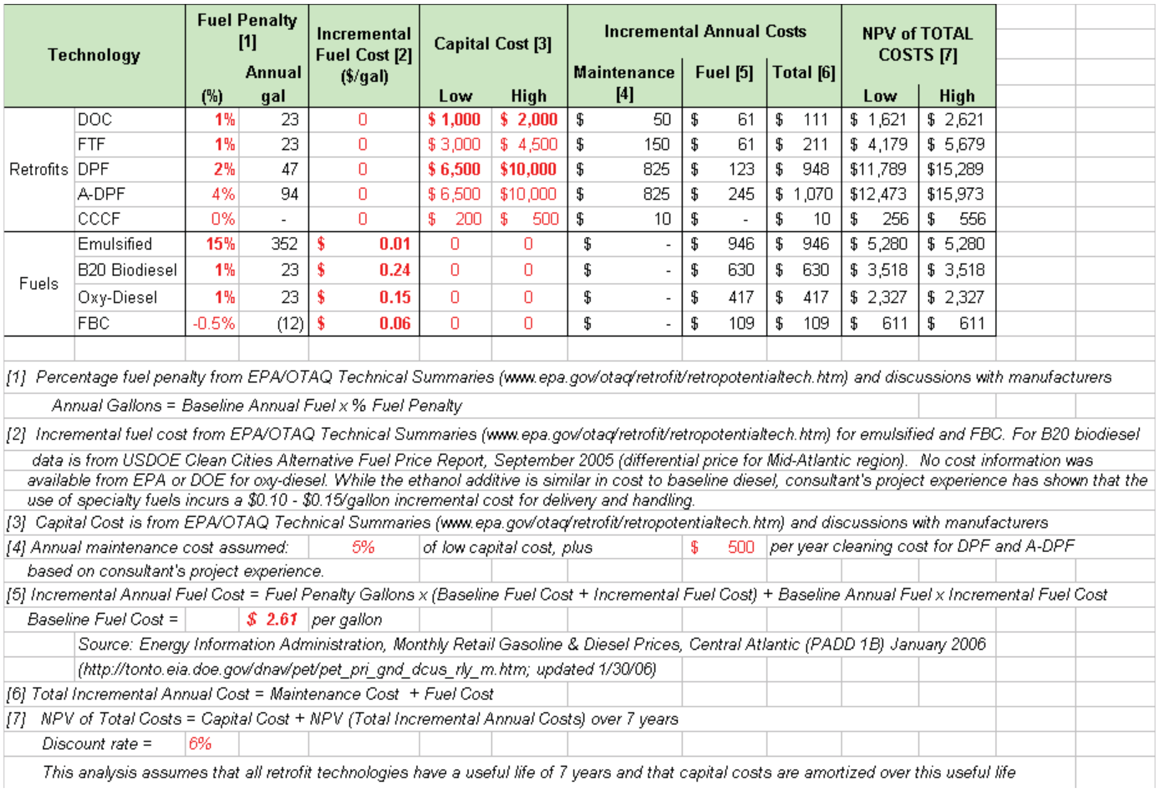
[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this useful life

Table C-9. Retrofit & Fuel Costs – 1998 Onroad Trucks Urban.



Technology		Fuel Penalty [1] (%)	Annual gal	Incremental Fuel Cost [2] ($/gal)	Capital Cost [3] Low	High	Incremental Annual Costs: Maintenance [4]	Fuel [5]	Total [6]	NPV of TOTAL COSTS [7] Low	High
Retrofits	DOC	1%	23	0	$ 1,000	$ 2,000	$ 50	$ 61	$ 111	$ 1,621	$ 2,621
	FTF	1%	23	0	$ 3,000	$ 4,500	$ 150	$ 61	$ 211	$ 4,179	$ 5,679
	DPF	2%	47	0	$ 6,500	$10,000	$ 825	$ 123	$ 948	$11,789	$15,289
	A-DPF	4%	94	0	$ 6,500	$10,000	$ 825	$ 245	$ 1,070	$12,473	$15,973
	CCCF	0%	-	0	$ 200	$ 500	$ 10	$ -	$ 10	$ 256	$ 556
Fuels	Emulsified	15%	352	$ 0.01	0	0	$ -	$ 946	$ 946	$ 5,280	$ 5,280
	B20 Biodiesel	1%	23	$ 0.24	0	0	$ -	$ 630	$ 630	$ 3,518	$ 3,518
	Oxy-Diesel	1%	23	$ 0.15	0	0	$ -	$ 417	$ 417	$ 2,327	$ 2,327
	FBC	-0.5%	(12)	$ 0.06	0	0	$ -	$ 109	$ 109	$ 611	$ 611

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

[3] Capital Cost is from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

[4] Annual maintenance cost assumed: 5% of low capital cost, plus $ 500 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = $ 2.61 per gallon

Source: Energy Information Administration, Monthly Retail Gasoline & Diesel Prices, Central Atlantic (PADD 1B) January 2006

(http://tonto.eia.doe.gov/dnav/pet/pet_pri_gnd_dcus_rly_m.htm; updated 1/30/06)

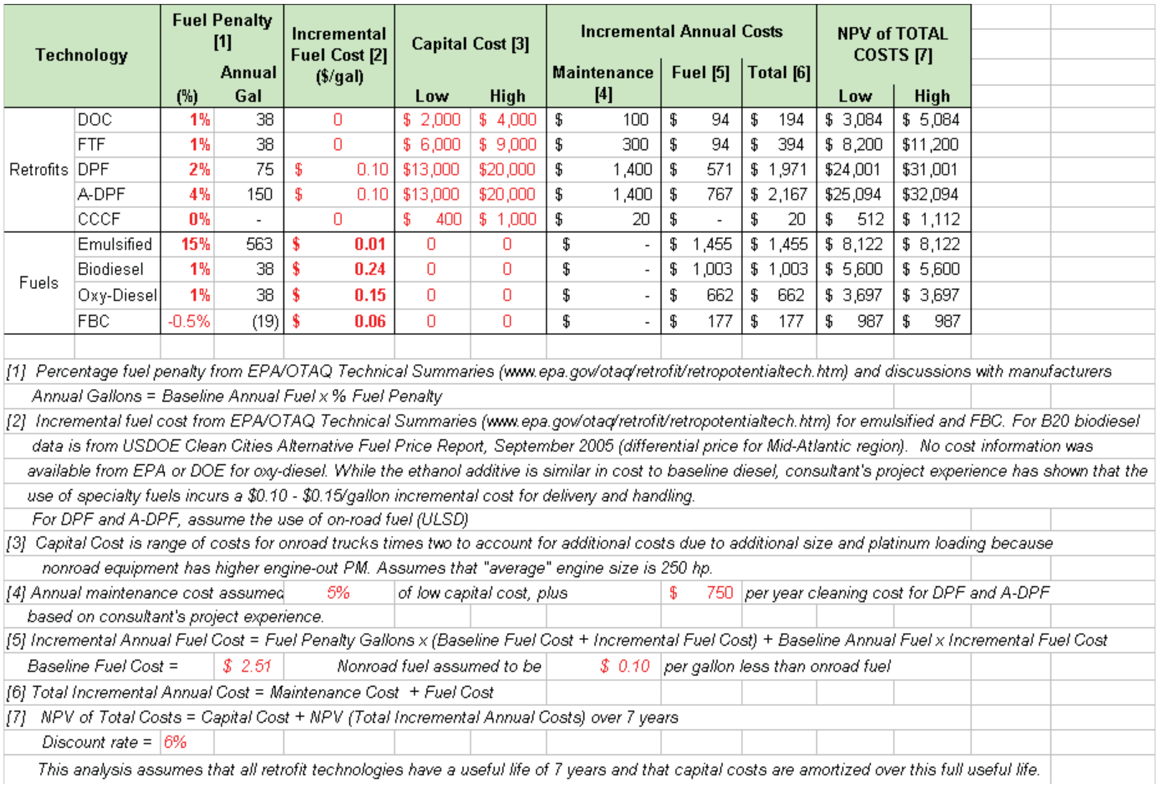
[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this useful life

Table C-10. Retrofit & Fuel Costs – Construction Equipment, High Usage.



Technology		Fuel Penalty [1]		Incremental Fuel Cost [2] ($/gal)	Capital Cost [3]		Incremental Annual Costs			NPV of TOTAL COSTS [7]	
		(%)	Annual Gal		Low	High	Maintenance [4]	Fuel [5]	Total [6]	Low	High
Retrofits	DOC	1%	38	0	$ 2,000	$ 4,000	$ 100	$ 94	$ 194	$ 3,084	$ 5,084
	FTF	1%	38	0	$ 6,000	$ 9,000	$ 300	$ 94	$ 394	$ 8,200	$11,200
	DPF	2%	75	$ 0.10	$13,000	$20,000	$ 1,400	$ 571	$ 1,971	$24,001	$31,001
	A-DPF	4%	150	$ 0.10	$13,000	$20,000	$ 1,400	$ 767	$ 2,167	$25,094	$32,094
	CCCF	0%	-	0	$ 400	$ 1,000	$ 20	$ -	$ 20	$ 512	$ 1,112
Fuels	Emulsified	15%	563	$ 0.01	0	0	$ -	$ 1,455	$ 1,455	$ 8,122	$ 8,122
	Biodiesel	1%	38	$ 0.24	0	0	$ -	$ 1,003	$ 1,003	$ 5,600	$ 5,600
	Oxy-Diesel	1%	38	$ 0.15	0	0	$ -	$ 662	$ 662	$ 3,697	$ 3,697
	FBC	-0.5%	(19)	$ 0.06	0	0	$ -	$ 177	$ 177	$ 987	$ 987

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers
Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.
For DPF and A-DPF, assume the use of on-road fuel (ULSD)

[3] Capital Cost is range of costs for onroad trucks times two to account for additional costs due to additional size and platinum loading because nonroad equipment has higher engine-out PM. Assumes that "average" engine size is 250 hp.

[4] Annual maintenance cost assumed 5% of low capital cost, plus $ 750 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost
Baseline Fuel Cost = $ 2.51 Nonroad fuel assumed to be $ 0.10 per gallon less than onroad fuel

[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years
Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this full useful life.

Table C-11. Retrofit & Fuel Costs – Construction Equipment, Average Usage.

Technology		Fuel Penalty [1]		Incremental Fuel Cost [2] ($/gal)	Capital Cost [3]		Incremental Annual Costs			NPV of TOTAL COSTS [7]	
		(%)	Annual Gal		Low	High	Maintenance [4]	Fuel [5]	Total [6]	Low	High
Retrofits	DOC	1%	19	0	$ 2,000	$ 4,000	$ 100	$ 48	$ 148	$ 2,825	$ 4,825
	FTF	1%	19	0	$ 6,000	$ 9,000	$ 300	$ 48	$ 348	$ 7,941	$10,941
	DPF	2%	38	$ 0.10	$13,000	$20,000	$ 1,400	$ 290	$ 1,690	$22,432	$29,432
	A-DPF	4%	76	$ 0.10	$13,000	$20,000	$ 1,400	$ 389	$ 1,789	$22,986	$29,986
	CCCF	0%	-	0	$ 400	$ 1,000	$ 20	$ -	$ 20	$ 512	$ 1,112
Fuels	Emulsified	15%	285	$ 0.01	0	0	$ -	$ 738	$ 738	$ 4,121	$ 4,121
	Biodiesel	1%	19	$ 0.24	0	0	$ -	$ 509	$ 509	$ 2,841	$ 2,841
	Oxy-Diesel	1%	19	$ 0.15	0	0	$ -	$ 336	$ 336	$ 1,876	$ 1,876
	FBC	-0.5%	(10)	$ 0.06	0	0	$ -	$ 90	$ 90	$ 501	$ 501

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

For DPF and A-DPF, assume the use of on-road fuel (ULSD)

[3] Capital Cost is range of costs for onroad trucks times two to account for additional costs due to additional size and platinum loading because nonroad equipment has higher engine-out PM. Assumes that "average" engine size is 250 hp.

[4] Annual maintenance cost assumed 5% of low capital cost, plus $ 750 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = $ 2.51 Nonroad fuel assumed to be $ 0.10 per gallon less than onroad fuel

[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this full useful life.

Table C-12. Retrofit & Fuel Costs – Linehaul Locomotive.

Technology		Fuel Penalty [1] (%)	Annual gal	Incremental Fuel Cost [2] ($/gal)	Capital Cost [3] Low	High	Incremental Annual Costs Maintenance [4]	Fuel [5]	Total [6]	NPV of TOTAL COSTS [7] Low	High
Retrofits	DOC	1%	1,170	0	$ 20,000	$ 40,000	$ 1,000	$ 2,937	$ 3,937	$ 41,976	$ 61,976
	FTF	1%	1,170	0	$ 60,000	$ 90,000	$ 3,000	$ 2,937	$ 5,937	$ 93,141	$ 123,141
	DPF	2%	2,340	$ 0.10	$130,000	$ 200,000	$ 8,500	$ 17,807	$26,307	$ 276,858	$ 346,858
	A-DPF	4%	4,680	$ 0.10	$130,000	$ 200,000	$ 8,500	$ 23,915	$32,415	$ 310,952	$ 380,952
	CCCF	0%	-	0	$ 4,000	$ 10,000	$ 200	$ -	$ 200	$ 5,116	$ 11,116
Fuels	Emulsified	15%	17,550	$ 0.01	0	0	$ -	$ 45,396	$45,396	$ 253,418	$ 253,418
	Biodiesel	1%	1,170	$ 0.24	0	0	$ -	$ 31,298	$31,298	$ 174,715	$ 174,715
	Oxy-Diesel	1%	1,170	$ 0.15	0	0	$ -	$ 20,662	$20,662	$ 115,344	$ 115,344
	FBC	-0.5%	(585)	$ 0.06	0	0	$ -	$ 5,517	$ 5,517	$ 30,795	$ 30,795

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

[3] Capital Cost is range of costs for nonroad construction equipment times ten to account for larger engine size (3,000 hp vs 250 hp)

[4] Annual maintenance cost assumed: 5% of low capital cost, plus $ 2,000 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = $ 2.51 Nonroad fuel assumed to be $ 0.10 per gallon less than onroad fuel.

[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this full useful life.

Table C-13. Retrofit & Fuel Costs – Switchyard Locomotive.

Technology		Fuel Penalty [1] (%)	Annual gal	Incremental Fuel Cost [2] ($/gal)	Capital Cost [3] Low	High	Incremental Annual Costs Maintenance [4]	Fuel [5]	Total [6]	NPV of TOTAL COSTS [7] Low	High
Retrofits	DOC	1%	447	0	$ 20,000	$ 40,000	$ 1,000	$ 1,122	$ 2,122	$ 31,846	$ 51,846
	FTF	1%	447	0	$ 60,000	$ 90,000	$ 3,000	$ 1,122	$ 4,122	$ 83,010	$ 113,010
	DPF	2%	894	$ 0.10	$130,000	$ 200,000	$ 8,500	$ 6,803	$15,303	$ 215,429	$ 285,429
	A-DPF	4%	1,788	$ 0.10	$130,000	$ 200,000	$ 8,500	$ 9,137	$17,637	$ 228,455	$ 298,455
	CCCF	0%	-	0	$ 4,000	$ 10,000	$ 200	$ -	$ 200	$ 5,116	$ 11,116
Fuels	Emulsified	15%	6,705	$ 0.01	0	0	$ -	$ 17,344	$17,344	$ 96,819	$ 96,819
	Biodiesel	1%	447	$ 0.24	0	0	$ -	$ 11,957	$11,957	$ 66,750	$ 66,750
	Oxy-Diesel	1%	447	$ 0.15	0	0	$ -	$ 7,894	$ 7,894	$ 44,067	$ 44,067
	FBC	-0.5%	(224)	$ 0.06	0	0	$ -	$ 2,108	$ 2,108	$ 11,765	$ 11,765

[1] Percentage fuel penalty from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) and discussions with manufacturers

Annual Gallons = Baseline Annual Fuel x % Fuel Penalty

[2] Incremental fuel cost from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm) for emulsified and FBC. For B20 biodiesel data is from USDOE Clean Cities Alternative Fuel Price Report, September 2005 (differential price for Mid-Atlantic region). No cost information was available from EPA or DOE for oxy-diesel. While the ethanol additive is similar in cost to baseline diesel, consultant's project experience has shown that the use of specialty fuels incurs a $0.10 - $0.15/gallon incremental cost for delivery and handling.

[3] Capital Cost is range of costs for nonroad construction equipment times ten to account for larger engine size (3,000 hp vs 250 hp)

[4] Annual maintenance cost assumed: 5% of low capital cost, plus $ 2,000 per year cleaning cost for DPF and A-DPF based on consultant's project experience.

[5] Incremental Annual Fuel Cost = Fuel Penalty Gallons x (Baseline Fuel Cost + Incremental Fuel Cost) + Baseline Annual Fuel x Incremental Fuel Cost

Baseline Fuel Cost = $ 2.51 Nonroad fuel assumed to be $ 0.10 per gallon less than onroad fuel.

[6] Total Incremental Annual Cost = Maintenance Cost + Fuel Cost

[7] NPV of Total Costs = Capital Cost + NPV (Total Incremental Annual Costs) over 7 years

Discount rate = 6%

This analysis assumes that all retrofit technologies have a useful life of 7 years and that capital costs are amortized over this full useful life.

Table C-14. Potential Emissions Reductions & Fuel Savings – Idle Reduction Technologies.

SWITCHER LOCOMOTIVES

Technology	Percentage of Idling Reduced [1]	Capital Cost [2]		Annual PM Reductions [3] (lbs)	Annual Fuel Savings [4] (gal)	Annual Cost Savings [5] ($)	Pay-back Period [6] (yrs)	
		Low	High				Short	Long
Engine Shut-down/Restart	30%	$ 5,000	$ 10,000	33.1	2,916	$ 7,319	0.68	1.37
Auxiliary Power Units	80%	$ 30,000	$ 40,000	88.3	7,776	$ 19,518	1.54	2.05
Supplementary Diesel Heat	30%	$ 15,000	$ 20,000	33.1	2,916	$ 7,319	2.05	2.73
Hybrid Locomotive	100%	$ 200,000	$ 200,000	110.3	9,720	$ 24,397	8.20	8.20

SLEEPER CAB TRUCKS

Technology	Percentage of Idling Reduced [1]	Capital Cost [2]		Annual PM Reductions [3] (lbs)	Annual Fuel Savings [4] (gal)	Annual Cost Savings [5] ($)	Pay-back Period [6] (yrs)	
		Low	High				Short	Long
Engine Shut-down/Restart	20%	$ 1,000	$ 1,500	3.3	320	$ 835	1.20	1.80
Electric Driven Auxiliaries	70%	$ 500	$ 4,000	11.4	1120	$ 2,923	0.17	1.37
Auxiliary Power Units	90%	$ 6,000	$ 9,000	14.7	1440	$ 3,758	1.60	2.39
Supplementary Diesel Heat	30%	$ 900	$ 3,000	4.9	480	$ 1,253	0.72	2.39
Grid Power Systems	50%	$ 100	$ 2,000	8.2	800	$ 2,088	0.05	0.96

[1] Consultant assumption based on industry discussion sand observation

[2] Capital Cost is from EPA/OTAQ Technical Summaries (www.epa.gov/otaq/retrofit/retropotentialtech.htm)

[3] Annual PM reductions = % idle reduced x Annual PM Emissions from Idling (from Table 13)

[4] Annual Fuel Savings = % idle reduced x Annual Fuel Used Idling (from Table 13)

[5] Annual Cost Savings = Annual Fuel Savings x Diesel Fuel Cost

Baseline onroad diesel fuel cost = $ 2.61 per gallon.

Source: Energy Information Administration, Monthly Retail Gasoline & Diesel Prices, Central Atlantic (PADD 1B) January 2006
(http://tonto.eia.doe.gov/dnav/pet/pet_pri_gnd_dcus_rly_m.htm; updated 1/30/06)

Baseline nonroad diesel fuel cost = $ 2.51 per gallon.

Assumption based on historical pricing

[6] Payback period = Capital Cost / Annual Cost Savings

Table C-15. Vehicle Idling Baseline Calculations.

Vehicle Type	Idling Factors [1]		Idling Usage [2]		Annual Totals from Idling [3]	
	PM Emissions (g/hr)	Fuel Used (gal/hr)	Days/yr	Hrs/day	PM Emissions (lbs)	Fuel Used (gal)
Urban Truck/Bus	3.7	0.8	200	3	4.9	480
Sleeper Cab Truck	3.7	0.8	200	10	16.3	1,600
Switch Yard Locomotive (warm weather)	26	3	140	9	110.3	9,720
Switch Yard Locomotive (cold weather)	32	11	60	9		

[1] Taken from "Guidance for Quantifying and Using Long Duration Switch Yard Locomotive Idling Emission Reductions in State Implementation Plans", US Environmental Protection Agency, EPA420-B-04-002, January 2004 and "Guidance for Quantifying and Using Long Duration Truck Idling Emission Reductions in State Implementation Plans and Transportation Conformity", US Environmental Protection Agency, EPA420-B-04-001, January 2004

[2] Consultant assumptions based on industry discussions and observation. Switchyard locomotive assumed to operate for 15 hours per day, with locomotive idling for 60% of duty cycle, consistent with EPA switcher test cycle.

[3] Annual PM = PM emission Factor x Idling Usage/453.59 g/lb

Annual Fuel = Fuel Usage Factor x Idling Usage

Appendix D

NEW VEHICLE & ENGINE EMISSION STANDARDS

INTRODUCTION

The Clean Air Act gives the Environmental Protection Agency (EPA) primary authority to regulate emissions from new onroad and nonroad vehicles and engines, including light-duty onroad vehicles (cars and light trucks), heavy-duty onroad vehicles (trucks and buses), and the engines used in nonroad equipment (construction, industrial, and agricultural equipment; locomotives; ships; and aircraft). California has authority to adopt its own emissions standards for new onroad vehicles and engines and some, but not all, new nonroad engines as long as they are at least as stringent as federal standards. Other states can adopt either the California or the federal standards. As we discuss below, California standards for new cars and light trucks are more stringent than federal standards, but that state's regulations for heavy-duty onroad engines and nonroad engines largely track federal standards. California has also begun to adopt mandatory retrofit requirements for some categories of heavy-duty onroad and nonroad engines.

The picture is somewhat different for existing vehicles. For existing onroad vehicles, states are free to set their own standards. However, for existing nonroad equipment, states can adopt either the California or the federal standards, but they cannot set their own standards.

State and local regulation of certain categories of nonroad equipment is fully preempted under §209(e) of the Clean Air Act, meaning that no state—including California—may set standards for this equipment. Section 209 preemption applies to new engines used in construction equipment and new engines used in farm equipment, provided the engines are smaller than 175 horsepower; preemption also applies to new locomotives and new engines used in locomotives. In addition, as discussed below, EPA regulates the emissions of nitrogen oxides (NO_X) (but not of other pollutants) from large ocean-going vessels with an eye to international treaty negotiations.

Congress has also given EPA authority to regulate the quality of fuel and fuel additives used in vehicles, based on direct health effects from the fuel itself and the effect of fuel on the ability of vehicles to meet emissions standards. California may adopt its own fuel regulations regardless of EPA standards, but other states are not always free to follow them. States may regulate a fuel characteristic or component only if EPA has not already done so or signaled an intent to do so in the future. Under the Clean Air Act, if EPA has regulated a fuel characteristic or component, states other than California must follow the federal regulation—and may not adopt their own or California's rules. The only exception to this is the situation in which the Administrator finds that state regulation is necessary for achieving the National Ambient Air Quality Standards (NAAQS).

CARS AND LIGHT TRUCKS

EPA began regulating emissions from new light-duty vehicles in the 1970s and continued tightening emissions standards over the years that followed. Most vehicles on the road today were manufactured subject to the Tier 1 standards that were adopted in 1991 and phased in between the 1994 and 1997 model years. These standards allowed higher NO_X emissions from light-duty trucks than from cars and also allowed higher NO_X and PM emissions from diesel than from gasoline vehicles (EPA 2000).

In 2000, EPA adopted Tier 2 standards for light-duty vehicles, which will be phased in between the 2004 and 2009 model years. The Tier 2 standards apply the same absolute emissions limit (grams per mile) to virtually all passenger vehicles regardless of size or weight, including cars, pick-up trucks, vans, and sport utility vehicles (SUVs). They also apply the same limit regardless of the fuel used, so that new light-duty diesel vehicles will have to be as clean as vehicles with gasoline engines (EPA 1999, 2004b).

The Tier 2 standards allow vehicle manufacturers to certify to any one of eight “bins” with different emissions levels: Bin 1 is the cleanest, with the most stringent standards, while Bin 8 has more lenient standards. The average NO_X emissions from all vehicles that a manufacturer sells must be no more than 0.07 grams/mile, equivalent to Tier 2, Bin 5. This is almost 90% lower than the Tier 1 NO_X standard for cars. The Tier 2 PM standard is also 90% lower than the corresponding Tier 1 standard (EPA 2000, 2004b).

EPA does not regulate carbon dioxide (CO_2) or other greenhouse gas emissions from vehicles. The position taken currently by EPA is that the Clean Air Act does not authorize it to regulate greenhouse gas emissions from any source. This issue is currently being litigated.

The Department of Transportation (DOT) imposes Corporate Average Fuel Economy (CAFE) requirements on auto manufacturers. Current CAFE standards require manufacturers to maintain a car-fleet-average fuel economy of 27.5 miles per gallon and an average fuel economy of 21.0 miles per gallon for light trucks (STAPPA 2005). Higher fuel economy standards decrease the amount of pollution emitted per mile because a vehicle is using less fuel per mile. DOT recently proposed revised CAFE standards that will begin to close the “SUV loophole” in the current regulations. The proposed standards divide light trucks into six categories based on vehicle footprint. The smallest truck category would be required to achieve an average fuel economy of 28.4 miles per gallon in 2011, while the largest would be required to achieve only 21.2 miles per gallon. The proposed revisions do not increase CAFE requirements for cars, nor do they extend fuel economy requirements to vehicles over 8,500 lb—a category that includes such vehicles as the Hummer H2, Ford Excursion, and Chevy Suburban (STAPPA 2005).

Other Regulations: California Light-Duty Vehicle Standards

Under the Clean Air Act, California can set its own emissions standards for new light-duty vehicles. The Tier 1/Low Emissions Vehicle California standards in effect through 2003 were very similar to Tier 1 EPA standards, although they included additional, more stringent "low emission vehicle" regulatory categories (i.e., Transitional Low Emission Vehicle (TLEV), Low Emission Vehicle (LEV), Ultra-Low Emission Vehicle (ULEV), and Super Ultra-Low Emission Vehicle (SULEV)) for vehicles 6,000–14,000 lb gross vehicle weight). The Tier 1/LEV program established an increasingly stringent fleet-average requirement for emissions of nonmethane organic gases (NMOG), which required auto manufacturers to certify an increasing share of their total vehicle sales in the lower emissions categories. The Tier 1/LEV program also required auto manufacturers to begin selling limited quantities of Zero Emission Vehicles (ZEVs), which at the time were generally battery-electric vehicles. In addition, unlike the EPA Tier 1 standards, the California Tier 1/LEV standards applied the same numerical emissions limits to both gasoline and diesel cars and light trucks (CARB 1999).

Beginning with the 2004 model year, California began to phase in LEVII standards. LEVII eliminates the Tier 1 and TLEV regulatory categories, significantly reduces allowable PM and NO_X emissions in the LEV and ULEV categories, and establishes a SULEV category for vehicles less than 6,000 lb gross vehicle weight. LEVII also establishes Partial Zero-Emission Vehicle (PZEV) and Advanced Technology Partial Zero-Emission Vehicle (AT-PZEV) categories. To certify as a PZEV or AT-PZEV, a vehicle must meet SULEV emissions standards, have zero fuel evaporative emissions, and carry a 15-year/150,000-mile full emissions warranty. PZEV and AT-PZEV vehicles can be used by auto manufacturers to meet California's ZEV requirements (CARB 1999).

California's LEVII standards apply to all cars and light trucks weighing less than 8,500 lb—meaning that most pick-ups and SUVs in California will be required to meet the same standards as cars. Under LEVII, the LEV and ULEV NO_X and PM limits are the same as EPA Tier 2, Bin 5; and the SULEV NO_X and PM limits are the same as EPA Tier 2, Bin 2 (DieselNet 2005).

The LEVII standards extend the requirements for automakers to certify increasing percentages of vehicles in the ULEV and SULEV categories. As a result, fleet-average emissions levels for new vehicles sold in California will continue to decline through 2010 (CARB 1999).

In September 2004 the California Air Resources Board (CARB) adopted new rules to regulate greenhouse gas emissions from cars and light trucks, including CO_2, methane, nitrous oxide, and hydrofluorocarbons. (Unlike the other substances, hydrofluorocarbons do not typically occur in engine exhaust but are used in vehicle air conditioning systems.) These new regulations are currently scheduled to phase in between the 2009 and 2016 model years. CARB estimates that the new regulations will reduce greenhouse gas emissions from California’s light-duty vehicle fleet by 18% in 2020 compared to today, and by 27% in 2030 (CARB 2004). New York, like a number of other states, has also promulgated these regulations.

Automakers have filed a legal challenge to the CARB greenhouse gas regulations, contending that they are, in effect, fuel economy standards and that only Congress has authority to set such standards.

TRUCKS AND BUSES

EPA considers any vehicle over 8,500 lb gross vehicle weight to be heavy-duty and, therefore, subject to different regulations than light-duty cars and trucks. The exception to this is that under EPA Tier 2 regulations for light-duty vehicles, certain very large SUVs and passenger vans used for personal transportation (8,500–10,000 lb gross vehicle weight) are re-classified as medium-duty passenger vehicles and are subject to the light-duty vehicle rules (DieselNet 2005).

Unlike light-duty vehicles that are tested and certified at the vehicle level using a chassis test, only the engines themselves are certified for heavy-duty vehicles, using an engine test. Consequently, the numerical emissions limits are not expressed as grams per mile, but rather as grams per brake-horsepower hour (g/bhp-hr), equivalent to grams of emissions per unit of work done by the engine.

EPA New Onroad Engine Emissions Standards

EPA first set exhaust smoke opacity standards for new heavy-duty onroad diesel engines beginning in model year 1970. Starting in model year 1974, new engines were required to meet numeric emissions limits for carbon monoxide (CO) and NO_X + hydrocarbons (HC), but PM was not regulated until 1988. Between 1988 and 1998, the EPA limits for HC and CO remained the same, but allowable levels of both NO_X and PM were reduced in several steps. At 4.0 g/bhp-hr NO_X and 0.10 g/bhp-hr PM, the emissions limits for the 1998 model year were 63% and 83% lower, respectively, than those for the 1988 model year.

In 1997, EPA adopted an even lower standard for heavy-duty diesel engine NO_X emissions to take effect in the 2004 model year. The next year EPA signed a consent decree with the six major heavy-duty engine manufacturers to settle a lawsuit brought by EPA. According to EPA, manufacturers had for a number of years been using an “emissions defeat device” that modified engine control software to improve fuel economy. While improving fuel economy, the device also increased NO_X emissions during certain high-speed steady-state (highway) driving modes that were not fully captured during the certification test cycle. Among other remedies, the consent decree mandated the 2004 NO_X standard of 2.5 g/bhp-hr for engines built after October 2002. Under the consent decree, engine manufacturers were also required to develop modified software for model year 1993–1998 engines that would reduce off-cycle highway NO_X emissions (this software is often referred to as an ECM or chip “reflash”) and to make this software available to vehicle owners free of charge. The consent decree required that all engines be upgraded with this new software at the time of normal engine overhaul or rebuild, which was assumed to occur after 200,000 to 300,000 miles in service (*U.S. v Caterpillar* 1999).

In December 2000, EPA adopted a NO_X standard of 0.20 g/bhp-hr and a PM standard of 0.01 g/bhp-hr for new onroad heavy-duty diesel engines. This PM standard will go into full effect in the 2007 model year, while the NO_X standard will be phased in between 2007 and 2010 on a percent-of-sales basis.

The 2007 emissions standards introduce additional steady-state tests and not-to-exceed limits in the certification process, to ensure that defeat devices like those that led to the 1998 consent decree will no longer be possible. The 2007 regulations also require that crankcase vent emissions from all diesel engines be controlled (DieselNet 2005). Previously, engines with turbo-chargers were allowed to vent their crankcase emissions to the atmosphere; these emissions were thus not included in the exhaust limits.

Urban Bus Retrofit/Rebuild Rule

In 1993, EPA finalized a retrofit/rebuild rule that applies to urban transit bus engines built before 1994. The rule requires that a certified retrofit/rebuild kit be used every time one of these engines is rebuilt or overhauled. These kits will either reduce PM emissions by 25% compared to the original certification level of the engine, or will achieve PM emissions of 0.1 g/bhp-hr (EPA 1998). To date, EPA has not proposed retrofit/rebuild requirements for any other onroad diesel engines.

Other Regulations: California Heavy-Duty Onroad Standards

California's regulation of onroad heavy-duty diesel engines began in 1973, but PM emissions were not regulated until 1987, one year before EPA began regulating PM emissions from the same types of engines. In addition, California's 1987 NO_X standard of 6.0 g/bhp-hr was not matched by federal standards until 1990. However, with the exception of engines for urban buses, the numerical emissions limits set by California for heavy-duty onroad engines have been virtually identical to EPA limits since then. California adopted more stringent standards for urban buses between model years 2004 and 2006; these standards included fleet averaging and mandatory fleet-wide emissions reduction requirements. For the 2005 model year, California also adopted supplemental steady-state tests and not-to-exceed limits that will not be mandated by EPA until model year 2007 (DieselNet 2005).

After 2007, California's numerical emissions limits for onroad heavy-duty diesel engines are virtually identical to EPA standards. However, in July 2005 California adopted rules that will require the addition of onboard diagnostic systems (OBD) for onroad diesel engines beginning in the 2010 model year (CARB, 2005).

Similar to the OBD systems required for light-duty vehicles under current EPA and California rules, these diesel OBD systems will be required to monitor engine systems that affect emissions and to warn the driver of failures that are likely to increase emissions, so that they can be repaired. Like inspection and maintenance (I&M) programs, OBD systems are designed to ensure that vehicles continue to meet new vehicle emission standards throughout their lives.

NONROAD EQUIPMENT

Generally speaking, EPA has three separate regulatory schemes for nonroad diesel engines: one for locomotive engines, one for marine engines, and one for all other engines (agricultural, industrial, construction, etc.). Each of these categories of engines will be discussed separately.

For regulatory purposes, nonroad engines are those used in mobile equipment not intended for use on public roadways. Therefore, the definition of "nonroad" is based primarily on mobility or portability. Despite the fact that it may be virtually identical to an engine used in a piece of mobile equipment, any diesel engine that will stay in the same place for 12 months or more—for example, to power a stationary generator or pump—is regulated as a stationary emissions source and is not discussed here (DieselNet 2005).

EPA New Engine Emission Standards: Construction, Agricultural, and Industrial Engines

The first EPA emissions standards for new nonroad engines other than marine engines and locomotives were adopted in 1994. These Tier 1 standards were phased in for engines of different sizes starting as early as model year 1996. In 1998, EPA finalized a rule that requires more stringent Tier 2 and Tier 3 standards to be implemented for engines built between 2001 and 2008, and in 2004 the Agency finalized Tier 4 standards that will take effect between 2008 and 2015 (EPA 2004c, 2004d).

As with emissions standards for onroad truck and bus engines, emissions limits for nonroad engines are expressed in terms of mass per unit of work done by the engine—that is, g/bhp-hr, or grams per kilowatt hour (g/kWh). Unlike the onroad standards, however, the nonroad standards provide different numerical limits depending on engine size, with smaller engines allowed to have relatively higher emissions. For example, the Tier 1 PM limits for a 175–300 horsepower engine were 0.40 g/bhp-hr, while they were 0.75 g/bhp-hr for an engine smaller than 11 horsepower.

In addition, the Tier 1–3 nonroad limits were significantly less strict than those in place for onroad engines at the same time. For example, a 250 horsepower engine installed in an onroad truck in 2000 would have had to meet a NO_X standard of 4.0 g/bhp-hr and a PM standard of 0.10 g/bhp-hr; virtually the same engine installed in an agricultural tractor in the same year would have been allowed to emit 6.9 g/bhp-hr NO_X and 0.40 g/bhp-hr PM.

When fully implemented in 2016, the Tier 4 nonroad limits will introduce general parity between onroad and nonroad standards for all diesel engines other than the smallest and largest engines. Nonroad engines smaller than 75 horsepower and some engines greater than 750 horsepower will still be allowed to emit significantly more NO_X and PM than onroad and other nonroad engines.

EPA has separate limits for heavy-duty spark-ignition engines (primarily fueled by gasoline). Given the relatively small stock of heavy-duty nonroad equipment powered by such engines, these limits are not discussed here.

EPA New Engine Emission Standards: Locomotives

EPA first regulated emissions from locomotive engines in 2000. In that year, a set of three standards (Tiers 0–2) became effective, with the operative standard depending on the year the engine was produced. Tier 0 standards apply retroactively to any engine manufactured between 1973 and 2001; Tier 1 applies to new locomotive engines produced from 2002 to 2004; and Tier 2 applies to new locomotive engines produced beginning in 2005 (EPA 2004d).

Unlike other diesel engine standards, the locomotive standards provide two different numerical emissions limits based on two different test cycles that replicate the major modes of locomotive operation: line-haul and switchyard. Line-haul operation is characterized by operating at the highest engine power settings for a significant amount of time. In switchyard operations, the engine spends a significantly greater amount of time at low power settings and at idle. The EPA Tier 0–2 regulations require that each engine meet both sets of numerical standards.

Unlike other diesel emissions standards, the Tier 0–2 locomotive standards apply not only when the engine is first produced, but also every time the engine is re-manufactured. In effect, this means that every locomotive engine built since 1973 must be upgraded to meet Tier 0 standards when it next has a major overhaul. It is estimated that upgrading an unregulated locomotive engine to Tier 0 standards reduces NO_X emissions by approximately 30%, but it produces no appreciable reduction in PM emissions. New engines built to comply with Tier 2 standards will have approximately 60% lower NO_X and 40% lower PM emissions than unregulated engines (DieselNet 2005).

In June 2004, EPA issued an Advanced Notice of Proposed Rulemaking indicating that it was considering Tier 3 locomotive engine standards. These standards would be implemented for new locomotive engines built as early as model year 2011 and would seek to reduce both PM and NO_X emissions by 90% or more compared to the Tier 2 standards. This would bring locomotive engine emissions roughly in line with emissions from onroad trucks under the 2007 rules and with emissions from other nonroad engines under the Tier 4 rules (EPA 2004c).

EPA New Engine Emission Standards: Marine Engines

For regulatory purposes, marine engines are divided into three categories based on size (cylinder displacement). The smallest Category 1 engines are very similar to the diesel engines used in onroad trucks and nonroad land-based vehicles. These marine engines are used for propulsion power in personal and some small commercial vessels such as tugboats and other harbor craft. They may also be used as auxiliary engines in larger commercial vessels. Category 2 engines, which are usually larger than 1,000 horsepower, are virtually identical to locomotive engines and are used for propulsion and auxiliary power in large commercial vessels.

The large Category 3 marine engines are unique. Ranging in size from 3,000 to 100,000 horsepower, they are used for propulsion power in very large ocean-going vessels such as cruise ships, freighters, and tankers.

In 1999, EPA adopted Tier 2 emissions standards for new Category 1 and Category 2 marine engines. The Tier 2 standards take effect between model years 2004 and 2007. These standards, which vary by engine size, set numerical limits that, for Category 1 engines, are similar to the Tier 2 nonroad standards and, for Category 2 engines, are similar to the Tier 1 locomotive standards. This regulation also designates voluntary "blue sky" emissions standards to which engine manufacturers may choose to certify. A blue sky certified marine engine has 40–50% lower NO_X and PM emissions than a Tier 2 engine (DieselNet 2005).

In 2003, EPA adopted Tier 1 NO_X standards for new Category 3 marine engines. These standards take effect beginning in model year 2004. They are equivalent to the limits in Annex VI to the International Convention for the Prevention of Pollution from Ships (also known as the MARPOL Convention), which was negotiated internationally under the auspices of the International Maritime Organization (IMO). The Tier 1 NO_X standards for Category 3 engines are 9.8–17 g/kWh, depending on engine speed; slower engines are allowed to emit more (EPA 2004i). PM and other pollutants are not regulated under either MARPOL Annex VI or the EPA Tier 1 standards.

As with locomotive engines, EPA issued an Advanced Notice of Proposed Rulemaking in June 2004 indicating that it was considering Tier 3 standards for Category 1 and 2 marine engines, to be implemented as early as model year 2011. These proposed standards would seek to reduce both PM and NO_X emissions by 90% or more compared to the Tier 2 standards and would bring Category 1 and 2 marine engine emissions roughly in line with emissions from other nonroad engines under the nonroad Tier 4 rules (EPA 2004c).

To date, EPA has not proposed any tightening of the current Tier 1 emissions standards for Category 3 marine engines. However, the current Administration, through the State Department, has supported MARPOL Annex VI and has indicated a desire to further tighten NO_X standards and apply PM limits in future IMO negotiations (Argus 2005). The international nature of ocean-going vessels complicates efforts to effectively regulate their emissions without international agreements.

Other Regulations: California Nonroad Standards

As noted previously, the Clean Air Act—subject to certain exceptions (e.g., locomotives)—allows California to set its own emissions standards for both new and existing nonroad equipment. However, California has, for the most part, adopted EPA regulations for new nonroad diesel equipment with only slight modifications. (The same is true for California's onroad diesel rules, although its nonroad gasoline rules for equipment like leaf blowers and marine pleasure craft outboard engines are stricter than federal requirements.) California has begun to adopt mandatory retrofit requirements for some categories of nonroad equipment.

FUEL QUALITY

Congress has also given EPA authority to regulate the quality of fuel and fuel additives used in vehicles, based on direct health effects from the fuel itself and the effect of fuel on the ability of vehicles to meet emissions standards. In the past, this has resulted in rules banning lead in gasoline, as well as limits on the volatility of gasoline sold in the summer to reduce evaporative emissions, and limits on allowable levels of sulfur in gasoline and onroad diesel fuel.

Gasoline Standards

Beginning in 1995, the Clean Air Act required that certain cities with severe ozone problems use reformulated gasoline (RFG) in the summer, and that areas in nonattainment for CO use oxygenated gasoline in the winter. The Clean Air Act requires the use of RFG in cities in 14 states, but it allows limited opt-in for other areas. The Energy Policy Act of 2005 allows opt-in to the RFG program at the request of a state, for the jurisdictions within the Ozone Transport Region.[51] Currently, RFG is used in part or all of 19 states and the District of Columbia;[52] approximately 30% of all gasoline sold in the U.S. is reformulated (EPA 2005a).

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[51] The jurisdictions within the ozone transport region are Connecticut, Delaware, District of Columbia, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont, and Virginia.

[52] The jurisdictions using RFG are all of the jurisdictions within the Ozone Transport Region, with the exception of Vermont and Maine. In addition, California, Georgia, Illinois, Indiana, Kentucky, Louisiana, Missouri, Texas, and Wisconsin use RFG.

RFG is formulated to meet a number of different specifications aimed at reducing emissions of ozone precursors and toxics such as benzene. One of these requirements is that it contains 2% oxygen by weight. The most commonly used oxygenate additive is ethanol, which is manufactured primarily from corn. Another additive that can be used as an oxygenate is methyl tertiary butyl ether (MTBE). MTBE is controversial because it can contaminate ground water. In addition, many people believe that RFG does not need to contain oxygen to improve air quality. California, New York, and Connecticut petitioned EPA to have the oxygen requirement removed from their RFG standards, but EPA denied their requests (EPA 2005a). These denials were made moot by the Energy Policy Act of 2005, which removed the requirement for RFG to contain oxygen, effective immediately in California and beginning in May 2006 everywhere else. While early versions of the legislation also contained an outright ban on MTBE in gasoline, this provision did not make it into the enacted law. California, New York, and Connecticut have all taken individual action that partially or completely bans the use of MTBE in gasoline.

The Energy Policy Act of 2005 also contains a requirement that renewable fuels begin to make up a specified portion of the gasoline sold in the U.S. The law requires that 4 billion gallons of renewable fuel be used in 2006, increasing to 7.5 billion gallons per year in 2012. The most likely fuel to be used to meet this requirement is ethanol from corn or other biomass source.

Based on the detrimental effect of sulfur on modern emissions after-treatment systems, EPA has also recently focused on reducing sulfur levels in both gasoline and diesel fuel. The Tier 2 light-duty regulations require that most gasoline refiners and importers meet a corporate average sulfur standard of 120 parts per million (ppm), with an absolute cap of 300 ppm, beginning in 2004. This will be reduced to an average of 30 ppm with a cap of 80 ppm by the end of 2006 (EPA 2004b).

Diesel Fuel Standards

EPA began to regulate the sulfur content of onroad diesel fuel in 1994, setting a limit of 500 parts per million (ppm). The sulfur content of fuel used in nonroad diesel engines was unregulated at that time: distillate fuels currently used in construction equipment, locomotives, and many marine vessels typically have sulfur levels as high as 3,000 ppm. The sulfur content of the heavier residual oils used to power ocean-going ships can be even higher, as much as 50,000 ppm or more.

In order to allow the use of advanced catalytic emissions controls that will be required to meet the 2007 PM and NO_X standards for heavy-duty vehicles, the most recent EPA regulations further reduce the allowable level of sulfur in onroad diesel fuel to 15 ppm beginning in late 2006 (EPA 2001). Under EPA rules, the sulfur level of nonroad diesel fuel will also be controlled for the first time. All nonroad diesel fuel will be limited to 500 ppm sulfur beginning in June 2007. Fuel sulfur limits will be reduced to 15 ppm for nonroad vehicles in 2010, and for locomotives and marine distillate fuels in 2012 (EPA 2004d).

These sulfur limits do not apply to the residual fuels used in ocean-going ships, which remain unregulated in the U.S. IMO MARPOL Annex VI establishes a global cap of 45,000 ppm on the sulfur content of marine residual fuels. This treaty also includes a mechanism to establish a 15,000 ppm cap in specific Sulfur Emission Control Areas (SECA)—typically near coasts—where additional emissions reductions are considered critical. While the U.S. has not formally ratified Annex VI, the Administration has supported ratification together with the designation of U.S. coastal waters as a SECA zone (Argus 2005).

The Clean Air Act allows California to establish its own regulations on the quality of fuel, both gasoline and onroad and nonroad diesel. Under the California rules, nonroad diesel fuel sold in California for other than locomotive or marine applications will be limited to 15 ppm sulfur beginning in 2006, four years earlier than the nonroad diesel sold in the rest of the country (CARB 2004a).

Appendix E

SUPPORTING INFORMATION RELATED TO AIRPORTS AND AIRCRAFT

EMISSIONS MODELING FOR AIRPORTS

There are very few instances in which specific, comprehensive emissions inventories have been completed for airports. In most circumstances, the National Emissions Inventory (NEI) information, as it pertains to airports, is based on default emissions estimating techniques. The purpose of this section is to inform the reader regarding modeling tools available to develop airport-specific emissions estimates. Table E-1 provides an example of the differences that occur when actual equipment counts and activity are used in place of default assumptions to estimate airport emissions.

Table E-1. Differences Between Default and Actual Values for Airport Equipment and Emissions.

	EPA Default Value	New Jersey DEP Specific Value
GSE Population	496	3730
GSE PM Emissions (TPY)	21.4	74.3
Aircraft PM Emissions (TPY)	11.7	41.3

The Emissions and Dispersion Modeling System (EDMS) was developed jointly by the Federal Aviation Administration (FAA EDMS 2004) and the United States Air Force (USAF) in the mid-1980s to assess air quality impacts of proposed airport development projects. In 1993, the U.S. Environmental Protection Agency (EPA) accepted the EDMS as a formal EPA "Preferred Guideline" (EPA NAAQS 2005) model for use in civil airports and military bases. In response to growing needs for air quality analyses and changes in regulations, primarily conformity requirements under the Clean Air Act Amendments of 1990, the FAA and the USAF re-engineered and enhanced EDMS in 1997. In 2001, EDMS was updated again to take advantage of the EPA's state of the art dispersion software, AERMOD. One year later, an updated EDMS was released, which included updated Ground Support Equipment (GSE) emission factors based on EPA's NONROAD model.

In 2002, EPA developed a mobile source emissions inventory model for nonroad equipment, NONROAD2002, covering all equipment except locomotives, aircraft, and commercial marine vessels (EPA NONROAD 2005). NONROAD2002 predicted exhaust emissions and diurnal and refueling emissions of hydrocarbons (HC), carbon monoxide (CO), oxides of nitrogen (NO_X), sulfur oxides (SO_X), exhaust particulate matter (PM), carbon dioxide (CO_2), as well as volume of fuel consumed by all types of mobile nonroad equipment except locomotives, aircraft, and commercial marine vessels. The level of detail from the model included fuel type (diesel, gasoline, LPG, and CNG), individual Source Category Classification (SCC), power range, geographic area (nationwide, state, or county), and temporal period (annual, seasonal, monthly, weekday/weekend) for calendar years 1970 to 2050.

For its NONROAD2005 model, EPA has chosen to allocate GSE population and emissions in proportion to the estimated emissions of aircraft NO_X, as reported in the 2002 National Emissions Inventory (NEI). The use of aircraft

NO_X provides an indication of the relative numbers of aircraft operations at different airports, with the greatest weight given to commercial aircraft, and especially larger commercial aircraft that would require most of the GSE. The actual proportioning methodology is rather simplistic. If the aircraft NO_X emissions (TPY) number is multiplied by 0.0106, the product yields GSE PM emissions (TPY). Similarly, if the aircraft NO_X emissions number is multiplied by 0.246, the product yields GSE population for the given airport.

EMISSIONS STANDARDS FOR ENGINES USED IN GSE

Knowing what the federal standards are, both for diesel engines and for spark-ignition (gasoline) engines, may be a helpful aid to estimating emissions from GSE where actual equipment inventories exist. Table E-2 and Table E-3 are presented for this purpose.

Table E-2. EPA Emissions Standards for Diesel Equipment.

Source: EPA Diesel 2002

Power Rating (HP)	Model Year	PM Emission Standard (g/bhp-hr) *(ABT)
11	2000	0.74
< 11	2005+	0.60
11 • x < 25	2000+	0.60
25 • x < 50	2000	0.60
25 • x < 50	2005+	0.44
50 • x < 100	2004+	0.30
100 • x < 175	2003+	0.22
175 • x < 750	1996+	0.4 (no ABT)
175 • x < 300	2003+	0.15
300 • x < 600	2001+	0.15
600 • x < 750	2002+	0.15
• 750	2000	0.4 (no ABT)
• 750	2006+	0.15

*ABT indicates a standard for which Averaging, Banking, and Trading is allowed as a means to achieve fleetwide compliance.

Table E-3. EPA Emission Standards for Large SI Engines.

Source: EPA-Nonroad 2002

Tier/Year	HC+NO_x	CO
Tier I/2004	4.0 g/kW-hr	50 g/kW-hr
Tier II/2007	2.7 g/kW-hr	4.4 g/kW-hr

BRIEF PROFILES OF AIRPORTS IN NEW YORK STATE

There are three major airports in the New York metropolitan area and several smaller airports scattered around New York State. The following subsection provides additional profile information on the various airports, including the air quality attainment status for the areas in which the airports are located and the major airlines serving the airports.

Table E-4. Commercial Service Airports and Nonattainment Status.

Source: Federal Aviation Administration – Updated September 7, 2005

ST	Airport	Code	City	EPA GREENNAME	County	Ozone (8 Hr)	$PM_{2.5}$
NJ	Newark Liberty International	EWR	Newark	New York-N. New Jersey-Long Island, NY-NJ-CT	Essex; Union	Moderate	V
NY	John F. Kennedy International	JFK	New York	New York-N. New Jersey-Long Island, NY-NJ-CT	Queens	Moderate	V
NY	LaGuardia	LGA	New York	New York-N. New Jersey-Long Island, NY-NJ-CT	Queens	Moderate	V
NY	Buffalo Niagara International	BUF	Buffalo	Buffalo-Niagara Falls, NY	Erie	Subpart 1	
NY	Albany International	ALB	Albany	Albany-Schenectady-Troy, NY	Albany	Subpart 1	
NY	Greater Rochester International	ROC	Rochester	Rochester, NY	Monroe	Subpart 1	
NY	Syracuse Hancock International	SYR	Syracuse	Syracuse, NY	Onondaga		
NY	Long Island MacArthur	ISP	Islip	New York-N. New Jersey-Long Island, NY-NJ-CT	Suffolk	Moderate	V
NY	Westchester County	HPN	White Plains	New York-N. New Jersey-Long Island, NY-NJ-CT	Westchester	Moderate	V
NY	Stewart International	SWF	Newburgh	Poughkeepsie, NY	Orange	Moderate	V
NY	Greater Binghamton	BGM	Binghamton		Broome		
NY	Elmira Corning Regional	ELM	Elmira Corning		Chemung		
NY	Ithaca Tompkins Regional	ITH	Ithaca		Tompkins		
NY	Chautauqua County Jamestown	JHW	Jamestown	Jamestown, NY	Chautauqua	Subpart 1	

John F. Kennedy International Airport (JFK) is operated by The Port Authority of New York and New Jersey (Port Authority 2006) under a lease with the City of New York since 1947. JFK is located in the southeastern section of Queens County, New York City, on Jamaica Bay. It is 15 miles by highway from midtown Manhattan. JFK covers 4,930 acres, including 880 acres in the Central Terminal Area (CTA). The airport has more than 30 miles of roadway. First commercial flights began in 1948. About $150 million was expended on original construction. The Port Authority has invested more than $4.3 billion in the airport. Over 35,000 people are employed at the airport. Kennedy Airport contributes approximately $30.1 billion in economic activity to the NY/NJ metropolitan region, generating some 265,180 jobs and $11.2 billion in wages and salaries. AirTrain JFK, a light rail transit system that connects airport passengers to New York City subways and buses and the Long Island Rail Road (LIRR), opened in December 2003.

The site that became **LaGuardia Airport (LGA)** was initially operated as a 105-acre private flying field, beginning in 1929 (Port Authority 2006). Ground was broken in 1937 for a new airport, which was built jointly by the City of New York and the Federal Works Progress Administration. It was dedicated in 1939 as New York City Municipal Airport. Later that same year, the name was changed to New York Municipal Airport-LaGuardia Field, and the airport opened to commercial traffic. LaGuardia Airport has been operated by The Port Authority of New York and New Jersey under a lease with the City of New York since 1947. Located in the Borough of Queens, New York City, it is eight miles from midtown Manhattan. LaGuardia covers 680 acres and has 72 aircraft gates. Original construction by the City of New York cost $40 million. The Port Authority's total capital investment in LaGuardia Airport to date is over $1 billion. Nearly 8,000 people are employed at the airport. LaGuardia contributes $6.8 billion in economic activity to the NY/NJ metropolitan region, generating more than 60,000 jobs and $2.4 billion in annual wages.

The site that became **Newark Liberty International Airport (EWR)** opened in 1928 as the metropolitan region's first major airport (Port Authority 2006). It was built by the City of Newark on 68 acres of marshland and quickly became the world's busiest commercial airport. During World War II, the Army Air Corps operated the facility. The Port Authority of New York and New Jersey has operated Newark Liberty International Airport under a lease with the City of Newark since 1948. Newark Liberty is located in Essex and Union Counties, New Jersey, and is about 16 miles from midtown Manhattan. Today, EWR covers about 2,027 acres, and the Port Authority is nearing completion of a $3.8 billion redevelopment program that includes: the extension of the AirTrain system, a second International Arrivals Facility, modernized passenger terminals, improved airport access, additional parking facilities, expanded roadways, and improved runways and taxiways. Over 24,000 people are employed at the airport. EWR contributes $11.3 billion in economic activity to the NY/NJ metropolitan region, including $3.3 billion in wages for some 110,000 jobs derived from airport activity. AirTrain Newark provides service 24 hours per day between terminals, parking lots, the rental car complex, and the train station where AirTrain connects passengers to NJ Transit and Amtrak Northeast Corridor train lines for locations north and south of the airport. There are 14 miles of roadways in the airport's Central Terminal Area and over 20,000 parking spaces airport-wide.

What is now **Buffalo Niagara International Airport (BUF)** was first created in 1926 on 518 acres of land (Niagara 2006). Today's BUF covers almost 1,000 acres. Passenger and airmail service between Buffalo and Cleveland commenced in mid-December 1927. In 1956, the Niagara Frontier Port Authority, a public body created by the New York State Legislature, acquired the airport from the City of Buffalo. It continued to operate the airport until it was assimilated into the Niagara Frontier Transportation Authority (NFTA), created by the New York State Legislature in 1967. All subsequent work on the airport was carried out by the NFTA. In 1959, the name of the airport was changed to the Greater Buffalo International Airport.

The 1200 acre **Albany International Airport (ALB)**, operated by the Albany County Airport Authority under a 40-year lease with Albany County, is the major air center for the Capital Region, northeastern New York, and western New England (Albany 2006). The airport has embarked on a five-year $232 million Capital Plan to improve and maintain safety and to meet future travel demand. The new terminal, which opened in June 1998, is designed to accommodate 1.5 million passenger boardings annually. ALB handles an average of 110 daily commercial arrivals and departures and regulates 1,000 tower operations each day. Currently, the airport offers parking for over 5,000 vehicles. ALB also operates a full-service Air Cargo Terminal serving FedEx, UPS, DHL and Mobile Air.

Greater Rochester International Airport (ROC) is a medium hub airport that handles 220 flights per day to over 20 cities (22 airports), serves more than 2.5 million passengers annually, and houses the operations of 16 air transportation providers (Monroe 2006). ROC provides service to cities in the Northeast and to major hubs in the Midwest. The airport is located four miles southwest of the City of Rochester and 12 miles south of Lake Ontario. The first development of the Greater Rochester International Airport was the construction of Hangar #1 in 1927. The airport was then named Britton Field. In 1928, the name was changed to Rochester Municipal Airport. From 1940 through the World War II years, the airport saw significant increases in scheduled flight activity and civilian pilot training. The Greater Rochester International Airport was taken over by the County of Monroe in 1948, adding a second runway and lengthening the original runway.

In 1927, land at Hinsdale Field in Amboy was chosen by the City of Syracuse as the site of the Syracuse Airport (Syracuse 2006). With the outbreak of World War II, the Amboy City Airport became a flight training center. At the end of 1941, the Army Air Corps authorized the construction of an air base at Syracuse and in 1942, three 5,500 foot runways were built by the Army. In 1946, the City of Syracuse took over the base on an interim lease and in 1948, the base was dedicated as a commercial airfield. The Clarence E. Hancock Airport opened in 1949. Today, **Syracuse Hancock International Airport (SYR)** has two runways open for use.

Islip Long Island MacArthur Airport (ISP) is located on Long Island in Ronkonkoma, Suffolk County, New York. It is owned and operated by the Town of Islip (Islip 2006). It is the only airport in Suffolk or Nassau County with scheduled service on major airlines. It serves over two million passengers a year. The airport began in 1942 as three paved runways built by the Civil Aeronautics Administration. The Town of Islip built a terminal in 1949, after taking the airport back from the Army Air Corps at the end of World War II. Allegheny Airlines was its first commercial airline in 1960, offering flights to Boston, Philadelphia, and Washington.

Westchester County Airport (HPN) was originally built as a defense satellite base for New York City. Shortly following WWII, the airport became home to an Air National Guard Unit; in 1947, airlines began the first scheduled commercial service from White Plains (Westchester 2006). Today, the Airport has earned a reputation as a general aviation airport and is noted as having one of the largest corporate fleets in the United States. The 700 acre facility has two intersecting runways, the longest 6,550 ft. The airport serves several commercial service operators and over 400 based aircraft, including helicopters. Passengers from New York and Connecticut frequent the airport for its nonstop commercial services to 10 major cities. Recently, Westchester County Airport enplaned and deplaned over one million passengers annually. In addition to the airport's title as a regional transportation hub, the 50 major companies located on the property also serve as an employment center. An updated study by the Westchester County Planning Department indicates that the airport employs over 1,500 people and provides a total economic impact greater than $600,000,000 annually.

Stewart International Airport (SWF) is the nation's first privatized commercial airport and operates under a 99-year lease agreement with the New York State Department of Transportation (Stewart 2006). SWF is located in the Towns of New Windsor and Newburgh, Orange County, New York, approximately five miles west of the City of Newburgh and approximately 60 miles north of midtown Manhattan. The airport has two runways. At 11,818 feet long and 150 feet wide, the primary runway is one of the longest runways in the Northeastern U.S.

The **Greater Binghamton Airport (BGM)** is owned and operated by the Broome County Department of Aviation. It is located eight miles north of the Binghamton Metropolitan area, providing for the transportation needs of the Greater Binghamton Area as well as other surrounding communities in southern New York and northern Pennsylvania (Binghamton 2006). There are over 40 daily flights (of which one-third are operated with jet aircraft) provided by four airlines to four major hub airports. In the 1940s, the demand for more scheduled air service increased in the Southern Tier of New York, which was then served by the Tri-Cities Airport located in Endicott. By 1944, the Civil Aeronautics Authority (CAA) ruled that the Tri-Cities Airport would not be approved for any other commercial operation except clear, daytime flights due to its location and terrain. In 1951, a new airport was dedicated at the present location of BGM. Since its opening, the airport has had several expansion projects, including a runway expansion (the North-South runway is currently 7,500 feet long), completed in 1988.

Elmira-Corning Regional Airport (ELM) has served the Southern Tier of central New York and Northern Tier of central Pennsylvania since 1945 (Elmira 2006). In the 1920s, American Airways (now American Airlines) needed a field to serve as an emergency landing strip on a New York-to-Buffalo air route. American Airways leased about 100 acres of land in the Big Flats area for that purpose, and from that grass field a modern air facility developed. In

1940, the federal government designated this field as a Defense Landing Area. Chemung County purchased an additional 239 acres, and the federal government installed three hard-surfaced runways, each being 4000 feet long. In 1943, the federal government turned the airport over to the jurisdiction of Chemung County.

Ithaca Tompkins Regional Airport (ITH) is owned and operated by Tompkins County. East Hill Airport was opened by Cornell University in 1948 (Ithaca 2006). In 1956, Tompkins County Board purchased the East Hill Airport from Cornell and renamed it Tompkins County Airport. In 1994, two major airport development projects were completed: an 800-foot runway extension lengthened the paved runway to 6,601 feet; and a new 33,000 square foot terminal was built, including parking improvements for both autos and aircraft. In 2001, the Tompkins County Board of Representatives adopted the new name of Ithaca Tompkins Regional Airport. The airport contributes more than $30 million annually to the local economy. The total number of aviation-related jobs created is almost 400 with a combined payroll of more than $5.3 million.

Chautauqua County considers itself the western gateway to New York State, occupying the extreme southwest corner of the state. It is considered to be a tourist destination because of its lakes, including approximately 50 miles of Lake Erie shoreline. **Chautauqua County Jamestown Airport (JHW)** offers commuter service to and from Pittsburgh (PA) International Airport (Chautauqua 2006).

Table E-5. Major North American Commercial Airlines Serving Selected Airports.

Airport• Airline•	Newark Liberty	John F. Kennedy	La-Guardia	Buffalo Niagara	Albany	Greater Rochester	Syracuse Hancock	Long Island Mac-Arthur	West-chester County	Stewart Newburgh	Greater Binghamton	Elmira Corning	Ithaca Tomkins	Chautauqua Jamestown
Aero Mexico		X												
Air Canada	X	X	X		X	X			X					
Air Tran	X		X	X		X								
Alaska	X													
America West	X	X												
American	X	X	X	X	X	X	X		X	X				
ATA			X											
Comair	X	X	X	X										
Continental	X	X	X	X	X		X	X	X			X		
Delta	X	X	X	X	X	X	X	X	X		X			
Frontier			X											
jetBlue		X	X	X		X	X							
MexiCana	X	X												
Midwest	X		X											
Northwest	X	X	X	X	X	X	X	X	X	X	X	X	X	
Song		X	X											
Southwest				X	X			X						
United	X	X	X	X	X	X	X		X		X			
U.S. Airways	X		X	X	X	X	X	X	X	X	X	X	X	X

Appendix F

PM2.5 SPECIATION PROFILES

Table F-1. PM2.5 Speciation Profiles.
Source: U.S. Environmental Protection Agency

Profile #	Profile Name	SPEC	POA	PEC	GSO4	PNO3	OTHER
22001*	Coal Combustion	pm2.5	0.0107	0.0183	0.1190	0.0000	0.8520
22002*	Residual Oil Combustion	pm2.5	0.1075	0.0869	0.5504	0.0005	0.2547
22003*	Distillate Oil Combustion	pm2.5	0.0384	0.0770	0.3217	0.0024	0.5605
22004	Natural Gas Combustion	pm2.5	0.6000	0.0000	0.2000	0.0055	0.1945
22005	Petroleum Heaters	pm2.5	0.0840	0.0000	0.4700	0.0055	0.4405
22006	Residential Heating	pm2.5	0.8642	0.0000	0.1069	0.0289	0.0000
22007	Liquid Waste Combustion	pm2.5	0.0540	0.1050	0.0680	0.0000	0.7730
22008*	Wood Waste Boiler	pm2.5	0.1177	0.2019	0.0282	0.0009	0.6513
22009	Solid Waste Combustion	pm2.5	0.0068	0.0350	0.0680	0.0000	0.8902
22010	Jet Fuel Combustion	pm2.5	0.2921	0.6587	0.0460	0.0032	0.0000
22011	Organic Chemical Manufacture	pm2.5	0.3720	0.0200	0.0401	0.0055	0.5624
22012	Carbon Black Manufacture	pm2.5	0.9960	0.0000	0.0000	0.0000	0.0040
22013	Inorganic Chemical Manufacture	pm2.5	0.0000	0.0000	0.0313	0.0035	0.9652
22014	Phosphate Manufacture	pm2.5	0.1126	0.0164	0.0564	0.0089	0.8057
22015	Chem Manufacture - Avg	pm2.5	0.1102	0.0183	0.0313	0.0035	0.8367
22016	Charcoal Manufacture	pm2.5	0.0216	0.0520	0.0650	0.0030	0.8584
22017	Paint Manufacture	pm2.5	0.4800	0.1000	0.0200	0.0005	0.3995
22018	Charcoal Broiling	pm2.5	0.9728	0.0123	0.0021	0.0002	0.0126
22019	Food & Ag - Handling	pm2.5	0.3600	0.0000	0.0055	0.0000	0.6345
22020	Food & Ag -Drying	pm2.5	0.1440	0.0000	0.0005	0.0005	0.8550
22021	Sintering Furnace	pm2.5	0.1310	0.0000	0.1991	0.0000	0.6699
22022	Open Hearth Furnace	pm2.5	0.2400	0.0000	0.3989	0.0055	0.3556
22023	Basic Oxygen Furnace	pm2.5	0.2400	0.0000	0.4000	0.0055	0.3545
22024	Electric Arc Furnace	pm2.5	0.2400	0.0000	0.0250	0.0055	0.7295
22025	Ferromanganese Furnace	pm2.5	0.1080	0.0150	0.0420	0.0570	0.7780
22026	Aluminum Production	pm2.5	0.0468	0.0230	0.0440	0.0041	0.8821
22027	Copper Production	pm2.5	0.0000	0.0000	0.0000	0.0000	1.0000
22028	Lead Production	pm2.5	0.0000	0.0000	0.0000	0.0000	1.0000
22029	Heat Treating	pm2.5	0.0766	0.0100	0.2809	0.0055	0.6270
22030	Secondary Aluminum	pm2.5	0.1560	0.0000	0.1600	0.0055	0.6785
22031	Secondary Copper	pm2.5	0.0120	0.0010	0.0005	0.0000	0.9865
22032	Secondary Lead	pm2.5	0.0000	0.0000	0.0753	0.0000	0.9247
22033	Fiberglass Manufacture	pm2.5	0.3360	0.0200	0.0055	0.0055	0.6330
22034	Brick Grinding & Screening	pm2.5	0.0000	0.0000	0.0055	0.0005	0.9940
22035	Asphaltic Concrete Manufacture	pm2.5	0.1534	0.0000	0.0055	0.0000	0.8411
22036	Asphalt Roofing	pm2.5	0.2760	0.0100	0.2290	0.0000	0.4850
22037	Glass Manufacture	pm2.5	0.0085	0.0006	0.4890	0.0002	0.5017
22038	Sand & Gravel	pm2.5	0.0000	0.0000	0.0055	0.0000	0.9945
22039	Clay/Fly Ash Sintering	pm2.5	0.1722	0.0744	0.0960	0.0109	0.6465
22040	Mineral Products - Avg	pm2.5	0.0631	0.0147	0.1406	0.0027	0.7789
22041	Catalytic Cracking	pm2.5	0.0331	0.0092	0.4643	0.0000	0.4934
22042	Organic Liquid	pm2.5	1.0000	0.0000	0.0000	0.0000	0.0000

Table F-1. PM2.5 Speciation Profiles.
Source: U.S. Environmental Protection Agency

Profile #	Profile Name	SPEC	POA	PEC	GSO4	PNO3	OTHER
22043	Petroleum Ind - Avg	pm2.5	0.0420	0.0000	0.2350	0.0028	0.7202
22044	Kraft Recovery Furnace	pm2.5	0.0628	0.0153	0.5485	0.0035	0.3699
22045	Pulp & Paper - Avg	pm2.5	0.3569	0.0263	0.2040	0.0037	0.4091
22046	Wood Products - Drying	pm2.5	0.7900	0.0439	0.0365	0.0125	0.1171
22047	Wood Products - Sanding	pm2.5	0.4200	0.0600	0.0000	0.0000	0.5200
22048	Wood Products - Sawing	pm2.5	0.4680	0.0300	0.0055	0.0055	0.4910
22049	Sandblasting	pm2.5	0.0006	0.0000	0.0055	0.0005	0.9934
22050	Cement Production	pm2.5	0.0480	0.0343	0.3514	0.0055	0.5608
22051	Gypsum Manufacture	pm2.5	0.0258	0.0100	0.5174	0.0005	0.4463
22052	Ammonium Nitrate Production	pm2.5	0.0000	0.0000	0.0000	0.7750	0.2250
22053	Ammonium Sulfate Production	pm2.5	0.0000	0.0000	0.7270	0.0000	0.2730
22054	Ind Manufacture - Avg	pm2.5	0.0883	0.0089	0.0986	0.0030	0.8012
22055*	Gasoline Vehicles	pm2.5	0.6774	0.1658	0.0229	0.0132	0.1207
22056*	Diesel Vehicles	pm2.5	0.2352	0.6410	0.0490	0.0033	0.0715
22057*	Non-catalyst Gasoline	pm2.5	0.7860	0.0801	0.0045	0.0063	0.1231
22058*	Paved Road Dust	pm2.5	0.1768	0.0112	0.0070	0.0022	0.8028
22059	Soil Dust	pm2.5	0.0545	0.0037	0.0004	0.0011	0.9403
22060*	Agricultural Burning	pm2.5	0.6389	0.0750	0.0154	0.0063	0.2644
22061	Residential Wood Combustion	pm2.5	0.5656	0.1077	0.0037	0.0022	0.3208
22062	Sea Salt	pm2.5	0.0000	0.0000	0.1000	0.0000	0.9000
22063	Cigarette Smoke	pm2.5	0.9600	0.0250	0.0006	0.0007	0.0137
22064	Tire Dust	pm2.5	0.6996	0.2870	0.0025	0.0015	0.0094
22065	Brake Lining Dust	pm2.5	0.2022	0.0381	0.0334	0.0016	0.7247
22070*	Wildfires	pm2.5	0.6360	0.0900	0.0000	0.0000	0.2740
22101*	Coal Combustion	pm2.5	0.0121	0.0208	0	0	0.9671
22102*	Residual Oil Combustion	pm2.5	0.2391	0.1933	0	0.0011	0.5665
22103*	Distillate Oil Combustion	pm2.5	0.0566	0.1135	0	0.0035	0.8264
35500*	LDGV 1996 Onroad, exhaust only	pm2.5	0.7059	0.2399	0.0355	0.0013	0.0174
35501*	LDGV 1996 Onroad	pm2.5	0.5682	0.1901	0.0324	0.0014	0.2079
35531*	LDGV 2030&2020 Onroad	pm2.5	0.4336	0.1413	0.0169	0.0011	0.4027
35600*	HDDV 1996 Onroad	pm2.5	0.2271	0.7500	0.0041	0.0017	0.0171
35601*	LDDV 1996 Onroad	pm2.5	0.3637	0.6130	0.0044	0.0006	0.0183
35602	Non-road Diesel 49-State 1996	pm2.5	0.2244	0.7411	0.0329	0.0016	0.0000
35603	Non-road Diesel Calif 1996	pm2.5	0.2271	0.7500	0.0015	0.0017	0.0197
35621	LDDV 2020 Onroad, base	pm2.5	0.3383	0.5261	0.0304	0.0007	0.1045
35625*	HDDV 2020 Onroad	pm2.5	0.2483	0.5641	0.0311	0.0021	0.1544
35626	LDDV 2020 Onroad, control	pm2.5	0.2218	0.6369	0.0355	0.0023	0.1035
35630	HDDV 2030&2020 Onroad, base	pm2.5	0.2322	0.7131	0.015	0.0016	0.0322
35631	LDDV 2030 Onroad, base	pm2.5	0.2813	0.3524	0.077	0.002	0.2873
35635*	HDDV 2030 Onroad	pm2.5	0.1792	0.1751	0.0576	0.002	0.5186
35636	LDDV 2030 Onroad, control	pm2.5	0.1289	0.1627	0.0647	0.0021	0.5695
35700*	Nonroad Gasoline 1996	pm2.5	0.7860	0.0801	0.0045	0.0063	0.1231
35730*	Nonroad Gasoline 2030&2020	pm2.5	0.786	0.0801	0.0005	0.0063	0.1265
99999	Overall Average/Default	pm2.5	0.0000	0.0000	0.0000	0.0000	1.0000
NAGBN*	Agricultural Burning	pm2.5	0.67	0.04	0.01	0.003	0.277
NCOAL*	Coal Combustion	pm2.5	0.20	0.01	0.16	0.005	0.625

Table F-1. PM2.5 Speciation Profiles.
Source: U.S. Environmental Protection Agency

Profile #	Profile Name	SPEC	POA	PEC	GSO4	PNO3	OTHER
NPAVE*	Paved Road Dust	pm2.5	0.12	0.0112	0.007	0.0004	0.8614
NWFIR*	Wildfires	pm2.5	0.77	0.16	0.02	0.002	0.0480
NWWAS*	Wood Waste Boiler	pm2.5	0.39	0.14	0.08	0	0.39

Notes:
*22001 see also 22101 and new profile NCOAL
*22002 see also 22102
*22003 see also 22103
*22008 see new profile NWWAS
*22055 gasoline vehicles in EPS-hildemann
*22056 diesel vehicles in EPS-gray
*22057 non-catalyst gasoline in EPS-hildemann
*22058 see new profile NPAVE
*22060 see new profile NAGBN
*22070 from WRAP. See new profile NWFIR
*22101 w/SULF=0 (use when sulfur provided as separate pollutant)
*22102 w/SULF=0 (use when sulfur provided as separate pollutant)
*22103 w/SULF=0 (use when sulfur provided as separate pollutant)
*35500 LDGV 1996 on-road, exhaust only-DRI w/1.2*OC
*35501 LDGV 1996 on-road, w/brake & tire
*35531 base & ctl, w/brake & tire
*35600 HDDV 1996 on-road - DRI comp w/1.2*OC
*35601 LDDV 1996 on-road - DRI comp w/1.2*OC
*35625 composite ctl (fuel & engines)
*35635 composite ctl (fuel & engines)
*35700 non-road gasoline 1996-hildemann (=20057 w/1.2*OC)
*35730 base & ctl
*NAGBN replacement for profile 22060
*NCOAL replacement for profile 22001
*NPAVE replacement for profile 22058
*NWFIR replacement for profile 22070
*NWWAS replacement for profile 22008

Appendix G

CHAPTER 1 SUPPORTING DATA TABLES

Table G-1. Annualized Mass data ($\mu g/m^3$) plotted in Figure 1-2.

	Buff	Roch	Pinn	Canal	IS52	NYBG	QCll	WHTE
Crustal	0.7	0.4	0.3	0.7	0.7	0.7	0.6	0.2
Sulfate	4.3	3.5	3.9	4.3	4.2	4.0	4.2	2.3
Nitrate	2.1	1.8	0.9	2.5	2.2	1.9	2.0	0.4
Ammonium	2.0	1.7	1.2	2.2	2.0	1.8	1.9	0.6
EC	0.6	0.4	0.2	1.3	1.1	1.2	0.7	0.1
OC	4.2	3.3	2.8	5.5	4.0	4.8	3.8	1.8
Other	1.3	1.1	1.4	1.0	1.4	0.5	1.1	1.5

Table G-2. Seasonal mass data ($\mu g/m^3$) plotted in Figures 1-4 & 1-5 (all data through 11/05).

Canal St.	Winter	Spring	Summer	Fall	**Bronx IS52**	Winter	Spring	Summer	Fall
Crustal	0.6	0.7	0.8	0.6	Crustal	0.7	0.7	0.8	0.6
Sulfate	3.4	4.2	6.8	3.0	Sulfate	3.4	4.0	6.3	3.3
Nitrate	3.4	3.2	1.7	2.0	Nitrate	3.0	2.5	1.6	1.8
Ammonium	2.2	2.3	2.8	1.5	Ammonium	2.0	1.9	2.6	1.5
EC	1.5	1.2	1.4	1.2	EC	1.1	1.1	0.9	1.1
OC	5.6	5.3	6.5	4.8	OC	3.8	3.9	5.0	3.6
Other	0.9	0.2	1.8	1.1	Other	1.6	1.0	2.0	0.9
Buffalo	Winter	Spring	Summer	Fall	**Pinnacle SP**	Winter	Spring	Summer	Fall
Crustal	0.5	0.7	0.8	0.6	Crustal	0.2	0.4	0.4	0.2
Sulfate	2.7	3.3	6.7	4.1	Sulfate	2.2	3.5	6.3	3.4
Nitrate	3.3	3.0	1.1	1.4	Nitrate	1.5	1.2	0.3	0.7
Ammonium	1.8	2.0	2.4	1.7	Ammonium	0.9	1.2	1.7	1.1
EC	0.5	0.6	0.7	0.7	EC	0.2	0.2	0.2	0.2
OC	3.8	3.7	5.5	3.7	OC	1.8	2.4	4.2	2.4
Other	1.6	0.5	1.8	0.9	Other	0.6	1.0	2.7	1.1
NY Bot. Gardens	Winter	Spring	Summer	Fall	**Queens College**	Winter	Spring	Summer	Fall
Crustal	0.6	0.7	0.7	0.6	Crustal	0.6	0.6	0.7	0.6
Sulfate	3.4	3.6	5.7	3.4	Sulfate	3.1	3.9	6.4	3.2
Nitrate	3.0	2.1	1.1	1.8	Nitrate	2.8	2.2	1.3	1.7
Ammonium	1.9	1.7	2.1	1.5	Ammonium	1.8	1.8	2.4	1.4
EC	1.6	1.1	1.1	1.2	EC	0.7	0.7	0.7	0.8
OC	4.7	3.8	6.1	4.4	OC	3.3	3.3	5.2	3.3
Other	0.1	0.4	1.0	0.4	Other	1.5	0.6	1.4	0.7
Rochester	Winter	Spring	Summer	Fall	**Whiteface Mountain**	Winter	Spring	Summer	Fall
Crustal	0.3	0.5	0.5	0.4	Crustal	0.1	0.2	0.2	0.1
Sulfate	2.4	3.2	5.0	3.3	Sulfate	1.3	1.8	3.4	2.3
Nitrate	3.1	2.3	0.8	1.5	Nitrate	0.6	0.3	0.1	0.4
Ammonium	1.6	1.7	1.9	1.4	Ammonium	0.5	0.5	0.8	0.6
EC	0.3	0.4	0.4	0.4	EC	0.1	0.2	0.1	0.2
OC	2.6	2.9	4.4	3.0	OC	0.9	1.3	3.2	1.6
Other	1.3	0.6	1.6	0.7	Other	0.9	0.8	2.9	1.0

Table G-3. Monthly mass data ($\mu g/m^3$) plotted in Figures 1-6 and 1-7. The top group shows OC values with no mass multiplier. The bottom group shows EC values. Note the values plotted in Figure 1-6 are the monthly average divided by the annual average.

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual Average
BUFF	2.37	3.10	2.44	2.98	1.83	3.52	3.61	3.10	2.62	2.19	2.33	1.62	2.67
CANL	3.96	4.69	3.68	3.69	4.96	5.22	5.49	4.76	3.11	3.49	4.70	4.47	4.30
IS52	2.36	2.62	2.60	2.72	2.84	2.95	3.92	3.06	2.29	2.28	3.06	2.96	2.78
NYBG	3.62	3.04	2.80	2.42	3.12	4.04	4.38	4.31	3.02	3.20	3.48	3.60	3.41
PINN	1.13	1.08	1.68	1.39	1.43	2.47	2.70	2.34	1.92	1.13	1.38	1.15	1.68
QCII	2.00	2.34	2.19	1.92	2.50	2.73	3.38	3.53	1.83	2.17	2.52	2.26	2.48
ROCH	1.50	1.90	2.06	1.61	1.72	2.57	2.81	2.48	1.99	1.76	1.87	1.40	2.00
WHTE	0.63	0.51	0.78	0.92	0.78	2.09	2.19	1.72	1.38	0.83	0.74	0.49	1.13
BUFF	0.48	0.61	0.58	0.73	0.52	0.72	0.64	0.65	0.70	0.69	0.57	0.41	0.61
CANL	1.44	1.48	1.17	1.24	1.32	1.40	1.36	1.32	1.02	1.19	1.33	1.44	1.30
IS52	0.95	1.10	1.10	1.20	1.01	0.96	0.92	0.86	0.89	1.12	1.23	1.28	1.06
NYBG	1.65	1.51	1.35	1.11	1.01	1.14	0.92	1.10	0.98	1.28	1.43	1.66	1.25
PINN	0.21	0.18	0.24	0.25	0.23	0.21	0.23	0.18	0.20	0.22	0.23	0.24	0.22
QCII	0.66	0.78	0.73	0.68	0.72	0.67	0.67	0.71	0.60	0.79	0.84	0.73	0.72
ROCH	0.34	0.36	0.40	0.36	0.36	0.40	0.40	0.40	0.42	0.41	0.38	0.33	0.38
WHTE	0.15	0.13	0.18	0.15	0.16	0.19	0.12	0.10	0.14	0.15	0.18	0.12	0.15

Table G-4. Seasonal mass data (•g/m3) used as the basis of local and regional comparisons presented in Table 1-5. These data are averaged over the same exact measurement dates for all five sites (unlike the data presented in Table 2 of this appendix).

Specie	Site	Spring	Summer	Fall	Winter	Annual
Nitrate	Pinn	1.06	0.28	0.78	1.38	0.87
	Canal	3.08	1.53	2.21	3.23	2.50
	IS52	2.26	1.47	1.99	2.86	2.13
	NYBG	2.20	1.04	1.69	2.74	1.90
	QCII	2.05	1.26	1.74	2.76	1.94
Sulfate	Pinn	3.08	5.75	3.58	2.14	3.65
	Canal	4.07	6.50	3.17	3.16	4.19
	IS52	3.80	6.31	3.36	3.26	4.16
	NYBG	3.43	5.76	3.15	3.06	3.83
	QCII	3.68	6.23	3.17	2.99	3.99
OC	Pinn	2.37	4.14	1.89	2.40	2.66
	Canal	5.22	6.25	5.49	4.95	5.44
	IS52	3.86	4.45	3.81	3.43	3.85
	NYBG	3.36	5.22	3.97	3.80	4.05
	QCII	3.39	4.63	3.29	3.22	3.59
EC	Pinn	0.24	0.19	0.23	0.21	0.22
	Canal	1.24	1.29	1.25	1.41	1.29
	IS52	1.19	0.85	1.06	1.28	1.10
	NYBG	1.08	1.00	1.18	1.63	1.22
	QCII	0.72	0.66	0.73	0.75	0.72

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ASSESSMENT OF CARBONACEOUS $PM_{2.5}$ FOR NEW YORK AND THE REGION

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