The Nature of the Fine Particle and Regional Haze Air Quality Problems in the MANE-VU Region: A Conceptual Description

Prepared for the Mid-Atlantic/Northeast Visibility Union

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Contributing Authors

Tom Downs, Maine Department of Environmental Protection
Iyad Kheirbek, NESCAUM (now with the NYC Dept. of Health and Mental Hygiene)
Gary Kleiman, NESCAUM (now with Gary Kleiman Environmental Consulting)
Paul Miller, NESCAUM
Leah Weiss, NESCAUM

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Scientific evidence has established a solid link between cardiac and respiratory health risks and transient exposure to ambient fine particle pollution. The same fine particles that are capable of penetrating deep into the lungs are also in the size range that is most efficient at absorbing and scattering visible light, thus impairing visibility. The emission sources, atmospheric chemistry, and meteorological phenomena that influence ambient concentrations of fine particle pollution can act on scales that range from hundreds to thousands of kilometers.

As presented in this report, a conceptual understanding of fine particles from a regional perspective across MANE-VU and throughout the eastern U.S. is well understood, yet remains complex due to the multiplicity of source regions, pollutant species, and seasonal weather patterns that influence fine particle formation. There is a compelling technical case on the need for additional regional measures in the eastern U.S. to reduce particulate levels and protect public health. As this report demonstrates, the reduction of fine particles in the eastern U.S. requires a careful balance of regional and local controls for a range of fine particulate species over the course of a year.

Fine particles may originate as either primary or secondary pollutants; primary fine particles are emitted directly from sources while secondary fine particles form in the atmosphere through chemical reactions of precursors emitted by sources. Exceedances of the fine particle national health standards can occur at any time of the year, with the highest levels reached in the winter. There are important differences in the chemical species that are responsible for high fine particle levels during summer and winter.

In 1999, the U.S. Environmental Protection Agency (USEPA) promulgated the Regional Haze Rule that implements a national visibility goal laid out in the Clean Air Act. This will ultimately restore natural visibility to 156 national parks and wilderness areas across the country (called "Class I" areas). In 2006, the USEPA revised the health-based 24-hour national ambient air quality standard (NAAQS) for fine particles with an aerodynamic diameter of 2.5 micrometers or less ($PM_{2.5}$).[a] To address these Clean Air Act requirements, states will have to develop State Implementation Plans (SIPs) detailing their approaches for reducing $PM_{2.5}$ pollution to meet the NAAQS. They also must develop plans that address the degradation of visibility that exists in various parts of the Mid-Atlantic and Northeast (referred to as the Mid-Atlantic/Northeast Visibility Union (MANE-VU) region). As part of this process, the USEPA urges states to include in their SIPs a conceptual description of the pollution problem in their nonattainment and Class I areas. This document provides the conceptual description of the fine particulate and regional haze problems in the MANE-VU states consistent with the USEPA's guidance.

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[a] The USEPA decided not to revise the annual $PM_{2.5}$ NAAQS of 15 µg/m³ at the time it revised the 24-hour NAAQS, a decision later remanded back to the USEPA by the Court of Appeals for the District of Columbia in early 2009. The annual $PM_{2.5}$ NAAQS, however, remained in place while the USEPA undertook a 5-year review of the standards and the latest scientific information, as required by the Clean Air Act. The statutory deadline for completion of the USEPA's 5-year review was October 17, 2011, which the agency missed. As the result of a federal court order, the USEPA agreed to propose revised standards by June 14, 2012, and issue final standards by December 14, 2012.

Scientific studies of the regional $PM_{2.5}$ problem have uncovered a rich complexity in the interaction of meteorology and topography with $PM_{2.5}$ formation and transport. Large scale high pressure systems covering hundreds of thousands of square miles are the source of classic severe fine particle episodes in the eastern United States, particularly in summer. These large, synoptic scale systems create particularly favorable conditions for the oxidation of sulfur dioxide (SO_2) to various forms of sulfate which, in turn, forms – or is incorporated into – $PM_{2.5}$ that is subsequently transported over large distances. These synoptic scale systems move from west to east across the United States, bringing air pollution emitted by large coal-fired power plants and other sources located outside MANE-VU into the region. This then adds to the pollution burden within MANE-VU on days when MANE-VU's own air pollution sources are themselves contributing to poor air quality. At times, the high pressure systems may stall over the East for days, creating particularly intense $PM_{2.5}$ episodes.

In the winter, temperature inversions occur that are effective at concentrating local primary particle emissions at the surface overnight and during early morning hours. This pollution can then be mixed into regionally transported particle pollution (aloft) later in the morning when convection is restored. Additionally, the lower temperature in the winter can shift the chemical equilibrium in the atmosphere slightly toward the production of nitrate particle pollution relative to sulfate formation. As a result, nitrate can become a significant fraction of measured $PM_{2.5}$ mass in parts of the eastern U.S. during winter months.

Primary and secondary emissions of carbon-containing compounds (e.g., diesel exhaust, biogenic organic carbon emissions, and anthropogenic volatile organic compound emissions) all contribute to a significant presence of carbonaceous aerosol across the MANE-VU region, which can vary from urban to rural locations and on a seasonal basis. In addition, short range pollution transport exists with primary and precursor particle pollutants pushed by land, sea, mountain, and valley breezes that can selectively affect relatively local areas.

With the knowledge of the emission sources, transport scales, atmospheric chemistry, and seasonal meteorology in various locations adjacent to and within MANE-VU, a conceptual picture of fine particle pollution and its impacts emerges. The conceptual description that explains elevated regional $PM_{2.5}$ peak concentrations in the summer differs significantly from that which explains the largely urban peaks observed during winter. On average, summertime concentrations of sulfate in the northeastern United States are more than twice that of the next most important fine particle constituent, organic carbon (OC), and more than four times the combined concentration of nitrate and black carbon (BC) constituents. Episodes of high summertime sulfate concentrations are consistent with stagnant meteorological flow conditions upwind of the MANE-VU region and the accumulation of airborne sulfate (via atmospheric oxidation of SO_2) followed by long-range transport from industrialized areas within and outside the region.

National assessments find that in the winter, sulfate levels in urban areas are higher than background sulfate levels across the eastern U.S., indicating that the local urban contribution to wintertime sulfate levels is significant relative to the regional sulfate contribution from long-range transport. A network analysis for the winter of 2002

suggests that the local enhancement of sulfate in urban areas of the MANE-VU region ranges from 25 to 40 percent and that the long-range transport component of $PM_{2.5}$ sulfate is still the dominant contributor in most eastern cities.

In the winter, urban OC and sulfate each account for about a third of the overall $PM_{2.5}$ mass concentration observed in Philadelphia and New York City. Nitrate also makes a significant contribution to urban $PM_{2.5}$ levels observed in the northeastern United States during the winter months. Wintertime concentrations of OC and nitrate in urban areas can be twice the average regional concentrations of these pollutants, indicating the importance of local source contributions. This is likely because winter conditions are more conducive to the formation of local inversion layers which prevent vertical mixing. Under these conditions, emissions from tailpipe, industrial and other local sources become concentrated near the Earth's surface, adding to background pollution levels associated with regionally transported emissions.

Every air pollution episode is unique in its specific details. The relative influences of the transport pathways and local emissions vary by hour, day, and season. The smaller scale weather patterns that affect pollution accumulation and its transport underscore the importance of local (in-state) controls for SO_2, nitrogen oxides (NO_X) and volatile organic compound (VOC) emissions. Larger synoptic scale weather patterns, and pollution patterns associated with them, support the need for SO_2 and NO_X controls across the broader eastern United States. Studies and characterizations of nocturnal low level jets also support the need for local and regional controls on SO_2 and NO_X sources as locally generated and transported pollution can both be entrained in low level jets formed during nighttime hours. The presence of land, sea, mountain, and valley breezes indicate that there are unique aspects of pollution accumulation and transport that are area-specific and will warrant policy responses at the local and regional levels beyond a one-size-fits-all approach.

The mix of emission controls is also important. Regional fine particle formation is primarily due to SO_2, but NO_X is also important because of its influence on the chemical equilibrium between sulfate and nitrate pollution during winter. While the effect of reductions in anthropogenic VOCs is less well characterized at this time, secondary organic aerosol (SOA) is a major component of fine particles in the region and reductions in anthropogenic sources of OC may have a significant effect on fine particle levels in urban nonattainment areas. Therefore, a combination of localized NO_X and VOC reductions in urban centers with additional SO_2 and NO_X reductions from across a larger region will help to reduce fine particles and precursor pollutants in nonattainment areas as well improve visibility across the entire MANE-VU region.

The balance between regional and local controls parallels the balance that needs to be achieved between pollutants. The regional contribution to fine particle pollution is driven by sulfates and organic carbon, whereas the local contribution to $PM_{2.5}$ is derived from SO_2, NO_X, organic carbon, and primary $PM_{2.5}$ (including black carbon/diesel exhaust).

Finally, control strategies which focus on regional SO_2 emissions reductions are needed throughout the summer and winter months, suggesting that a year-round approach to control is needed. Urban nonattainment counties with local emissions of NO_X and

VOC will be driven to reduce these emissions during the summer for ozone benefits, but these same pollutants – as well as primary particulate emissions – contribute to high PM$_{2.5}$ levels in winter, suggesting that annual controls for all of these pollutants make sense in a multi-pollutant context. Finally, residential wood smoke near Class I areas is clearly a winter issue, and further controls may be desirable near specific Class I sites where organic carbon is a contributor to the worst visibility days during the winter months.

1. INTRODUCTION

1.1. Background

Fine particle pollution is a persistent public health problem in the Mid-Atlantic/Northeast Visibility Union (MANE-VU) region. Because of its physical structure, fine particulate matter (PM$_{2.5}$) can bypass conductive airways and deliver exogenous materials, such as reactive organic chemicals that adsorb onto the particle core, into the deep lung.[2] Studies of particulate matter (PM) in urban areas have found associations of short- (daily) and long-term (annual and multiyear) exposure to airborne PM as well as PM$_{2.5}$ with cardiopulmonary health outcomes. These effects include increased symptoms, hospital admissions and emergency room visits, and premature death (Pope *et al.*, 2004).

In addition to health implications, visibility impairment in the eastern United States is largely due to the presence of light-absorbing and light-scattering fine particles in the atmosphere. The United States Environmental Protection Agency (USEPA) has identified visibility impairment as the best understood of all environmental effects of air pollution (Watson, 2002). A long-established physical and chemical theory relates the interaction of particles and gases in the atmosphere with the transmission of visual information along a sight path from object to observer.

The Clean Air Act requires states that have areas designated "nonattainment" of the fine particle national ambient air quality standard (NAAQS) to submit State Implementation Plans (SIPs) demonstrating how they plan to attain the fine particle NAAQS.[3] The Clean Air Act also contains provisions for the restoration and maintenance of visibility in 156 federal Class I areas.[4] SIPs for dealing with visibility impairment (or regional haze) must include a long-term emissions management strategy aimed at reducing fine particle pollution in these rural areas.

As part of both the PM$_{2.5}$ NAAQS and visibility SIP processes, the USEPA urges states to include a conceptual description of the pollution problem. The USEPA has provided guidance on developing a conceptual description, which is contained in Chapter 11 of the document "Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, PM$_{2.5}$, and Regional Haze" (USEPA, 2007) (Appendix A of this report reproduces Chapter 11 of the USEPA

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[2] PM$_{2.5}$ or "fine particles" refer to those particles with a diameter ≤ 2.5 micrometers (µm).

[3] The 2006 PM$_{2.5}$ NAAQS includes a requirement that the three-year average of yearly annual average PM$_{2.5}$ design values must be below 15 µg/m^3 and a requirement that the three-year average of the 98th percentile 24-hour average concentration must be below 35 µg/m^3.

[4] The Class I designation applies to national parks exceeding 6,000 acres, wilderness areas and national memorial parks exceeding 5,000 acres, and all international parks that were in existence prior to 1977. In the MANE-VU area, this includes: Acadia National Park, Maine; Brigantine Wilderness (within the Edwin B. Forsythe National Wildlife Refuge), New Jersey; Great Gulf Wilderness, New Hampshire; Lye Brook Wilderness, Vermont; Moosehorn Wilderness (within the Moosehorn National Wildlife Refuge), Maine; Presidential Range – Dry River Wilderness, New Hampshire; and Roosevelt Campobello International Park, New Brunswick.

guidance document). This report provides the MANE-VU states with the basis for their conceptual descriptions, consistent with the USEPA's guidance. In the guidance, the USEPA recommends addressing 13 questions related to $PM_{2.5}$ and 8 questions related to visibility to help define the problem in a nonattainment or Class I area. This report addresses these questions, as well as provides some in-depth data and analyses that can assist states in developing conceptual descriptions tailored to their specific areas.

1.2. PM Formation

Fine particles directly emitted into the atmosphere are called "primary" fine particles, and they come from both natural and human sources and include suspended liquid and solid aerosols. These fine particles commonly include 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. Combustion sources include motor vehicles, power generation facilities, industrial facilities, residential wood burning, agricultural burning, and forest fires.

Fine particles are also comprised of "secondary" fine particles, which are formed from precursor gases reacting in the atmosphere or through the growth of pre-existing particles by absorption and adsorption. Although direct nucleation from the gas phase is a contributing factor, most secondary material accumulates on pre-existing particles in the 0.1 to 1.0 micrometer (µm) range and typically account for a significant fraction of the fine PM mass. Examples of secondary particle formation include the conversion of sulfur dioxide (SO_2) to sulfuric acid (H_2SO_4) droplets that further react with ammonia (NH_3) to form various sulfate particles (e.g., ammonium sulfate ($NH_4)_2SO_4$, ammonium bisulfate (NH_4HSO_4), and letovicite (($NH_4)_3H(SO_4)_2$). The dominant source of SO_2 emissions in the eastern U.S. is fossil fuel combustion, primarily at coal-fired power plants and industrial boilers.

Similarly, secondary $PM_{2.5}$ is created by the conversion of nitrogen dioxide (NO_2) to nitric acid (HNO_3) which reacts further with ammonia to form ammonium nitrate (NH_4NO_3) particles. Nitrate particles are formed from the nitrogen oxides (NO_X) emitted by power plants, automobiles, industrial boilers, and other combustion sources. Nitrate production in the northeastern U.S. is ammonia-limited and controlled by the availability of sulfate and temperature, especially along the East Coast.[5] While human sources account for most nitrate precursors in the atmosphere, there are some natural sources, including lightning, soil emissions, and stratospheric intrusion. Large sources of ammonia arise from major livestock production and fertilizer application throughout the Midwest, Gulf Coast, Mid-Atlantic, and southeastern United States, in addition to the sources of ammonia associated with human activities.

The carbon fraction of fine PM may refer to black carbon (BC) and primary organic and/or secondary organic carbon (OC). Most black carbon is primary, which is also sometimes referred to as elemental carbon (EC) or soot. Black carbon is light-absorbing carbonaceous material arising from the combustion of diesel, wood, and other fuels. Not all light-absorbing carbonaceous material is purely elemental carbon, and the

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[5] Ammonia reacts preferentially with sulfuric acid, and if sufficient excess ammonia is available, it can then combine with nitric acid to form particulate nitrate.

scientific literature is transitioning to the use of "light-absorbing carbon" (LAC) in place of EC when considering optical properties of carbonaceous aerosols (Bond & Bergstrom, 2006). Organic carbon includes both primary emissions and secondary organic PM in the atmosphere. Secondary organic particles are formed by reactions involving volatile organic compounds (VOCs), which yield compounds with low saturation vapor pressures that nucleate or condense on existing particles at ambient temperature. Organic carbon in both the gas and solid phase is emitted by automobiles, trucks, and industrial processes, as well as by many types of vegetation. The relative amounts of organic carbon from different sources remain highly uncertain, and data are needed to be able to assess the relative contribution of primary versus secondary and anthropogenic versus biogenic production.

1.3. PM Impacts on Visibility

Under natural atmospheric conditions, the view in the eastern United States would extend about 60 to 80 miles (100 to 130 kilometers) (Malm, 2000). Unfortunately, views of such clarity have become a rare occurrence in the East. As a result of man-made pollution, the average visual range in the eastern half of the country has diminished to about 15-30 miles, approximately one-third the visual range that would be observed under unpolluted natural conditions.

In general, the ability to see distant features in a scenic vista is determined less by the amount of light reaching the observer than by the contrast between those features and their surroundings. For example, the illumination of a light bulb in a greenhouse is barely discernible on a sunny day but would be highly visible at night. Similarly, a mountain peak is easily seen if it appears relatively dark against the sunlit sky. If, on the other hand, a milky haze "fills" the space between the observer and the mountain peak, the contrast between the mountain and its background is diminished as both take on a similar hue (Figure 1-1).

Figure 1-1. View of a good visibility day (left) and a poor visibility day (right) at Acadia National Park, Maine in June 2003.
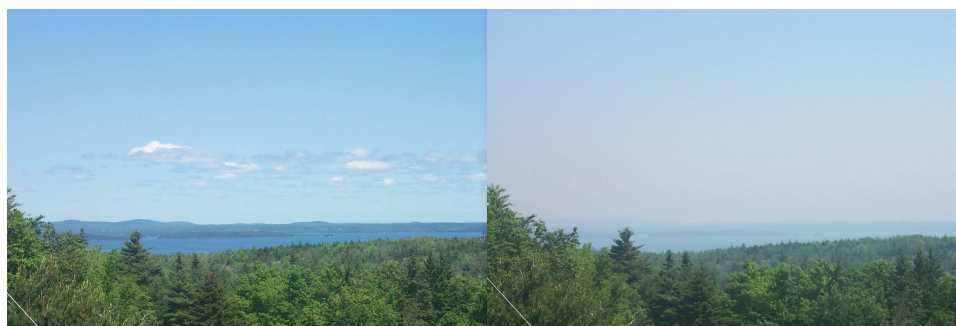
Source: CAMNET, http://www.hazecam.net

In simple terms, this hazy effect occurs when small particles in the atmosphere absorb or scatter visible light, thereby reducing the amount of visual "information" that reaches the observer. This occurs to some extent even under natural conditions, primarily as a result of the light scattering effect by naturally occurring aerosols (known as Mie

scattering), such as in the Great Smoky Mountains.[6] The substantial visibility impairment caused by manmade pollution is almost entirely attributable to the increased presence of fine particles in the atmosphere.[7]

Figure 1-2 presents a simplified schematic of the way such small particles interact with packets of light or "photons" as they travel from a distant object to an observer. Along the way, particles suspended in the air can deflect or scatter some of the photons out of the sight path. Intervening particles can also absorb photons, similarly removing them from the total amount of light reaching the observer.

Figure 1-2. Schematic of visibility impairment due to light scattering and absorption (adapted from Malm, 2000).

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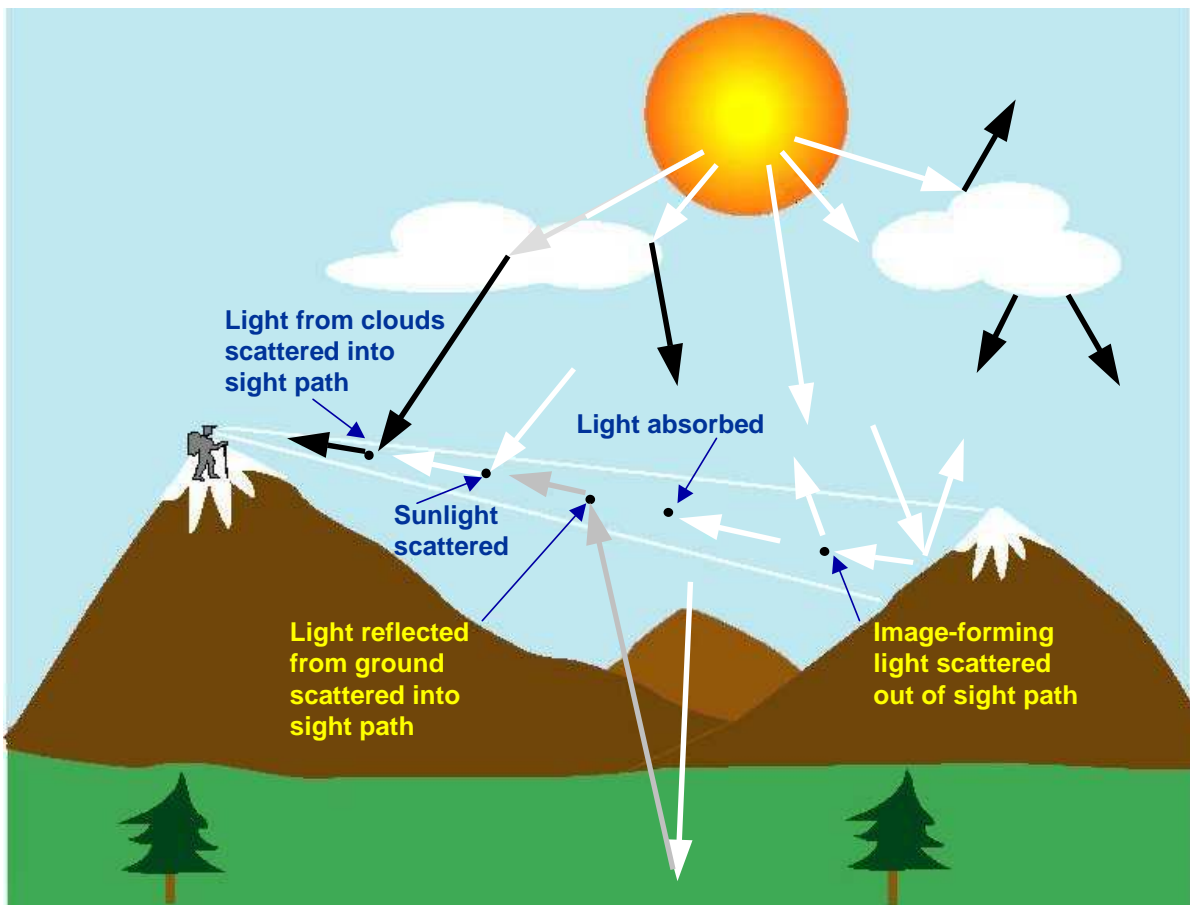
[6] Atmospheric aerosol is a more general term for fine particles suspended in the atmosphere and refers to any particle (solid or liquid) that is suspended in the atmosphere.
[7] The only light-absorbing gaseous pollutant present in the atmosphere at significant concentrations is nitrogen dioxide (NO₂). However, the contribution of NO₂ to overall visibility impacts in the MANE-VU region is negligible and hence its effects are not generally included in this discussion or in standard calculations of visibility impairment.

At the same time, particles in the air can scatter light into the sight path, further diminishing the quality of the view. The extraneous light can include direct sunlight and light reflected off the ground or from clouds. Because it is not coming directly from the scenic element, this light contains no visual information about that element. When the combination of light absorption and light scattering (both into and out of the sight path) occurs in many directions due to the ubiquitous presence of small particles in the atmosphere, the result is commonly described as "haze."

1.4. PM$_{2.5}$ Design Values in the MANE-VU Region

SIP developers use monitoring data in several important ways to support SIP activities. This section as well as Section 1.5 present measurements from the Federal Reference Method (FRM) and Interagency Monitoring of Protected Visual Environments (IMPROVE) monitoring networks needed in establishing SIP requirements. Following USEPA guidance (40CFR Part 50, Appendix N; USEPA, 2003a; USEPA, 2003b), we use these data to preview the design values and baseline conditions that SIP developers must consider for each nonattainment area and Class I area.

The current daily standard was revised in 2006 from 65 μg/m^3 to 35 μg/m^3 at the 98th percentile level. To meet this standard, the 98th percentile value (of valid measurements recorded at a site) must not be greater than this level. The USEPA designated areas in nonattainment of the revised 24-hour standard in October 2009. Table 1-1 lists nonattainment areas for the 24-hour NAAQS in the MANE-VU region, which includes portions of western and eastern Pennsylvania, northern Delaware, central New Jersey, downstate New York, including Long Island, and southwestern Connecticut. These areas will have to comply with the new standard by December 2014, with the possibility of an extension up to 2019. Fine particle data from the USEPA's Air Quality System (AQS) database for years 2006 through 2008 were used to determine the attainment status of monitoring sites in MANE-VU.

The current annual fine particle NAAQS was established in 1997 at 15 μg/m^3, which was retained in 2006 when the 24-hour PM$_{2.5}$ NAAQS was tightened.[8] To meet this standard, the 3-year average of a site's annual mean concentration must not be greater than this level. Table 1-1 shows a summary of areas in nonattainment of the annual standard as designated in 2004 based on air quality monitoring data during 2001-2003 from the USEPA's AQS database. As tabulated, 12 areas failed to achieve the annual standard, with design values ranging from 15.1 to 20.4 μg/m^3. The nonattainment areas were concentrated in Pennsylvania and the coastal urban corridor. Sulfates and organic carbon represent the largest contributors to these high fine particle levels. Since being designated nonattainment for the annual NAAQS in 2004, 8 of the original 12

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[8] The USEPA decided not to revise the annual PM$_{2.5}$ NAAQS of 15 μg/m^3 at the time it revised the 24-hour NAAQS, a decision later remanded back to the USEPA by the Court of Appeals for the District of Columbia in early 2009. The annual PM$_{2.5}$ NAAQS, however, remained in place while the USEPA undertook a 5-year review of the standards and the latest scientific information, as required by the Clean Air Act. The statutory deadline for completion of the USEPA's 5-year review was October 17, 2011, which the agency missed. As the result of a federal court order, the USEPA agreed to propose revised standards by June 14, 2012, and issue final standards by December 14, 2012.

nonattainment areas recorded annual $PM_{2.5}$ levels in 2006-2008 that met the annual $PM_{2.5}$ NAAQS.

Table 1-1. 2001-03 Annual and 2006-08 24-hour $PM_{2.5}$ Design Values for Nonattainment Areas in MANE-VU

State(s)	Nonattainment Area	2001-03 Annual Design Value	2006-08 24-hr Design Value
PA	Allentown	--	36
MD	Baltimore	16.3*	--
PA	Harrisburg-Lebanon-Carlisle	15.4*	36
PA	Johnstown	15.3*	30**
PA	Lancaster	16.8*	37
PA	Liberty/Clairton	20.4	53
MD	Martinsburg, WV-Hagerstown	16.1*	--
NY-NJ-CT	New York-N. New Jersey-Long Island	16.8	38
PA-NJ-DE	Philadelphia-Wilmington	15.4	36
PA	Pittsburgh-Beaver Valley	16.5	36
PA	Reading	16.1*	--
DC-MD-VA	Washington, DC	15.1*	--
PA	York	16.9*	--

* 2006-2008 annual design value met the 15 µg/m³ annual NAAQS.
**Based on 2005-07 data due to incomplete data in 2008.

1.5. Regional haze baseline conditions

The Regional Haze Rule requires states and tribes to submit plans that include calculations of current and estimated baseline and natural visibility conditions. They will use monitoring data from the IMPROVE program as the basis for these calculations. Table 1-2 and Table 1-3 present the five-year average[9] of the 20 percent worst day mass concentrations and 20 percent best day mass concentrations respectively in six Class I areas. Five of these areas are in MANE-VU and one (Shenandoah) is nearby but located in a neighboring regional planning organization (RPO) region.[10] Table 1-4 and Table 1-5 give particle contributions to light extinction for the six Class I areas for the 20 percent worst and best days. Each of these tables show the relative percent contribution for all six Class I sites. Sulfate and organic carbon dominate the fine mass, with sulfate even more important to light extinction.

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[9] Great Gulf calculations are based on four years of data (2001-2004).

[10] Note that values presented for Shenandoah, a Class I area in the Visibility Improvement State and Tribal Association of the Southeast (VISTAS) region, are for comparative purposes only. VISTAS will determine uniform rates of progress for areas within its region.

To guide the states in calculating baseline values of reconstructed extinction and in estimating natural visibility conditions, the USEPA released two documents in the fall of 2003 outlining recommended procedures (USEPA 2003a; USEPA 2003b). The IMPROVE Steering Committee has endorsed an alternative method for the calculation of these values. The IMPROVE alternative method was used to create Table 1-6, which provides detail on the uniform visibility goals for the 20 percent worst conditions at the six Class I areas.

The first column of data in Table 1-6 gives the alternative proposed natural background levels for the worst visibility days at the six sites. MANE-VU decided to use this approach, at least initially, for 2008 SIP planning purposes (NESCAUM, 2006). The second column shows the baseline visibility conditions on the 20 percent worst visibility days. These values are based on IMPROVE data from the official five-year baseline period (2000-2004) and again were calculated using the IMPROVE alternative approach. Using these baseline and natural background estimates, we derive the uniform rate of progress shown in the third column.[11] The final column displays the interim 2018 progress goal based on 14 years of improvement at the uniform rate.

Table 1-2. Fine mass and percent contribution for 20 percent worst days

20% Worst-day Fine Mass (μg/m^3)/ % contribution to fine mass					
Site	SO$_4$	NO$_3$	OC	EC	Soil
Acadia	6.3/ 56%	0.8/ 7%	3.2/ 28%	0.4/ 4%	0.5/ 5%
Brigantine	11.6/ 56%	1.7/ 8%	5.8/ 28%	0.7/ 3%	1/ 5%
Great Gulf	7.3/ 59%	0.4/ 3%	3.8/ 31%	0.4/ 3%	0.6/ 5%
Lye Brook	8.5/ 58%	1.1/ 7%	3.9/ 27%	0.5/ 3%	0.6/ 4%
Moosehorn	5.7/ 54%	0.7/ 7%	3.4/ 32%	0.4/ 4%	0.4/ 4%
Shenandoah	13.2/ 68%	0.7/ 3%	4.2/ 22%	0.6/ 3%	0.7/ 4%

Table 1-3. Fine mass and percent contribution for 20 percent best days

20% Best-day Fine Mass (μg/m^3)/ % contribution to fine mass					
Site	SO$_4$	NO$_3$	OC	EC	Soil
Acadia	0.8/ 42%	0.1/ 6%	0.8/ 41%	0.1/ 5%	0.1/ 6%
Brigantine	1.8/ 43%	0.5/ 11%	1.5/ 35%	0.2/ 6%	0.2/ 5%
Great Gulf	0.7/ 43%	0.1/ 7%	0.7/ 40%	0.1/ 5%	0.1/ 6%
Lye Brook	0.6/ 44%	0.1/ 11%	0.4/ 33%	0.1/ 5%	0.1/ 7%
Moosehorn	0.8/ 37%	0.1/ 6%	1/ 47%	0.1/ 5%	0.1/ 5%
Shenandoah	1.4/ 45%	0.5/ 16%	1/ 29%	0.2/ 5%	0.2/ 5%

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[11] We calculate the rate of progress as (baseline – natural background)/60 to yield the annual deciview (dv) improvement needed to reach natural background conditions in 2064, starting from the 2004 baseline.

Table 1-4. Light extinction and % particle contribution for 20 percent worst days

20% Worst-day light extinction (Mm^{-1})/ % Particle contribution to light extinction						
Site	SO$_4$	NO$_3$	OC	EC	Soil	CM
Acadia	69.2/ 64%	8/ 7%	11.2/ 10%	4.3/ 4%	0.5/ 0%	1.9/ 2%
Brigantine	127.1/ 66%	15.7/ 8%	24.2/ 13%	7/ 4%	1/ 1%	5.4/ 3%
Great Gulf	76.6/ 68%	3/ 3%	14.4/ 13%	3.9/ 3%	0.6/ 1%	3/ 3%
Lye Brook	87.3/ 67%	9.1/ 7%	15.3/ 12%	4.8/ 4%	0.6/ 0%	1.8/ 2%
Moosehorn	58.5/ 60%	6.4/ 7%	11.9/ 12%	4.4/ 5%	0.4/ 0%	2.1/ 3%
Shenandoah	155.5/ 79%	5.8/ 3%	16.1/ 8%	5.7/ 3%	0.7/ 0%	2.5/ 1%

Table 1-5. Light extinction and % particle contribution for 20 percent best days

20% Best-day light extinction (Mm^{-1})/ % Particle contribution to light extinction						
Site	SO$_4$	NO$_3$	OC	EC	Soil	CM
Acadia	6.8/ 28%	1.1/ 4%	2.2/ 9%	0.9/ 4%	0.1/ 0%	0.7/ 6%
Brigantine	14.8/ 35%	3.9/ 9%	4.5/ 11%	2.4/ 6%	0.2/ 1%	3.2/ 11%
Great Gulf	5.8/ 27%	1/ 4%	2/ 9%	0.8/ 4%	0.1/ 0%	0.9/ 8%
Lye Brook	4.4/ 23%	1.2/ 6%	1.3/ 7%	0.6/ 3%	0.1/ 0%	0.5/ 6%
Moosehorn	6.7/ 26%	1.1/ 4%	3.1/ 12%	1/ 4%	0.1/ 0%	1.1/ 8%
Shenandoah	11.2/ 36%	4.2/ 13%	2.9/ 9%	1.6/ 5%	0.2/ 1%	1.1/ 5%

Table 1-6. Natural background and baseline calculations for select Class I areas

Site	20% Worst Days Natural Background (dv)	20% Worst Days Baseline 2000-04(dv)	Uniform Rate (dv/yr)	Interim Progress Goal 2018 (dv)	20% Best Days Baseline 2000-04(dv)
Acadia	12.54	22.89	0.17	20.47	8.77
Brigantine	12.34	29.01	0.28	25.12	14.33
Great Gulf	12.12	22.82	0.18	20.32	7.66
Lye Brook	11.85	24.44	0.21	21.50	6.37
Moosehorn	12.10	21.72	0.16	19.48	9.15
Dolly Sods	10.45	29.05	0.31	24.71	12.28
James River Face	11.20	29.12	0.30	24.94	14.21
Shenandoah	11.44	29.31	0.30	25.14	10.92

As demonstrated in Table 1-2, the inorganic constituents of fine particles (sulfates and nitrates) are the dominant contributors to visibility impairment, accounting for about 80 percent of total light extinction. Within the MANE-VU sites, the relative split between these two components is ~8 to 1 sulfate to nitrate (at Shenandoah, the average 20 percent worst day contribution of sulfates is even more dominant). Carbonaceous components account for the bulk of the remaining light extinction, ranging from 12 to nearly 20 percent, mostly in the form of organic carbon. The remaining components add little to the extinction budget on the worst days, with a few percent attributable to coarse mass and around a half percent from fine soil.

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2. A DETAILED LOOK AT FINE PARTICLE POLLUTION AND REGIONAL HAZE IN THE MANE-VU REGION

Developing a conceptual description of fine particle pollution or regional haze requires combining experience and atmospheric-science expertise with multiple data sources and analysis techniques. This includes measured data on ambient pollutant concentrations as well as emission inventory and meteorological data, chemical transport modeling, and observationally based models (NARSTO, 2003). Here, we begin with a conceptual description based on the existing scientific literature and regional data analyses concerning PM$_{2.5}$ and its effect on visibility. This includes numerous review articles and reports on the subject. Subsequent chapters review monitoring data, emissions inventory information, and modeling results to support the conceptual understanding of regional fine particle pollution presented here.

Most past assessments of fine particle pollution and visibility impairment have tended to be national in scope. For purposes of this discussion, we have selectively reviewed the literature in order to present a distinctly eastern U.S. focus. While we already know much about fine particle pollution and visibility impairment and their causes in the MANE-VU region (see NESCAUM, 2001, 2006; NARSTO, 2003; Watson, 2002), significant gaps in understanding remain with respect to the nitrate and organic component of PM$_{2.5}$. While research continues, we have assembled the relevant information that is available to provide an overview of our current understanding of the regional context for PM$_{2.5}$ nonattainment and visibility impairment in the MANE-VU region.

2.1. Chemical composition of particulate matter in the rural MANE-VU region

Sulfate alone accounts for anywhere from one-half to two-thirds of total fine particle mass on high PM$_{2.5}$ days in rural areas of MANE-VU. Even on low PM$_{2.5}$ days, sulfate generally accounts for a major fraction of total fine particle mass in the MANE-VU region (NESCAUM, 2001, 2004) as well as across the eastern United States (NARSTO, 2003).

After sulfate, organic carbon (OC) consistently accounts for the next largest fraction of total fine particle mass. Based on measurements at IMPROVE sites, its contribution typically ranges from 20 to 30 percent of total fine particle mass on the days with the highest levels of PM$_{2.5}$. Measurements at two widely separated rural locations in New York State (western and northern ends) over a three-year period found total carbon contributions (organic and elemental) to the measured fine particle mass of about 30 percent (Sunder Raman *et al.*, 2008). Aircraft measurements during a few summer days in 2002 over southern New England and the Mid-Atlantic observed a higher fraction of total mass from organic carbon, varying from 70 percent in clean air to 40 percent in high concentration sulfate plumes (Kleinman *et al.*, 2007). The fact that the contribution from organic carbon is higher on low PM$_{2.5}$ days is likely indicative of the role played by organic emissions from vegetation ("biogenic hydrocarbons"). Furthermore, there are

also indications that secondary organic aerosol formation from biogenic hydrocarbons may be enhanced in the presence of acidic sulfate seed aerosol (Surratt *et al*., 2007).

Relative contributions to overall fine particle mass from nitrate (NO_3), elemental carbon, and fine soil are all smaller (typically under 10 percent), but the relative ordering among the three species varies with location and season. Figure 2-1 below, reflects the difference between nitrate and organic contributions to rural fine particle concentrations during different seasons (monitoring data for additional sites in the MANE-VU region are in Appendix B) and over two different annual time periods.

Almost all particle sulfate originates from sulfur dioxide (SO_2) oxidation and typically associates with ammonium (NH_4) in the form of ammonium sulfate (($NH_4)_2SO_4$). Ninety-five percent of SO_2 emissions are from anthropogenic sources (primarily from fossil fuel combustion), while the majority of ammonium comes from agricultural activities and, to a lesser extent, from transportation sources in some areas (NARSTO, 2003).

Two major chemical pathways produce sulfate from SO_2 in the atmosphere. In the gas phase, production of sulfate involves the oxidation of SO_2 to sulfuric acid (H_2SO_4), ammonium bisulfate (NH_4HSO_4), or ammonium sulfate, depending on the availability of ammonia (NH_3). In the presence of small wet particles (typically much, much smaller than rain drops or even fog), a highly efficient aqueous phase process can oxidize SO_2 to sulfate extremely quickly (~10 percent per hour).

Figure 2-1. Comparison of contributions during different seasons at Lye Brook Wilderness Area on 20% worst visibility (high PM$_{2.5}$) days for 2000-2004 (upper) and 2004-2007 (lower)
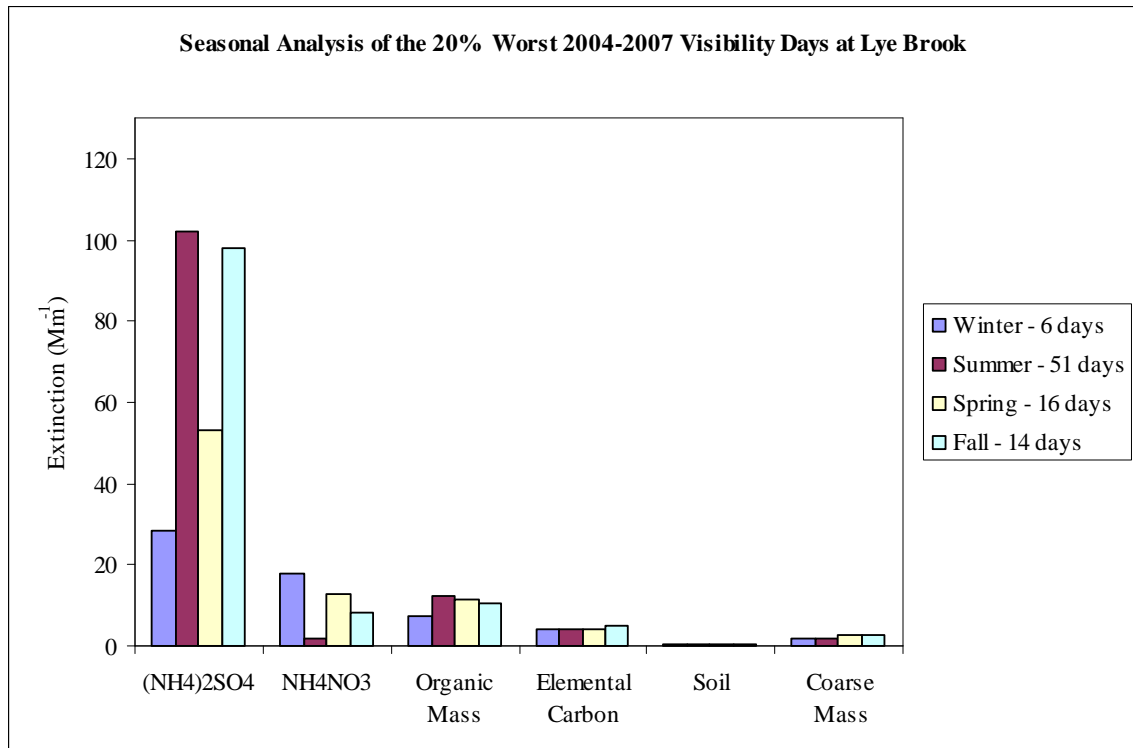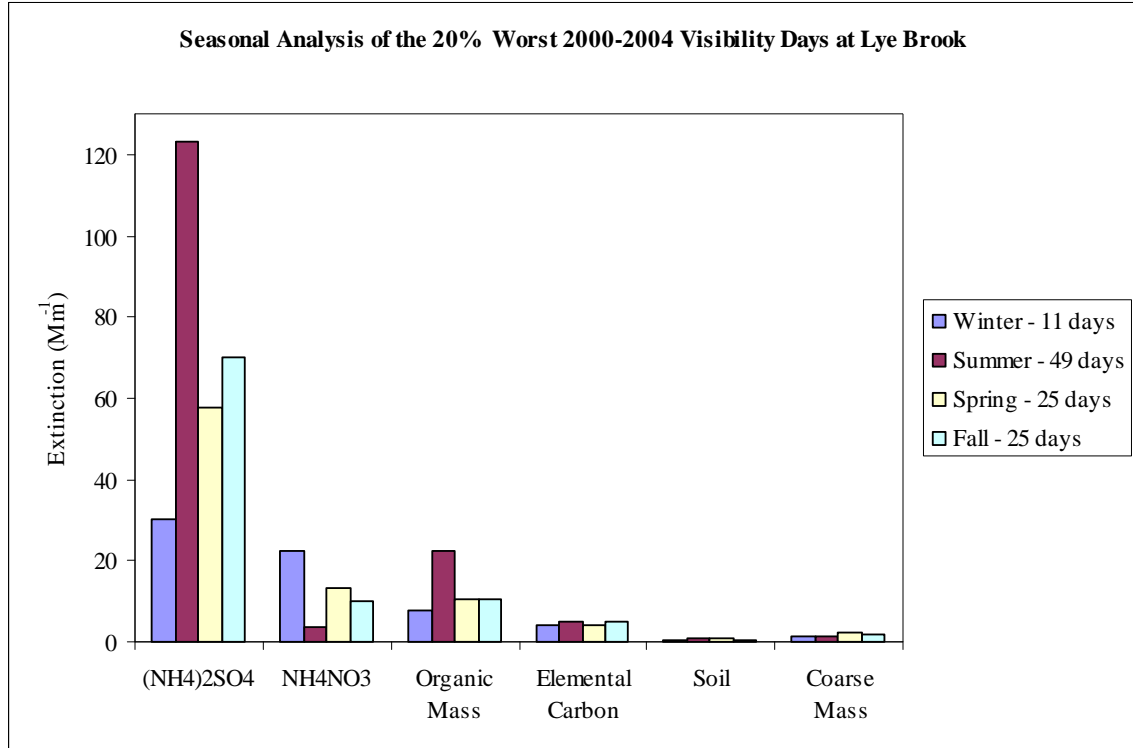
Not only is sulfate the dominant contributor to fine particle mass in the region, it accounts for anywhere from 60 percent to almost 80 percent of the *difference* between fine particle concentrations and extinction on the lowest and highest mass days at rural locations in the northeast and mid-Atlantic states (See Figure 2-2). Notably, at urban locations such as Washington, DC, sulfate accounts for only about 40 percent of the difference in average fine particle concentrations for the 20 percent most versus least visibility impaired days (NESCAUM, 2001).

**Figure 2-2. Comparison of species contributions on best and worst days
at Lye Brook Wilderness Area**
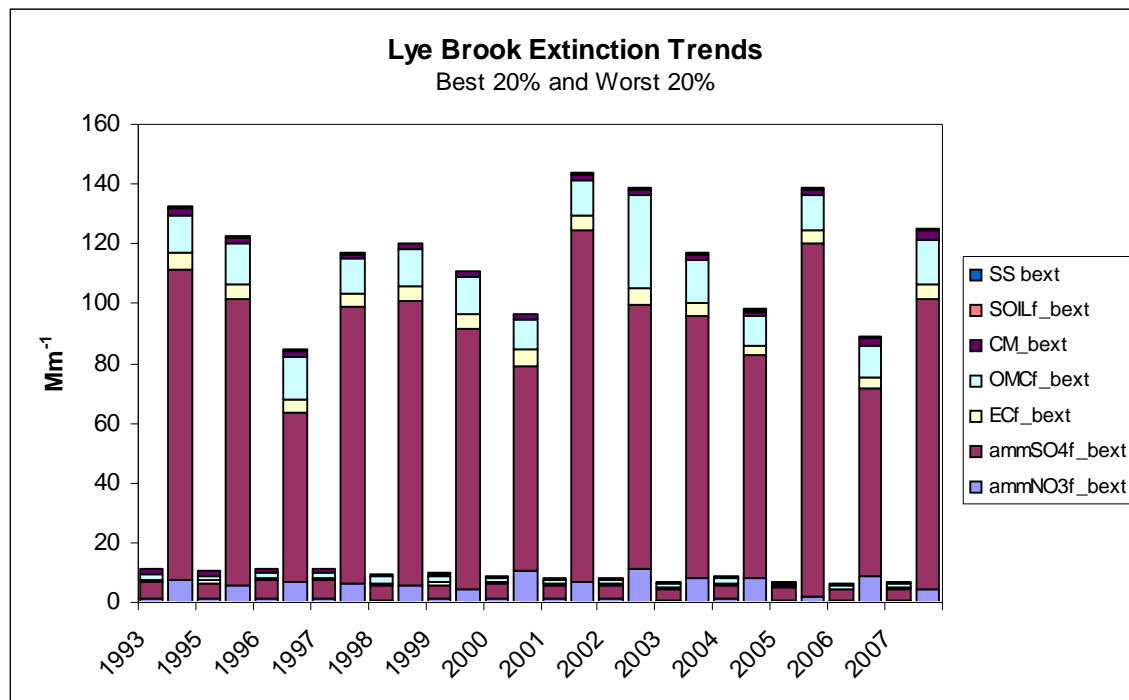
2.2. Rural versus urban chemistry

Contributions to fine particle mass concentrations at rural locations include long-range pollutant transport as well as non-anthropogenic background contributions. At a rural site in south-central Ohio, secondary sulfate, secondary organic and nitrate varied according to the season, with the sulfate and secondary organic peaking in the warm months and the nitrate peaking in the cold months. The high percentage of secondary sulfate observed at the rural Ohio site suggested regional transport (Kim *et al*., 2007).

Urban areas generally show mean PM$_{2.5}$ levels exceeding those at nearby rural sites. In the Northeast, this difference implies that local urban contributions are roughly 25 percent of the annual mean urban concentrations, with regional aerosol contributing the remaining, and larger, portion (NARSTO, 2003). Monitoring data show that light absorbing carbon has the greatest urban excess over rural for PM$_{2.5}$ components in the eastern United States. Light absorbing carbon also has the sharpest spatial gradients between urban and rural areas, indicating the local influence of urban emissions sources, such as diesel vehicles (Hand *et al.*, 2011). Urban PM$_{2.5}$ levels can also vary spatially at

the urban scale (~5–50 km) to a greater extent than broader regional PM$_{2.5}$ levels (~50–1000 km). A review by Turner & Allen (2008) summarizes intraurban PM$_{2.5}$ studies finding, in some urban areas, greater heterogeneity in PM$_{2.5}$ levels than in more rural regions. In addition, short range pollution transport exists with primary and precursor particle pollutants pushed by land, sea, mountain, and valley breezes that can selectively affect relatively local areas. The Chesapeake Bay breeze is one example within the MANE-VU region (Loughner, *et al.*, 2011).

This rural versus urban difference in typical concentrations also emerges in a source apportionment analysis of fine particle pollution in Philadelphia (see Chapter 10 of NARSTO, 2003) using two different mathematical models, UNMIX and Positive Matrix Factorization (PMF). This analysis provides additional insight concerning sources of fine particle pollution in urban areas of the densely populated coastal corridor between Washington, DC and New England. Specifically, this analysis found the following apportionment of PM$_{2.5}$ mass in the study area:

- Local SO$_2$ and sulfate: ~ 10 percent
- Regional sulfate: ~ 50 percent
- Residual oil: 4-8 percent
- Soil: 6-7 percent
- Motor vehicles: 25-30 percent

The analysis does not account for biogenic sources, which most likely are embedded in the motor vehicle fraction (NARSTO, 2003). The Philadelphia study suggests that both local pollution from nearby sources and transported "regional" pollution from distant sources contribute to the high sulfate concentrations observed in urban locations along the East Coast on an annual average basis. Summertime sulfate and organic carbon are strongly regional in eastern North America. Typically 75–95 percent of the urban sulfate concentrations and 60–75 percent of the urban OC concentrations arise from cumulative region-wide contributions (NARSTO, 2003). Urban air pollutants are essentially added on top of this regional background. Nitrate plays a noticeably more important role at urban sites compared to northeastern and mid-Atlantic rural monitoring sites, perhaps reflecting a greater contribution from vehicles and other urban pollution sources (NESCAUM, 2001). In Midwest urban areas, nitrates were the driving anthropogenic component of observed wintertime PM$_{2.5}$ exceedances observed at more northern latitudes compared to other constituents, including sulfates (Katzman *et al.*, 2010).

It is difficult to discern any significant meaning about the cause of "excess" mass from a single pair of sites. There are many factors that influence the concentrations at a particular site and it is likely that for every pair of sites that shows an urban excess, one could find some pair of locations that might show something similar to an urban "deficit." While paired sites from an urban and a rural location will *typically* show greater concentrations in the urban location and lower levels of pollution in rural areas, great care must be exercised in the interpretation of any two-site analysis such as the comparisons of speciated components of PM$_{2.5}$ presented here. Nonetheless, such comparisons do provide a general feel for the typical chemical composition of PM$_{2.5}$ in the eastern U.S. and the relative differences in chemical composition between rural and

more urban locations. More detailed, "network"-wide analyses (*e.g., see* NESCAUM 2004; relevant sections are attached in Appendix C to this report) indicate that the results provided are not anomalous of typical urban environments in the MANE-VU region.

Figure 2-3 and Figure 2-4 compare two urban-rural pairs of speciation monitors: the New York nonattainment area (Elizabeth and Chester, New Jersey) and the Boston metropolitan area (Boston and Quabbin Reservoir, Massachusetts). The first three sites are Speciation Trends locations, while the Reservoir site is part of the IMPROVE protocol network.[12]

Figure 2-3. New York nonattainment area (Elizabeth, NJ) compared to an upwind background site (Chester, NJ)

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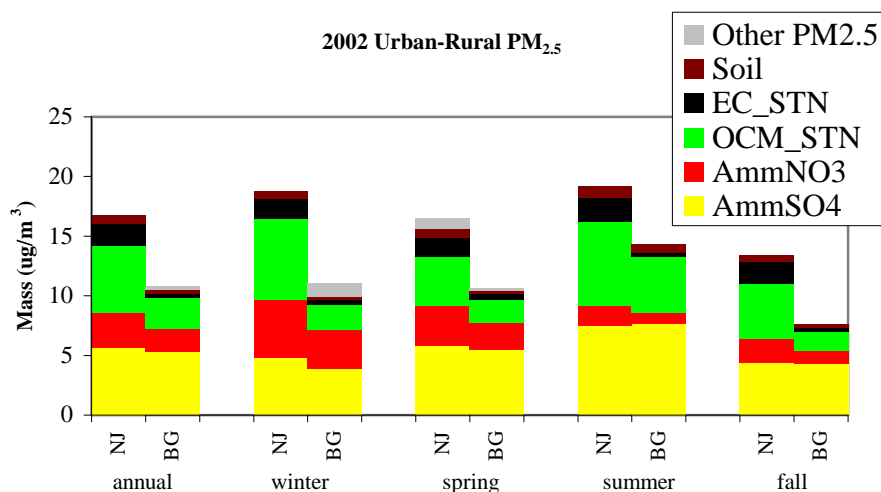
[12] To provide a more direct comparison of the differences between the urban and rural sites, only those days for which both monitors in a pair had data were used. Four seasonal averages were computed for 2002, with seasons defined as winter (January, February, December), spring (March, April, May), summer (June, July, August) and fall (September, October, November). July 7 was excluded from the analysis because the Quebec forest fires affecting the region on that day would have dominated the summertime averages. The major fine particle species categories considered included ammonium sulfate, ammonium nitrate, organic carbon, elemental carbon, and soil mass. The traditional assumptions about these constituents were made; all sulfate was fully neutralized and a multiplier of 1.4 was used to account for mass of organic carbon. An "other PM$_{2.5}$ mass" category was created to delineate the difference between gravimetric mass determined from the Teflon filter and the reconstructed mass sum of the individual mass constituents. Where no "other" mass is graphed, the sum of the species either equaled or exceeded the directly measured mass. No adjustments were made to account for the different operational definitions of carbon between the IMPROVE network and the USEPA's Speciated Trends Network (STN). Average blank corrections were applied to all samples. In the case of New York City, both rural and urban monitors were STN. The Boston pair reflects not only inter-site differences, but also differences in definition of organic and elemental carbon. However, the general interpretation of the data differences remains consistent. Based on current understanding, the rural elemental carbon would be even lower than what is shown on the graph if it were made consistent with the STN definition of EC. Likewise, the organic carbon value would increase slightly for the rural value, as the EC would be allocated to OC. The urban OC levels are so much greater than those in the rural area that a slight increase in rural OC makes little difference.

**Figure 2-4. Boston urban area (Boston, MA) compared
to an upwind background site (Quabbin Reservoir, MA)**
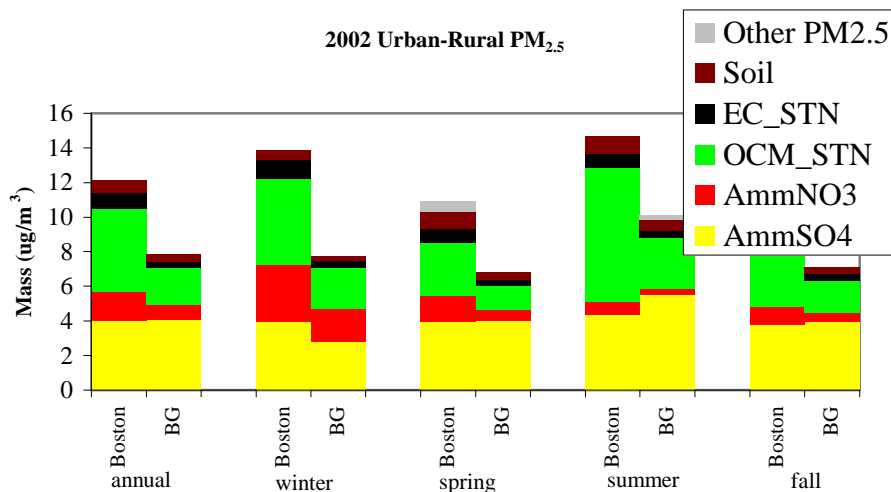
The urban-rural differences show consistency for both the New York City nonattainment area and Boston. On an annual scale, the sulfate levels are comparable, with increased mass loading at these urban sites driven primarily by differences in nitrates and carbon with smaller differences in "soil" levels. One interesting aspect of this comparison is the seasonal differences in the urban-rural sulfate split. On an annual basis, sulfate appears to be similar at urban and rural locations (based on these two pair of sites); however, during the colder months, the urban sulfate levels are elevated relative to the rural levels. This behavior is opposite during the summer. During the wintertime, the Northeast urban corridor itself is a substantial source of sulfur emissions. These local emissions can be trapped near the surface during the winter and have a corresponding higher impact on the urban area relative to the rural area.

For both urban and rural areas, the summertime OC levels are significantly greater than wintertime concentrations. Although the oxidation chemistry slows in winter, the cooler temperatures change the phase dynamics, driving more mass into the condensed over the gas phase. This along with more frequent temperature inversions (which limit atmospheric ventilation of the urban boundary layer) can lead to the observed increases in the relative influence of both organic and nitrate levels during winter months. EC, OC, and nitrate all are observed to have higher measured levels in the urban area (but still lower than the comparable summer values measured at the same sites), driven by local sources of these constituents.

2.3. Geographic considerations and attribution of PM$_{2.5}$/haze contributors

In the East, both annual average and maximum daily fine particle concentrations are highest near heavily industrialized areas and population centers. Not surprisingly, given the direct connection between fine particle pollution and haze, the same pattern emerges when one compares measures of light extinction on the most and least visibility

impaired days at parks and wilderness areas subject to federal haze regulations in the MANE-VU region (NESCAUM, 2001). An accumulation of particle pollution often results in hazy conditions extending over thousands of square kilometers (km^2) (NARSTO, 2003). Substantial visibility impairment is a frequent occurrence in even the most remote and pristine areas of the MANE-VU region (NESCAUM, 2001).

$PM_{2.5}$ mass declines fairly steadily along a southwest to northeast transect of the MANE-VU region. This decline is consistent with the existence of large fine particle emissions sources (both primary and secondary) to the south and west of MANE-VU. This trend is driven, in large part, by the marked southwest-to-northeast gradient in ambient sulfate concentrations during three seasons of the year as illustrated in Figure 2-5 based on data from IMPROVE and the USEPA's Speciation Trends Network (STN).[13] Wintertime concentrations, by contrast, are far more uniform across the entire region. Figure 2-6 shows that on an annual basis, both total $PM_{2.5}$ and sulfate mass are highest in the southwestern portions of the MANE-VU region (note the different scales for each pollutant). High concentrations of nitrate and organic particle constituents, which play a role in localized wintertime $PM_{2.5}$ episodes, tend to be clustered along the northeastern urban corridor and in other large urban centers.

While these figures provide some preliminary context for identifying sources contributing to the region's particulate matter and visibility problems, they say nothing about the relative efficiency of a state's or region's emissions in contributing to the problem. It is clear that distance from the emissions source matters. Local, nearby sources are exceedingly important and sources within about 200 km are much more efficient (on a per ton emitted basis) at producing pollution impacts at eastern Class I sites such as Shenandoah National Park than emissions sources farther away (USNPS, 2003). At a rural site in southwestern New York State (Pinnacle State Park), Bae *et al.* (2011) found that most total $PM_{2.5}$ high pollution episodes likely arose from sources located several hundred kilometers from the monitoring site. Measurements of elemental (black) carbon at a relatively remote location on Whiteface Mountain in northeastern New York State found a high degree of correlation between black carbon and sulfate, $PM_{2.5}$ mass, ozone, and other pollutants during high black carbon episodes ($r^2 \geq 0.074$). This suggested long distance transport over several days to the site because of the time needed for secondary pollutants (e.g., sulfate) to form and generate such high correlations with the primary black carbon pollution (Dutkiewicz *et al.*, 2011).

In general, the "reach" of sulfate air pollution resulting from SO_2 emissions is longest (650–950 km). The reach of ammonia emissions or reduced nitrogen relative to nutrient deposition is the shortest (around 400 km), while oxides of nitrogen and sulfur — in terms of their impacts with respect to acidic deposition – have a reach between 550–650 km and 600–700 km, respectively (USNPS, 2003). A review by Allen & Turner (2008) summarizing findings of air monitoring field programs indicated that aerosol transport occurred over distances of 100-1000 km. Transport has a significant influence on urban concentrations in a number of cities, including New York, NY, Baltimore, MD, and Pittsburgh, PA, and is a dominant factor in rural particulate matter levels. Wagstrom and Pandis (2011) found that approximately 50 percent of elemental carbon at major

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[13] The STN is now the Chemical Speciation Network (CSN).

urban areas in the eastern United States arose from local sources, with 80 percent coming from sources within 200 kilometers. Elemental carbon at a rural location in the Great Smoky Mountains arose from sources 100 to 550 kilometers away. Sulfate and secondary organic aerosol sources were more regional in nature, coming from sources greater than 200 kilometers away from a receptor, with longer transport distances in the winter in the northeastern United States.

Monitoring evidence indicates that non-urban visibility impairment in eastern North America is predominantly due to sulfate particles, with organic particles generally second in importance (NARSTO, 2003). This makes sense, given the "long reach" of SO$_2$ emissions once they are chemically transformed into sulfate and given the ubiquitous nature of OC sources in the East. The poorest visibility conditions occur in highly industrialized areas encompassing and adjacent to the Ohio River and Tennessee Valleys. These areas feature large coal-burning power stations, steel mills, and other large emissions sources. Average fine particle concentrations and visibility conditions are also poor in the highly populated and industrialized mid-Atlantic seaboard but improve gradually northeast of New York City (Watson, 2002).

A review of source apportionment and ensemble trajectory analyses conducted by USEPA (2003) found that all back trajectory analyses for eastern sites associated sulfate with the Ohio River Valley area. Six-hour back trajectories from Whiteface Mountain in northeastern New York State also associated high sulfate trajectories with the industrialized Midwestern United States (Khan *et al.*, 2010). A source apportionment analysis of fine particles at two widely separated rural locations in New York State (western and northern ends) also identified the Ohio River Valley region as a common potential source region for secondary sulfate at the sites (Sunder Raman & Hopke, 2007). These studies also are frequently able to associate other types of industrial pollutants (e.g., copper or zinc smelting, steel production, etc.) with known source areas, lending credibility to their performance. Several studies in the USEPA review noted transport across the Canadian border, specifically sulfates from the Midwest United States into Canada, and smelter emissions from Canada into the northeastern United States.

A recent, comprehensive analysis of air quality problems at Shenandoah National Park conducted by the U.S. National Park Service (USNPS, 2003) focused on contributions to particulate pollution and visibility impairment south of the MANE-VU region. In descending order of importance, the Park Service analysis determined that Ohio, Virginia, West Virginia, Pennsylvania, and Kentucky comprise the top five of 13 key states contributing to ambient sulfate concentrations and haze impacts at the park. West Virginia, Ohio, Virginia, Pennsylvania, and Kentucky comprise the top five contributing states with respect to sulfur deposition impacts at the park. Finally, Virginia, West Virginia, Ohio, Pennsylvania, and North Carolina were found to be the top five states contributing to deposition impacts from oxidized nitrogen at the park (USNPS, 2003).

Figure 2-5. 2002 Seasonal average SO$_4$ based on IMPROVE and STN data
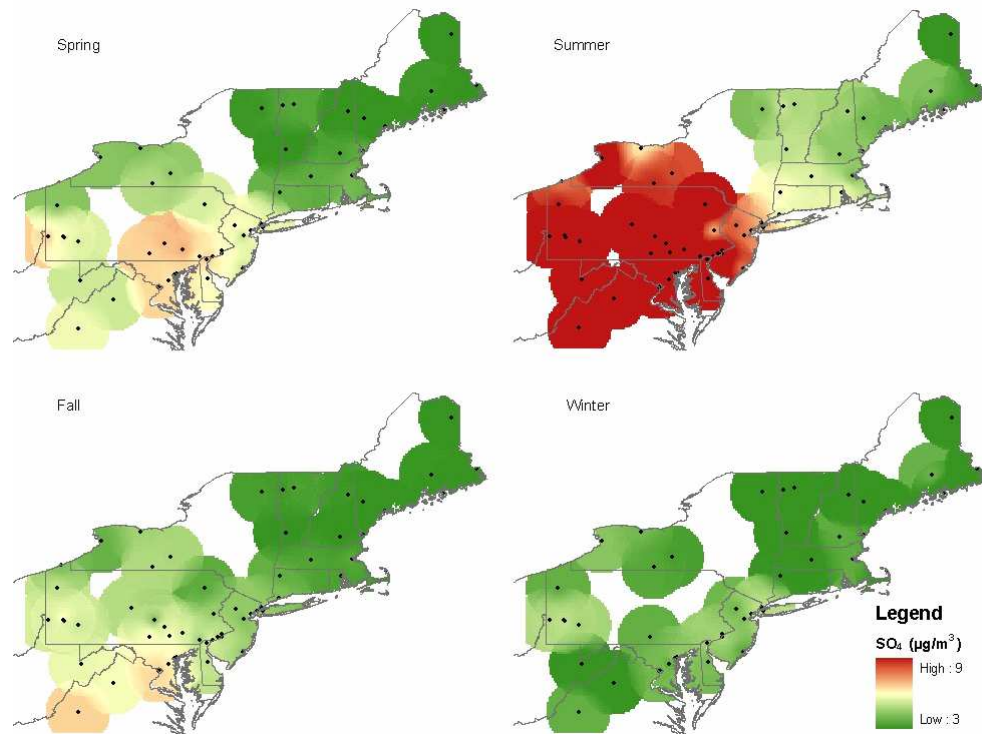
Figure 2-6. 2002 Annual average PM$_{2.5}$, sulfate, nitrate and total carbon for MANE-VU based on IMPROVE (I) and STN (S) data. PM$_{2.5}$ mass data are supplemented by measurements from the FRM network (•).
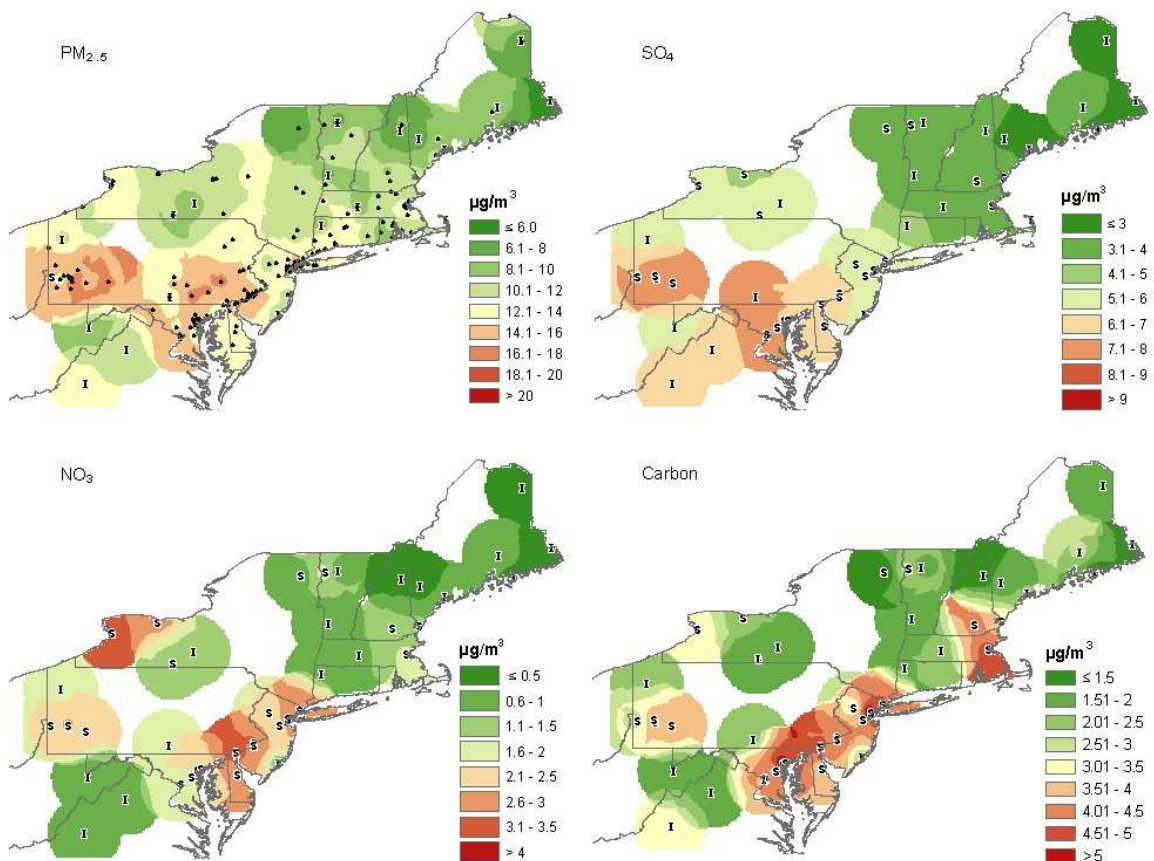
In sum, the Park Service found that emission sources located within a 200 km (125 mile) radius of Shenandoah cause greater visibility and acidic deposition impacts at the park, on a per ton basis, than do more distant emissions sources (USNPS, 2003). When mapping deposition and concentration patterns for all three pollutants using contour lines, the resulting geographic pattern shows a definite eastward tilt in the area of highest impact. This is the result of prevailing wind patterns, which tend to transport most airborne pollutants in an arc[14] from the north-northeast to the east. The Park Service found, for example, that emissions originating in the Ohio River Valley end up three times farther to the east than to the west (USNPS, 2003).

The recent sulfate attribution work completed by MANE-VU (NESCAUM, 2006) finds that a variety of different states contribute to observed sulfate in rural locations across the MANE-VU region, but that in the southwest portions of the region, neighboring RPOs contribute to a more significant degree relative to rural areas in the northeast portions. Figure 2-7 shows relative contributions of RPOs to sulfate at three MANE-VU Class I areas and one VISTAS Class I area based on a variety of analysis methods. Figure 2-8 shows the individual state contributions to sulfate at Brigantine Wilderness Area on the New Jersey coast according to tagged REMSAD modeling.

Figure 2-7. 2002 Annual average contribution to PM₂.₅ sulfate as determined by multiple analysis methods for four Class I areas spanning MANE-VU and Virginia

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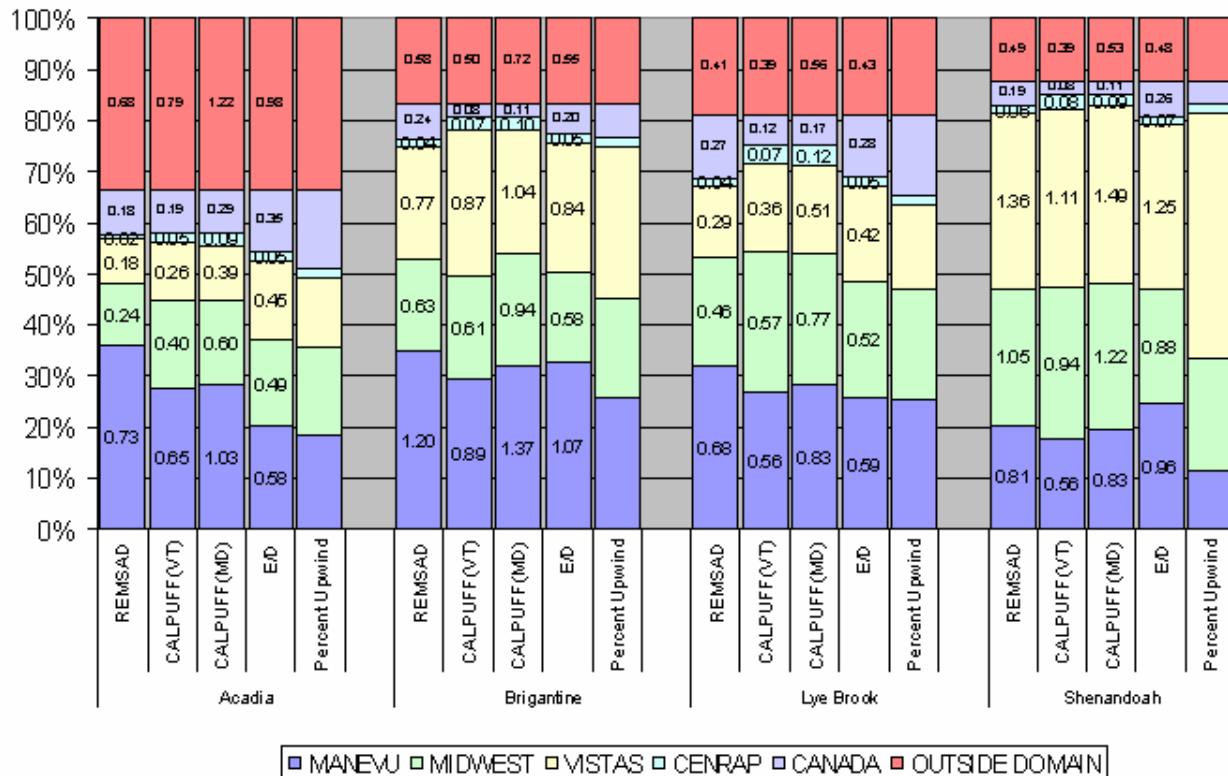
[14] The prevailing winds are eastward to northeast. This leads to greater pollution transport to the east-northeast relative to other directions.

**Figure 2-8. 2002 Annual average mass contribution to PM$_{2.5}$ at
Brigantine Wilderness in New Jersey (IMPROVE) and sulfate contributions as determined by
tagged REMSAD model simulations (NESCAUM, 2006)**

2000-2004 average
IMPROVE mass

REMSAD 2002 average
contributions to sulfate
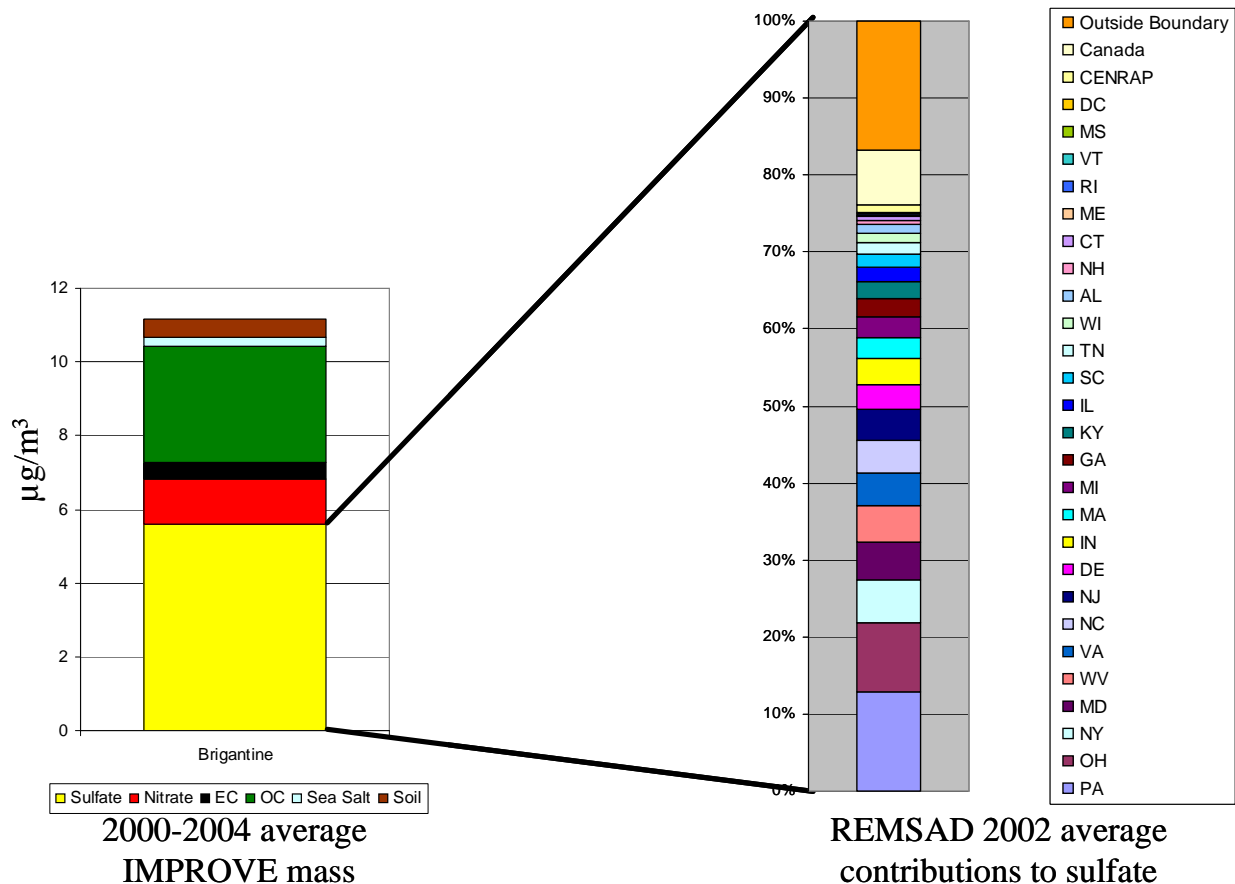
2.4. CAIR Modeling

In 2005, the USEPA promulgated the Clean Air Interstate Rule (CAIR), requiring additional NO$_X$ reductions in 25 eastern states and the District of Columbia.[15] The CAIR modeling by the USEPA provides information on the upwind areas (by state) contributing to downwind nonattainment for PM$_{2.5}$ in MANE-VU counties. Table 2-1 presents the upwind states significantly contributing to PM$_{2.5}$ nonattainment in counties within MANE-VU during 2001, according to significance criteria used by the USEPA (USEPA, 2005, from Table VII-3). The states listed in the table as significantly contributing to downwind nonattainment in MANE-VU counties include states outside of MANE-VU, indicating the broad regional scale of the PM$_{2.5}$ transport problem.

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[15] CAIR was subsequently remanded back to the USEPA as the result of legal challenges. The court rejected the USEPA's regulatory approach as a policy matter under CAIR, but did not reject the rule's technical basis described in this section.

Table 2-2 provides the maximum contribution from each state to annual average PM$_{2.5}$ nonattainment in a downwind state (not necessarily restricted to MANE-VU nonattainment counties) based on CAIR modeling.

Table 2-1. Upwind states that make a significant contribution to PM$_{2.5}$ in each downwind nonattainment county (2001 modeling)

Downwind State/County		Upwind States								
DE	New Castle	MD/DC	MI	NY	OH	PA	VA	WV			
DC	District of Columbia	NC	OH	PA	VA	WV					
MD	Anne Arundel	NC	OH	PA	VA	WV					
MD	Baltimore City	NC	OH	PA	VA	WV					
NJ	Union	MD/DC	MI	NY	OH	PA	WV				
NY	New York	MD/DC	OH	PA	WV						
PA	Allegheny	IL	IN	KY	MI	OH	WV				
PA	Beaver	IN	MI	OH	WV						
PA	Berks	MD/DC	MI	NY	OH	VA	WV				
PA	Cambria	IN	MD/DC	MI	OH	WV					
PA	Dauphin	MD/DC	MI	OH	VA	WV					
PA	Delaware	MD/DC	MI	OH	VA	WV					
PA	Lancaster	IN	MD/DC	MI	NY	OH	VA	WV			
PA	Philadelphia	MD/DC	MI	OH	VA	WV					
PA	Washington	IN	KY	MI	OH	WV					
PA	Westmoreland	IN	KY	MD/DC	MI	OH	WV				
PA	York	MD/DC	MI	OH	VA	WV					

Table 2-2. Maximum downwind PM$_{2.5}$ contribution (µg/m^3) for each of the 37 upwind states (2001 data)

Upwind State	Maximum Downwind Contribution	Upwind State	Maximum Downwind Contribution
Alabama	0.98	Nebraska	0.07
Arkansas	0.19	New Hampshire	<0.05
Connecticut	<0.05	New Jersey	0.13
Delaware	0.14	New York	0.34
Florida	0.45	North Carolina	0.31
Georgia	1.27	North Dakota	0.11
Illinois	1.02	Ohio	1.67
Indiana	0.91	Oklahoma	0.12
Iowa	0.28	Pennsylvania	0.89
Kansas	0.11	Rhode Island	<0.05
Kentucky	0.9	South Carolina	0.4
Louisiana	0.25	South Dakota	<0.05
Maine	<0.05	Tennessee	0.65

Maryland/DC	0.69	Texas	0.29
Massachusetts	0.07	Vermont	<0.05
Michigan	0.62	Virginia	0.44
Minnesota	0.21	West Virginia	0.84
Mississippi	0.23	Wisconsin	0.56
Missouri	1.07		

Consistent with the CAIR results, modeling by Bergin *et al.* (2007) indicated that an average of 77 percent of each state's $PM_{2.5}$ (and ozone) concentrations sensitive to the NO_X and SO_2 emissions evaluated in the model were caused by emissions from other states. Specific to the MANE-VU region, Delaware, Maryland, New Jersey, and Virginia had high concentrations of $PM_{2.5}$ (and ozone) caused by interstate emissions.

2.5. Seasonal differences

Eastern and western coastal regions of the United States and Canada show marked seasonality in the concentration and composition of fine particle pollution, while central interior regions do not (NARSTO, 2003). While MANE-VU extends inland as far as the Pennsylvania and Ohio border, the majority of $PM_{2.5}$ NAAQS nonattainment areas and Class I areas affected by the Regional Haze Rule cluster along the East Coast and thus typically show strong seasonal influences. Maximum $PM_{2.5}$ concentrations typically occur during the summer over most of the rural Northeast, with observed summer values for rural areas in the region, on average, twice those of winter. In urban locations, summertime and wintertime $PM_{2.5}$ levels are more comparable and whether one season dominates over the other is more of a function of inter-annual variability of meteorology and fire activity (i.e., summertime fire activity can push average $PM_{2.5}$ values higher in some years).

The reason for the wintertime strength of $PM_{2.5}$ levels in urban areas is related to the greater concentration of local pollution that accumulates when temperature inversions are present, significantly boosting the wintertime $PM_{2.5}$ levels. Winter nitrate concentrations are generally higher than those observed in summer and, as mentioned above, urban concentrations typically exceed rural concentrations year-round. In addition, local mobile source carbon grows in importance during wintertime. Hence, in some large urban areas such as Philadelphia and New York City, peak concentrations of $PM_{2.5}$ can occur in winter.

The conceptual descriptions that explain elevated regional $PM_{2.5}$ peak concentrations in the summer differs significantly from those that explain the largely urban peaks observed during winter. On average, summertime concentrations of sulfate in the northeastern United States are more than twice that of the next most important fine particle constituent, OC, and more than four times the combined concentration of nitrate and black carbon (BC) constituents (NARSTO, 2003). Episodes of high summertime sulfate concentrations are consistent with stagnant meteorological flow conditions upwind of MANE-VU and the accumulation of airborne sulfate (via atmospheric oxidation of SO_2) followed by long-range transport of sulfur emissions from industrialized areas within and outside the region.

National assessments (NARSTO, 2003) have indicated that in the winter, sulfate levels in urban areas are almost twice as high as background sulfate levels across the

eastern U.S., indicating that the local urban contribution to wintertime sulfate levels is comparable in magnitude to the regional sulfate contribution from long-range transport. MANE-VU's network analysis for the winter of 2002 suggests that the local enhancement of sulfate in urban areas of MANE-VU is somewhat less with ranges from 25 to 40 percent and that the long-range transport component of PM$_{2.5}$ sulfate is still the dominant contributor in most eastern cities.

In the winter, urban OC and sulfate each account for about a third of the overall PM$_{2.5}$ mass concentration observed in Philadelphia and New York City. Nitrate also makes a significant contribution to urban PM$_{2.5}$ levels observed in the northeastern United States during the winter months. Wintertime concentrations of OC and NO$_3$ in urban areas can be twice the average regional concentrations of these pollutants, indicating the importance of local source contributions (NARSTO, 2003). This is likely because winter conditions are more conducive to the formation of local inversion layers that prevent vertical mixing. Under these conditions, emissions from tailpipe, industrial, and other local sources become concentrated near the Earth's surface, adding to background pollution levels associated with regionally transported emissions.

It is worth noting that while sulfate plays a significant role in episodes of elevated particle pollution during summer and winter months, the processes by which sulfate forms may vary seasonally. Nearly every source apportionment study reviewed by USEPA (2003) identified secondary sulfate originating from coal combustion sources as the largest or one of the largest contributors to overall fine particle mass in the region. It often accounted for more than 50 percent of PM$_{2.5}$ mass at some locations during some seasons. In a few cases, source apportionment studies identified a known local source of sulfate, but most assessments (in conjunction with back trajectory analysis) have pointed to coal-fired power plants in the Midwest as an important source for regional sulfate. Studies with multiple years of data have also tended to identify a distinguishable chemical "signature" for winter versus summer sources of sulfate, with the summer version typically accounting for a greater share of overall fine particle mass. Researchers have speculated that the two profiles represent two extremes in the chemical transformation processes that occur in the atmosphere between the source regions where emissions are released and downwind receptor sites. We note that while coal combustion is often referred to as the "sulfate source" because of the dominance of its sulfate contribution, coal combustion is often a source of significant amounts of organic carbon and is usually the single largest source of selenium (Se) and other heavy metal trace elements (USEPA, 2003).

Similarly, chemical transformations of organic particles can differ between seasons. At a semi-rural site in New Hampshire, total carbon concentrations were higher in the winter than the summer months. Primary emissions of carbon (from local heating or industrial emissions) appeared to be the main sources during the winter, while secondary aerosol formation dominated in the other seasons (Shakya *et al.*, 2012).

Figure 2-9. Moving 60-day average of fine aerosol mass concentrations based on long-term data from two northeastern cities
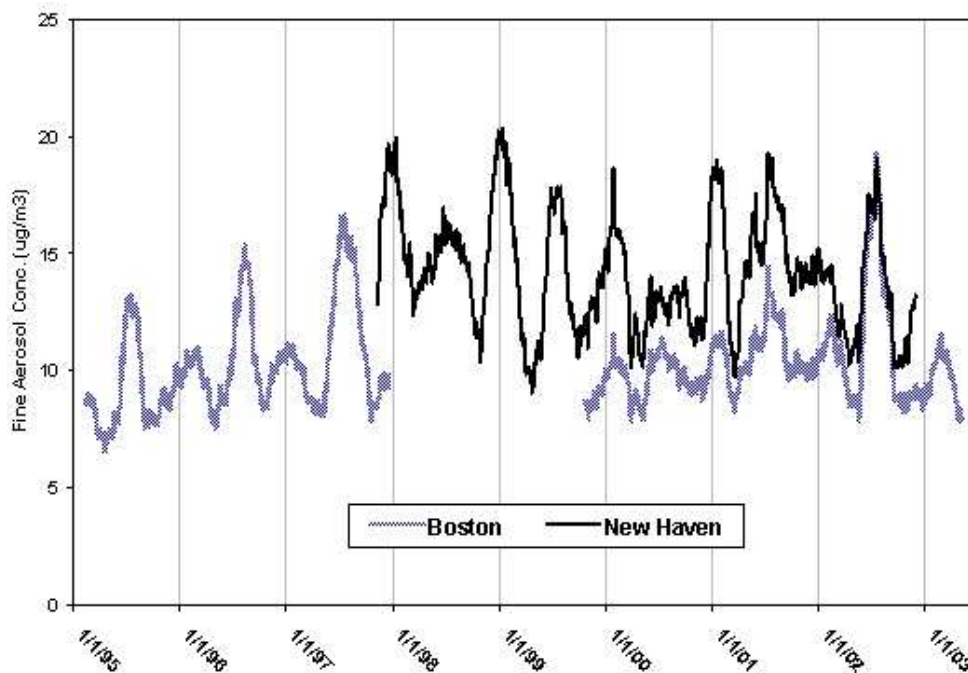
In general, fine particle concentrations in MANE-VU are highest during the warmest (summer) months but also exhibit a secondary peak during the coldest (winter) months that can dominate during some years, particularly in urban locations. This bimodal seasonal distribution of peak values is readily apparent in Figure 2-9. The figure shows the smoothed 60-day running average of fine particle mass concentrations using continuous monitoring data from two northeastern cities over a period of several years.

Figure 2-10 also demonstrates this bimodal pattern. Though slightly more difficult to discern in just a single year's worth of data, a "W" pattern does emerge at almost all sites across the region during 2002 with the winter peak somewhat lower than the summer peak at most sites. Urban monitors in Wilmington, Delaware and New Haven, Connecticut have wintertime peak values approaching those of summer.

In the summertime, MANE-VU sites repeatedly experience sulfate events due to transport from regions to the south and west. During such events, both rural and urban sites throughout MANE-VU record high (i.e., > 15 µg/m³) daily average PM₂.₅ concentrations. Meteorological conditions during the summer frequently allow for summer "stagnation" events when very low wind speeds and warm temperatures (upwind and over MANE-VU) allow pollution levels to build in an air mass as it slowly moves across the continent. During these events, atmospheric ventilation is poor and local emission sources add to the burden of transported pollution with the result that concentrations throughout the region (both rural and urban) are relatively uniform. Generally, there are enough of these events to drive the difference between urban and rural sites down to less than 1 µg/m³ during the warm or hot months of the year. As a result, concentrations of fine particles aloft will often be higher than at ground-level during the summertime, especially at rural monitoring sites. Thus, when atmospheric

"mixing" occurs during summer[16] mornings (primarily 7 to 11 a.m.), fine particle concentrations at ground-level can actually increase (see Hartford, CT or Camden, NJ in Figure 2-11).

Figure 2-10. The 30-day average PM$_{2.5}$ concentrations from 8 northeastern cities during 2002

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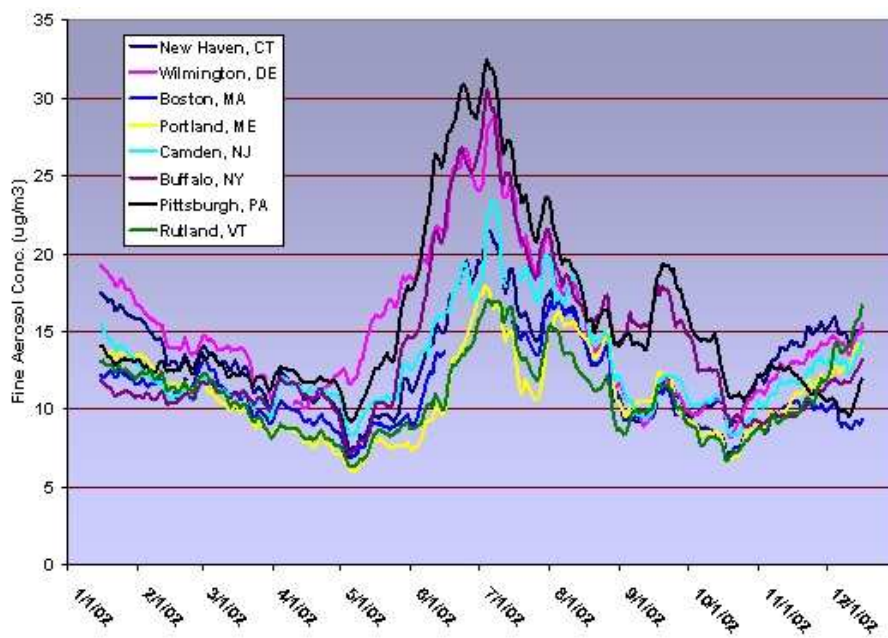
[16] Here we define summer as May, June, July and August.

Figure 2-11. Mean hourly fine aerosol concentrations during 2002 summer months
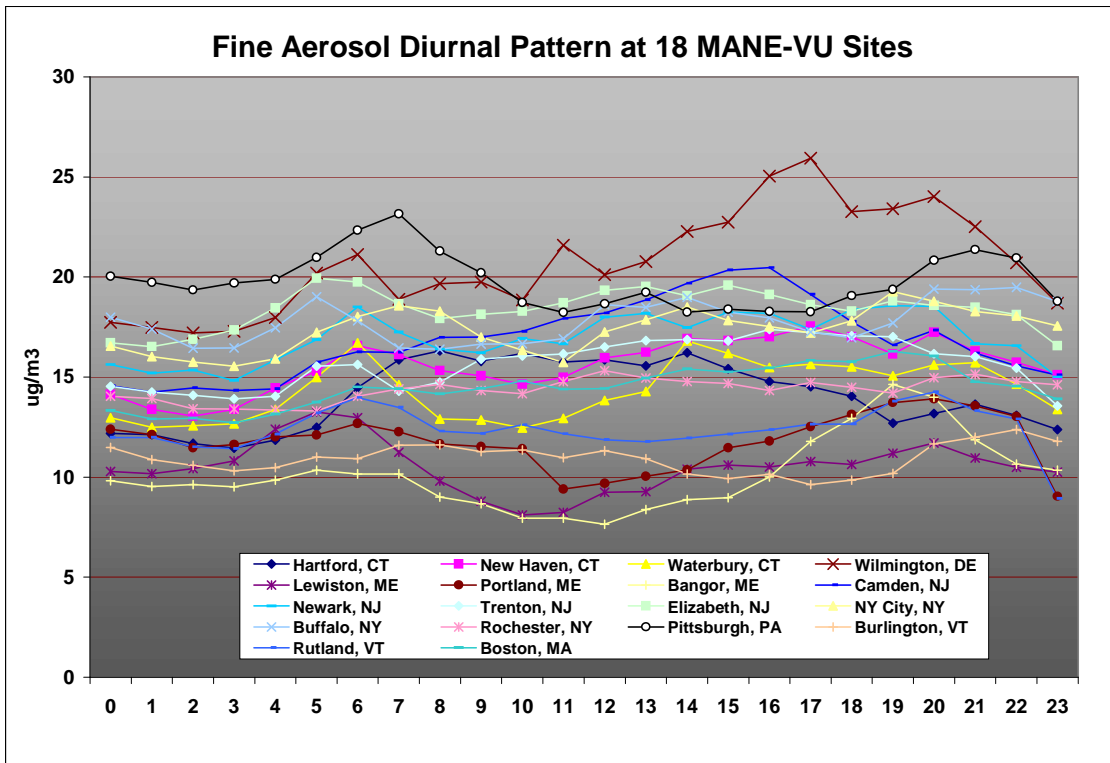
Figure 2-12. Mean hourly fine aerosol concentrations during 2002 winter months
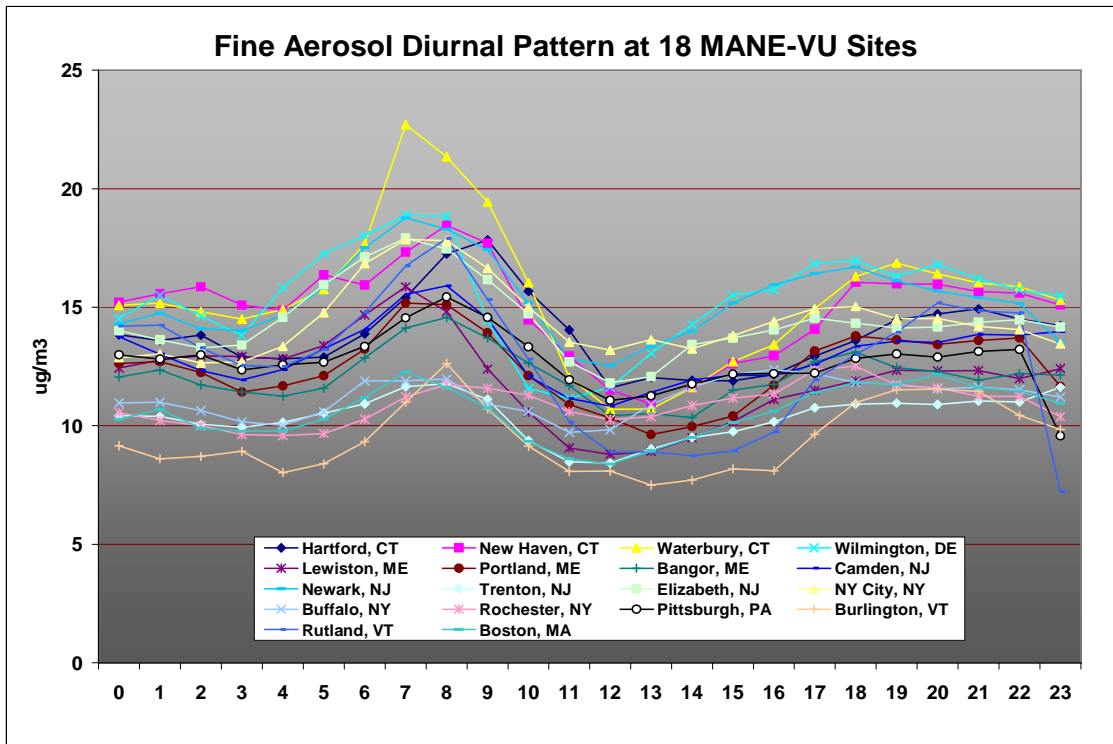
During the wintertime, strong inversions frequently trap local emissions overnight and during the early morning, resulting in elevated urban concentrations. These inversions occur when the Earth's surface loses thermal energy by radiating it into the atmosphere (especially on clear nights). The result is a cold, stable layer of air near the ground. At sunrise, local emissions (both mobile and stationary) begin increasing in strength and build-up in the stable ground layer (which may extend only 100 meters or less above the ground). Increasing solar radiation during the period between 10 a.m. and noon typically breaks this cycle by warming the ground layer so that it can rise and mix with air aloft. Because the air aloft during wintertime is typically less polluted than the surface layer, this mixing tends to reduce ground-level particle concentrations (see Figure 2-12). This diurnal cycle generally drives wintertime particle concentrations, although the occasional persistent temperature inversion can have the effect of trapping and concentrating local emissions over a period of several days, thereby producing a significant wintertime pollution episode.

Rural areas experience the same temperature inversions but have relatively fewer local emissions sources so that wintertime concentrations in rural locations tend to be lower than those in nearby urban areas. Medium and long-range fine particle transport events do occur during the winter but to a far lesser extent than in the summertime. In sum, it is the interplay between local and distant sources together with seasonal meteorological conditions that drives the observed 3–4 $\mu g/m^3$ wintertime urban-rural difference in PM$_{2.5}$ concentrations.

Visually hazy summer days in the Northeast can appear quite different from hazy winter days. The milky, uniform visibility impairment shown in Figure 2-13 is typical of summertime regional haze events in the Northeast. During the winter, by comparison, reduced convection and the frequent occurrence of shallow inversion layers often creates a layered haze with a brownish tinge, as shown in Figure 2-14. This visual difference suggests seasonal variation in the relative contribution of different gaseous and particle constituents during the summer versus winter months (NESCAUM, 2001). Rural and inland areas tend not to experience these layered haze episodes as frequently due to the lack of local emission sources in most rural areas (valleys with high wood smoke contributions are an exception).

Overall (regional) differences in summer versus winter particle mass concentrations and corresponding visibility impairment (as measured by light extinction) are largely driven by seasonal variation in sulfate mass concentrations. This is because winter meteorological conditions are less conducive to the oxidation of sulfate from SO$_2$ (as borne out by the previously cited source apportionment studies). In addition, seasonal differences in long-range transport patterns from upwind SO$_2$ source regions may be a factor.

The greater presence of nitrate during the cold season is a consequence of the chemical properties of ammonium nitrate. Ammonia bonds more weakly to nitrate than it does to sulfate, and ammonium nitrate tends to dissociate at higher temperatures. Consequently, ammonium nitrate becomes more stable at lower temperatures and hence contributes more to PM$_{2.5}$ mass and light extinction during the winter months relative to the summer (NESCAUM, 2001).

Figure 2-13. Summertime at Mt. Washington

Clean Day   **Typical Haze Event**

Figure 2-14. Wintertime in Boston
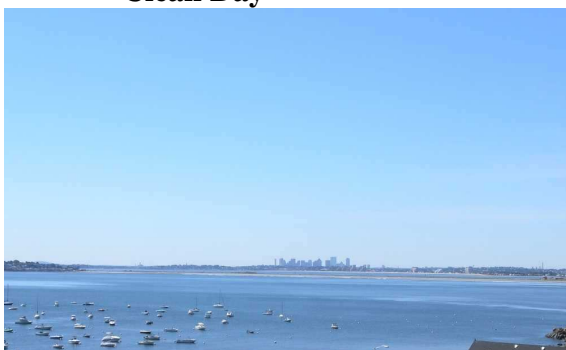
Clean Day   **Typical Haze Event**
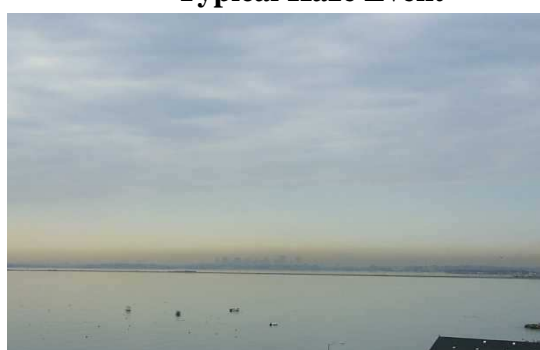
2.6. Exceptional events

Not all PM$_{2.5}$ and haze events are attributable to anthropogenic sources. There may be irregular instances in which transported aerosol levels arise from natural or other relatively rare events that are not under the control of air quality planners. An example of this would be long range transport of aerosols in smoke plumes from wildfires. These types of events need to be identified so that efforts are targeted towards meeting air quality goals that are not unrealistically tied to events beyond the control of air quality planners.

To address these instances, Congress amended section 319 of the Clean Air Act when it passed the Safe Accountable Flexible Efficient-Transportation Equity Act: A Legacy for Users (SAFE–TEA–LU) of 2005 to include a provision for identifying "exceptional events." The concept of "exceptional events" is to identify instances of air pollution for which the normal CAA planning and regulatory processes are not appropriate.

The amended CAA section 319 defines an exceptional event as an event that affects air quality; is an event that is not reasonably controllable or preventable; is an event caused by human activity that is unlikely to recur at a particular location or a natural event; and is determined by EPA to be an exceptional event. The statutory definition of exceptional event specifically excludes stagnation of air masses or meteorological inversions; a meteorological event involving high temperatures or lack of precipitation; or air pollution relating to source noncompliance. The USEPA has

established by rule what is required by states to demonstrate the occurrence of exceptional events, thus allowing the exclusion of air quality monitoring data that would otherwise show an exceedance or violation of a NAAQS (72 Fed. Reg. 13560 (March 22, 2007)).

The long range transport of smoke from forest wildfires would qualify as "exceptional events" to the extent they are considered by the USEPA as meeting the criteria set out in CAA section 319. Observed events of air pollutants associated with long range transport from wildfires include air quality impacts in the Washington, DC area from forest fires in central Quebec (Colarco *et al.*, 2004), and in Houston, TX from forest fires in eastern Alaska and western Canada (Morris *et al.*, 2006). Smoke from wildfires in Quebec was seen as recently as May 31, 2010 in portions of New England down through Boston.

2.7. Summary

The presence of fine particulate matter in ambient air significantly degrades public health and obscures visibility during most parts of the year at sites across the MANE-VU region. Particle pollution generally, and its sulfate component specifically, constitute the principle driver for regional visibility impacts. While the broad region experiences visibility impairment, it is most severe in the southern and western portions of MANE-VU that are closest to large power plant SO_2 sources in the Ohio River and Tennessee Valleys.

Summer visibility impairment is driven by the presence of regional sulfate, whereas winter visibility depends on a combination of regional and local influences coupled with local meteorological conditions (inversions) that lead to the concentrated build-up of pollution.

Sulfate is the key particle constituent from the standpoint of designing control strategies to improve visibility conditions in the northeastern United States. Significant further reductions in ambient sulfate levels are achievable, though they will require more than proportional reductions in SO_2 emissions.

Long-range pollutant transport and local pollutant emissions are important, especially along the eastern seaboard, so one must also look beyond the achievement of further sulfate reductions. During the winter months, in particular, consideration also needs to be given to reducing urban sources of SO_2, NO_X and OC (NARSTO, 2003).

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3. MANE-VU EMISSION INVENTORY CHARACTERISTICS FOR FINE PARTICLES

The pollutants that affect fine particle formation and visibility are sulfur oxides (SO$_X$), NO$_X$, VOCs, ammonia (NH$_3$), and particles with an aerodynamic diameter less than or equal to 10 and 2.5 µm (i.e., primary PM$_{10}$ and PM$_{2.5}$). The emissions dataset illustrated in this section is the 2002 MANE-VU Version 2 regional haze emissions inventory. The MANE-VU regional haze emissions inventory version 3.0, released in April 2006, has superseded version 2 for modeling purposes.

Note that in future inventory efforts, the USEPA's Motor Vehicle Emission Simulator (MOVES) model will be replacing the previous mobile source inventory models – MOBILE for on-road and NONROAD for non-road emissions inventory development. On average, this change is anticipated to: (1) lower modeled CO and VOC emissions; (2) increase modeled NO$_X$ and PM emissions, (3) lower the percent NO$_X$ reduction from modeled control measures; and (4) raise the percent PM reduction from modeled control measures for the on-road and non-road sectors (Dolce, 2009).

3.1. Emissions inventory characteristics

3.1.1. Sulfur dioxide (SO$_2$)

SO$_2$ is the primary precursor pollutant for sulfate particles. Ammonium sulfate particles are the largest contributor to PM$_{2.5}$ mass on an annual average basis at MANE-VU nonattainment sites. It also accounts for more than 50 percent of particle-related light extinction at northeastern Class I areas on the clearest days and for as much as or more than 80 percent on the haziest days. Hence, SO$_2$ emissions are an obvious target of opportunity for both addressing PM$_{2.5}$ nonattainment and for reducing regional haze in the eastern United States. Combustion of coal and, to a substantially lesser extent, of certain petroleum products accounts for most anthropogenic SO$_2$ emissions. In fact, in 1998 a single source category — coal-burning power plants — was responsible for two-thirds of total SO$_2$ emissions nationwide (NESCAUM, 2001).

Figure 3-1 shows SO$_2$ emissions trends in MANE-VU states[17] extracted from the National Emissions Inventories (NEI) for the years 1996, 1999, and the 2002 MANE-VU inventory. Most of the states (with the exception of Maryland) show declines in year 2002 annual SO$_2$ emissions as compared to 1996 emissions. Some of the states show an increase in 1999 followed by a decline in 2002 and others show consistent declines throughout the entire period. The upward trend in emissions after 1996 probably reflects electricity demand growth during the late 1990s combined with the availability of banked SO$_2$ emissions allowances from initial over-compliance with control requirements in Phase 1 of the USEPA Acid Rain Program. This led to relatively low market prices for allowances later in the decade, which encouraged utilities to purchase allowances rather than implement new controls as electricity output expanded. The observed decline in the 2002 SO$_2$ emissions inventory reflects implementation of the second phase of the USEPA

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[17] The description of MANE-VU state inventories discussed throughout this section does not include the portion of Virginia in the Washington, DC metropolitan area.

Acid Rain Program, which in 2000 further reduced allowable emissions and extended emissions limits to more power plants.

Figure 3-2 shows the percent contribution from different source categories to overall annual 2002 SO$_2$ emissions in MANE-VU states. The chart shows that point sources dominate SO$_2$ emissions, which primarily consist of stationary combustion sources for generating electricity, industrial energy, and heat. Smaller stationary combustion sources called "area sources" (primarily commercial and residential heating) are another important source category in MANE-VU states. By contrast, on-road and non-road mobile sources make only a relatively small contribution to overall SO$_2$ emissions in the region (NESCAUM, 2001).

Figure 3-1. State level sulfur dioxide emissions
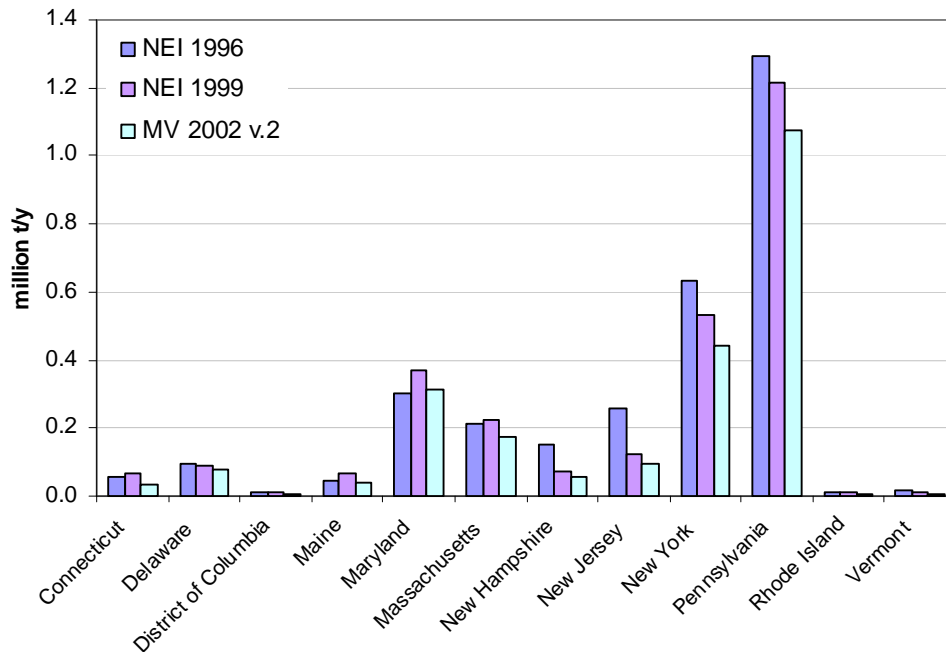
Figure 3-2. 2002 MANE-VU state SO$_2$ inventories

Figure Key: Bars = Percentage fractions of four source categories; Circles = Annual emissions amount in 10^6 tons per year. Note that Version 2 of the MANE-VU inventory was used and the Virginia portion of the Washington, DC metropolitan area is not shown in the figure.

3.1.2. Volatile organic compounds (VOCs)

Existing emission inventories generally refer to VOCs based on their historical contribution to ozone formation. From a fine particle perspective, VOCs (also referred to as hydrocarbons) are of concern because they can react in the atmosphere to form secondary organic aerosol (SOA) as a result of condensation and oxidation processes. The SOA component of fine particles also obscures visibility, but this component has a smaller impact on visibility (on a per unit mass basis) relative to sulfate or nitrate, which have an affinity for water that allows them to significantly "grow" as particles under humid conditions. Nonetheless, organic carbon typically has the second largest visibility impact at most Class I sites next to sulfate, given its large mass contribution.

As shown in Figure 3-3, the VOC inventory is dominated by mobile and area sources. Most VOC emissions in MANE-VU, however, come from natural sources, which are not shown in the figure. Among the human-caused VOC emissions, on-road mobile sources of VOCs include exhaust emissions from gasoline passenger vehicles and diesel-powered heavy-duty vehicles as well as evaporative emissions from transportation fuels. VOC emissions may also originate from a variety of area sources (including solvents, architectural coatings, and dry cleaners) as well as from some point sources (e.g., industrial facilities and petroleum refineries).

Naturally occurring (biogenic) VOC emissions are caused by the release of natural organic compounds from plants in warm weather. Natural, or biogenic, VOCs contribute significantly to fine particle formation. Biogenic VOCs are not included in

Figure 3-3, but nationally, they represent roughly two-thirds of all annual VOC emissions (USEPA, 2006). Biogenic emissions are extremely difficult to estimate, as it requires modeling the behavior of many plants as well as their responses to the environment.

With regard to fine particle formation, understanding the transport dynamics and source regions for organic carbon is likely to be more complex than for sulfate. This is partly because of the large number and variety of VOC species, the fact that their transport characteristics vary widely, and the fact that a given species may undergo numerous complex chemical reactions in the atmosphere. Thus, the organic carbon contribution to fine particles in the East is likely to include manmade pollution transported from a distance, manmade pollution from nearby sources, and biogenic emissions, especially terpenes from coniferous forests.

For fine particles derived from organic carbon, the oxidation of hydrocarbon molecules containing seven or more carbon atoms is generally the most significant pathway for their formation (Odum *et al*., 1997). Recent research, however, suggests that smaller reactive hydrocarbons like isoprene not only contribute significantly to ground-level ozone, which may indirectly impact organic aerosol formation, but also contribute directly to ambient organic aerosol through heterogeneous processes (Claeys *et al*., 2004; Kroll *et al*., 2005).

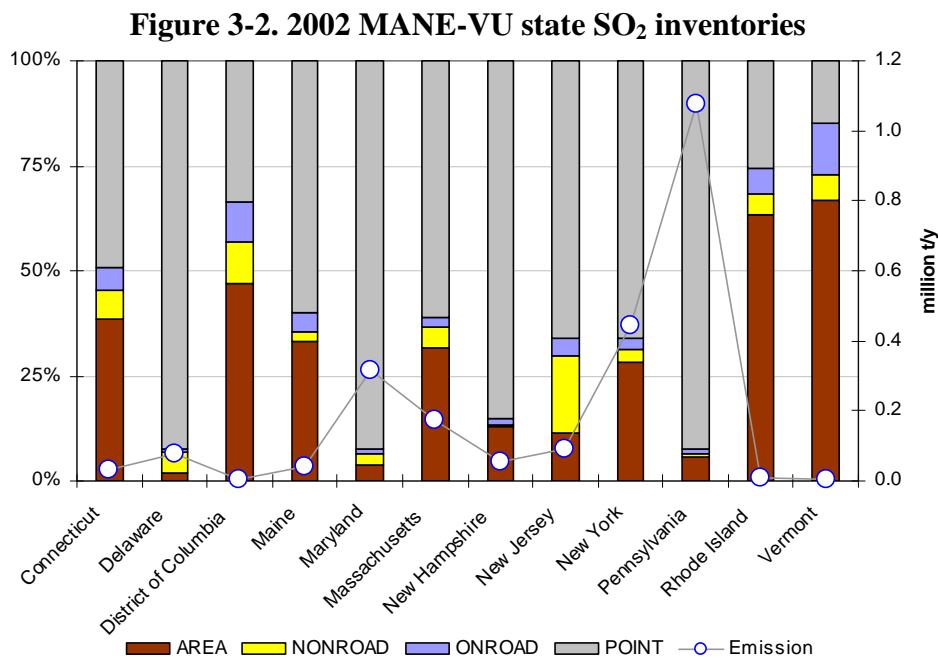**Figure 3-3. 2002 MANE-VU state VOC inventories**

Figure key: Bars = Percentage fractions of four source categories; Circles = Annual emissions amount in 10[6] tons per year. Note that Version 2 of the MANE-VU inventory was used and the Virginia portion of the Washington, DC metropolitan area is not shown in the figure. Biogenic VOCs are not included in this figure.

Figure 3-4. State level nitrogen oxides emissions
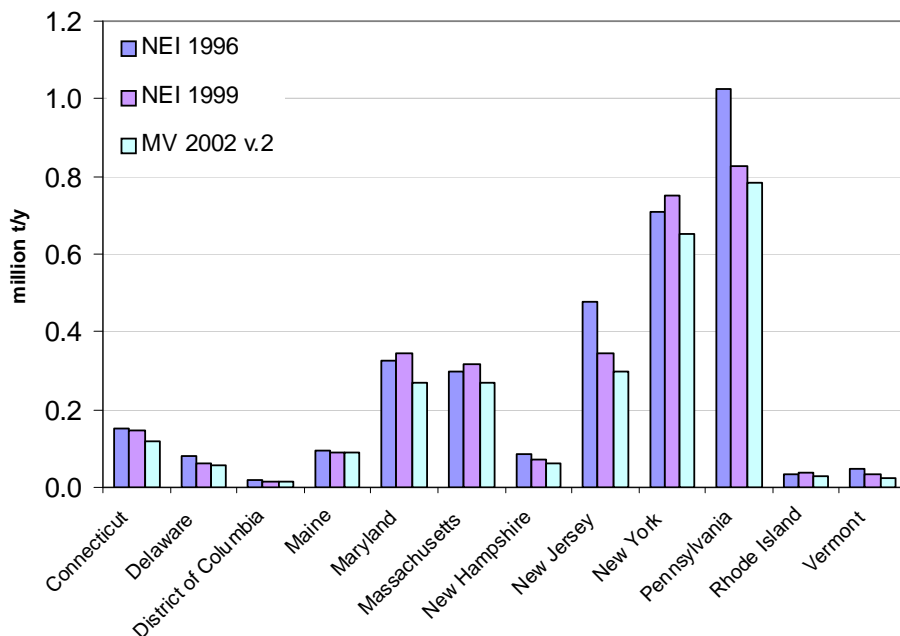
3.1.3. Oxides of nitrogen (NO$_X$)

NO$_X$ emissions contribute directly to PM$_{2.5}$ nonattainment and visibility impairment in the eastern U.S. by forming nitrate particles. Nitrate generally accounts for a substantially smaller fraction of fine particle mass and related light extinction than sulfate and organic carbon regionally in MANE-VU. Notably, nitrate may play a more important role at urban sites and in the wintertime. In addition, NO$_X$ may have an indirect effect on summertime visibility by virtue of its role in the formation of ozone, which in turn promotes the formation of secondary organic aerosols (NESCAUM, 2001).

Figure 3-4 shows NO$_X$ emissions in MANE-VU at the state level. Between 1980 and 1998, nationwide emissions of NO$_X$ from all sources showed little change (USEPA, 2000a). Since the late 1990s, with the implementation of more stringent mobile source, power plant, and other source sector NO$_X$ limits, along with an increasing electric generation fuel shift from coal to natural gas, national NO$_X$ emissions have dropped by almost 50 percent (USEPA, 2011). Most states in MANE-VU experienced declining NO$_X$ emissions from 1996 through 2002 (the initial MANE-VU inventory base year), except Massachusetts, Maryland, New York, and Rhode Island, which show an increase in NO$_X$ emissions in 1999 before declining to levels below 1996 emissions in 2002.

Monitored ambient NO$_X$ trends during the summer from 1997 to 2009 corroborate the downward trend in NO$_X$ emissions seen in the emissions inventories for MANE-VU and the nation. As shown in Figure 3-5, the monthly averaged NO$_X$ concentrations indicate decreases in NO$_X$ over this time period in the MANE-VU region. The NO$_X$ reductions likely come from decreasing vehicle NO$_X$ emissions due to more stringent motor vehicle standards as well as NO$_X$ reductions from the Ozone Transport Region (OTR) NO$_X$ Budget Program and the NO$_X$ SIP Call (mainly power plants). The higher NO$_X$ levels measured during the colder months may be for several reasons, including relatively lower atmospheric mixing heights during colder months (i.e., less volume for pollutants to disperse in), less stringent NO$_X$ requirements for power plants during months outside the April – September ozone season, and increased space heating demands (e.g., NO$_X$ from combustion of residential heating oil and wood, as well as increased generation for electric heat).

Power plants and mobile sources generally dominate state and national NO$_X$ emissions inventories. Nationally, power plants account for more than one-quarter of all NO$_X$ emissions, amounting to over six million tons. The electric sector plays an even larger role, however, in parts of the industrial Midwest where high NO$_X$ emissions have a particularly significant power plant contribution. By contrast, mobile sources dominate the NO$_X$ inventories for more urbanized mid-Atlantic and New England states to a far greater extent, as shown in Figure 3-6. In these states, on-road mobile sources – a category that mainly includes highway vehicles – represent the most significant NO$_X$

Figure 3-5. Average monthly monitored NO$_X$ trends in MANE-VU, 1997-2009
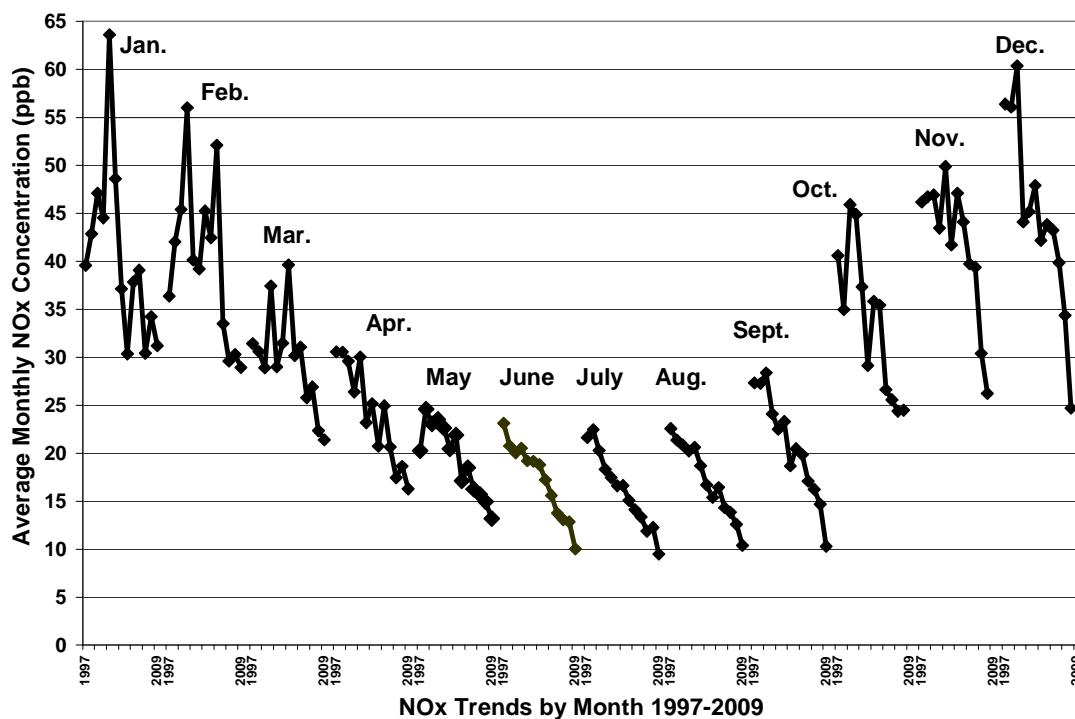
Note: The NO$_X$ trends are 24-hour levels averaged by month during 1997-2009 (figure courtesy of Tom Downs, Maine Department of Environmental Protection).

source category. Emissions from non-road (i.e., off-highway) mobile sources, primarily diesel-fired engines, also represent a substantial fraction of the inventory.

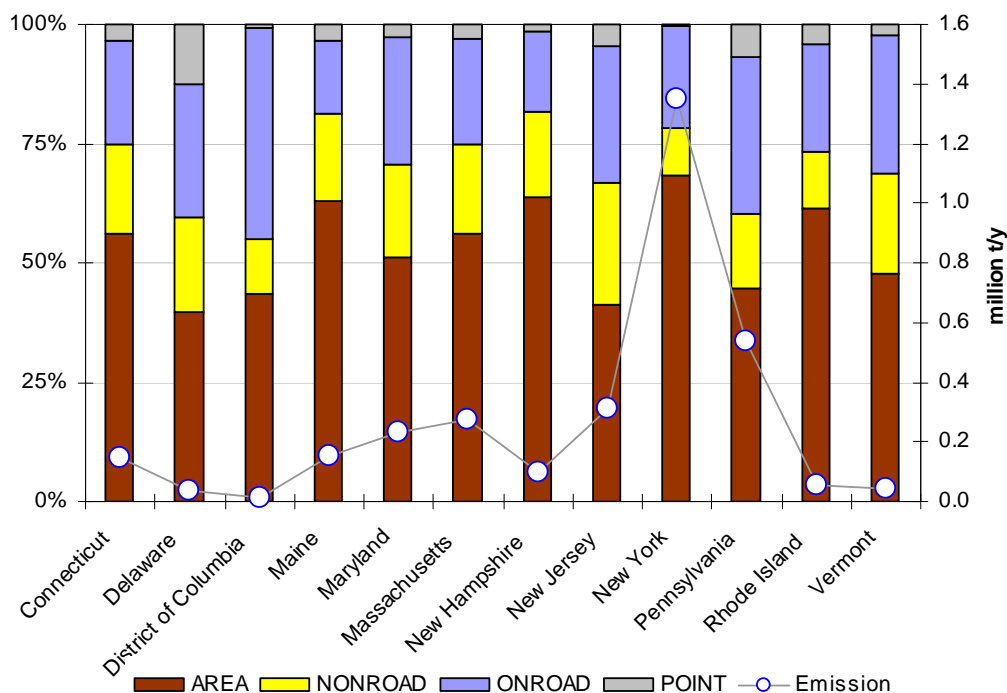**Figure 3-6. 2002 MANE-VU state NO$_X$ inventories**
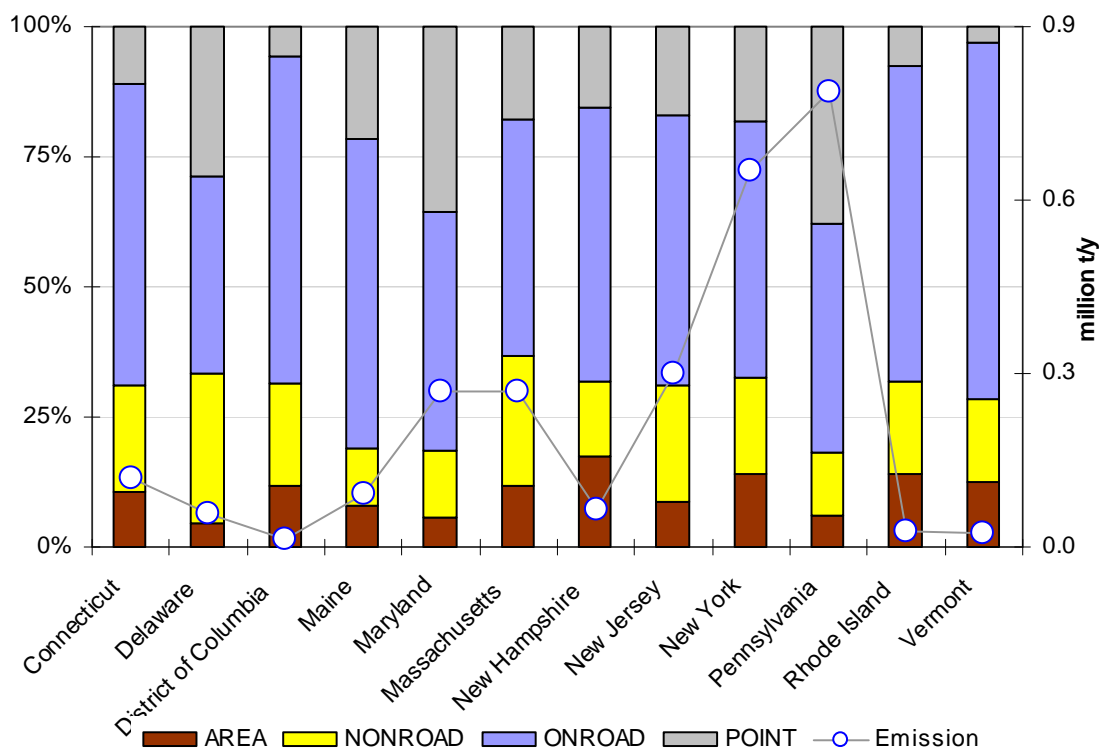
Figure key: Bars = Percentage fractions of four source categories; Circles = Annual emissions amount in 10^6 tons per year. Note that Version 2 of the MANE-VU inventory was used and the Virginia portion of the Washington, DC metropolitan area is not shown in the figure.

3.1.4. Primary particulate matter (PM$_{10}$ and PM$_{2.5}$)

Directly-emitted or "primary" particles (as distinct from secondary particles that form in the atmosphere through chemical reactions involving precursor pollutants like SO$_2$ and NO$_X$) also contribute to fine particle levels in the atmosphere. For regulatory purposes, we make a distinction between particles with an aerodynamic diameter less than or equal to 10 micrometers and smaller particles with an aerodynamic diameter less than or equal to 2.5 micrometers (i.e., primary PM$_{10}$ and PM$_{2.5}$, respectively).

Figure 3-7 and Figure 3-8 show PM$_{10}$ and PM$_{2.5}$ emissions for MANE-VU states for the years 1996, 1999, and 2002. Note that, as opposed to the other constituents of PM, the 2002 inventory values for PM$_{10}$ are drawn from the 2002 NEI. Most states show a steady decline in annual PM$_{10}$ emissions over this time period. By contrast, emission trends for primary PM$_{2.5}$ are more variable.

Crustal sources are significant contributors of primary PM emissions. This category includes fugitive dust emissions from construction activities, paved and unpaved roads, and agricultural tilling. Typically, monitors estimate PM$_{10}$ emissions from these types of sources by measuring the horizontal flux of particulate mass at a fixed downwind

sampling location within perhaps 10 meters of a road or field. Comparisons between estimated emission rates for fine particles using these types of measurement techniques and observed concentrations of crustal matter in the ambient air at downwind receptor sites suggest that physical or chemical processes remove a significant fraction of crustal material relatively quickly. As a result, it rarely entrains into layers of the atmosphere where it can transport to downwind receptor locations. Because of this discrepancy between estimated emissions and observed ambient concentrations, modelers typically reduce estimates of total PM$_{2.5}$ emissions from all crustal sources by applying a factor of 0.15 to 0.25 before including in modeling analyses.

From a regional haze perspective, crustal material generally does not play a major role. On the 20 percent best-visibility days during the baseline period (2000-2004), it accounted for 6 to 11 percent of particle-related light extinction at MANE-VU Class I sites. On the 20 percent worst-visibility days, however, crustal material generally plays a much smaller role relative to other haze-forming pollutants, ranging from 2 to 3 percent. Moreover, the crustal fraction includes material of natural origin (such as soil or sea salt) that is not targeted under USEPA's Regional Haze Rule. Of course, the crustal fraction can be influenced by certain human activities, such as construction, agricultural practices, and road maintenance (including wintertime salting) — thus, to the extent that these types of activities are found to affect visibility at northeastern Class I sites, control measures targeted at crustal material may prove beneficial.

Experience from the western United States, where the crustal component has generally played a more significant role in driving overall particulate levels, may be helpful where it is relevant in the eastern context. In addition, a few areas in the Northeast, such as New Haven, Connecticut and Presque Isle, Maine, have some experience with the control of dust and road-salt as a result of regulatory obligations stemming from their past nonattainment status with respect to the NAAQS for PM$_{10}$.

Current emissions inventories for the entire MANE-VU area indicate residential wood combustion represents 25 percent of primary fine particulate emissions in the region. This implies that rural sources can play an important role in addition to the contribution from the region's many highly populated urban areas. An important consideration in this regard is that residential wood combustion occurs primarily in the winter months, while managed or prescribed burning activities occur largely in other seasons. The latter category includes agricultural field-burning activities, prescribed burning of forested areas, and other burning activities such as construction waste burning. Limiting burning to times when favorable meteorological conditions can efficiently disperse resulting emissions can manage many of these types of sources.

Figure 3-7. State level primary PM$_{10}$ emissions
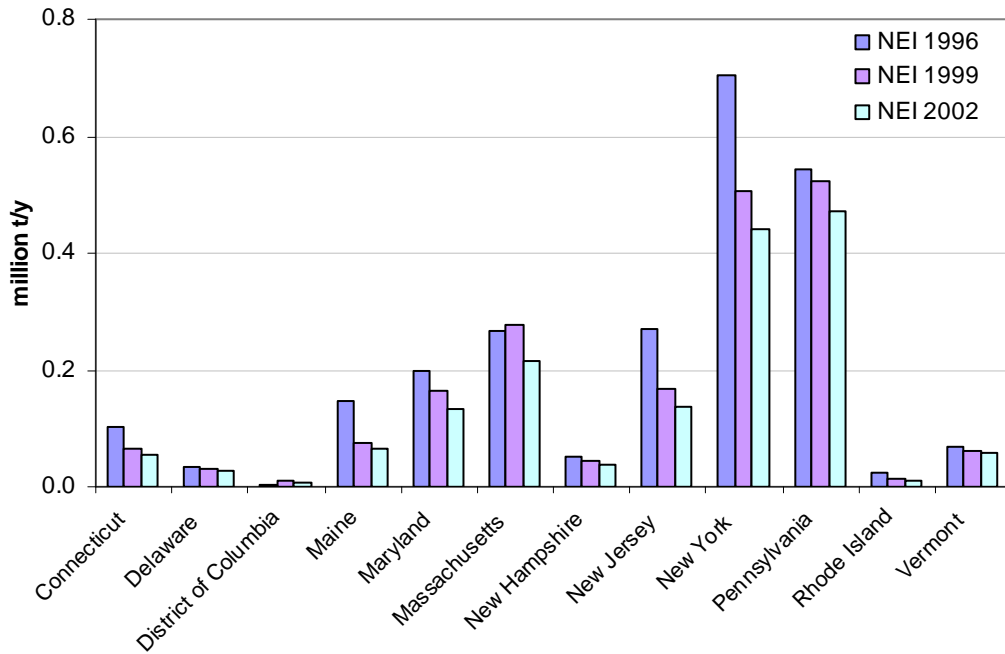
Figure 3-8. State level primary PM$_{2.5}$ emissions
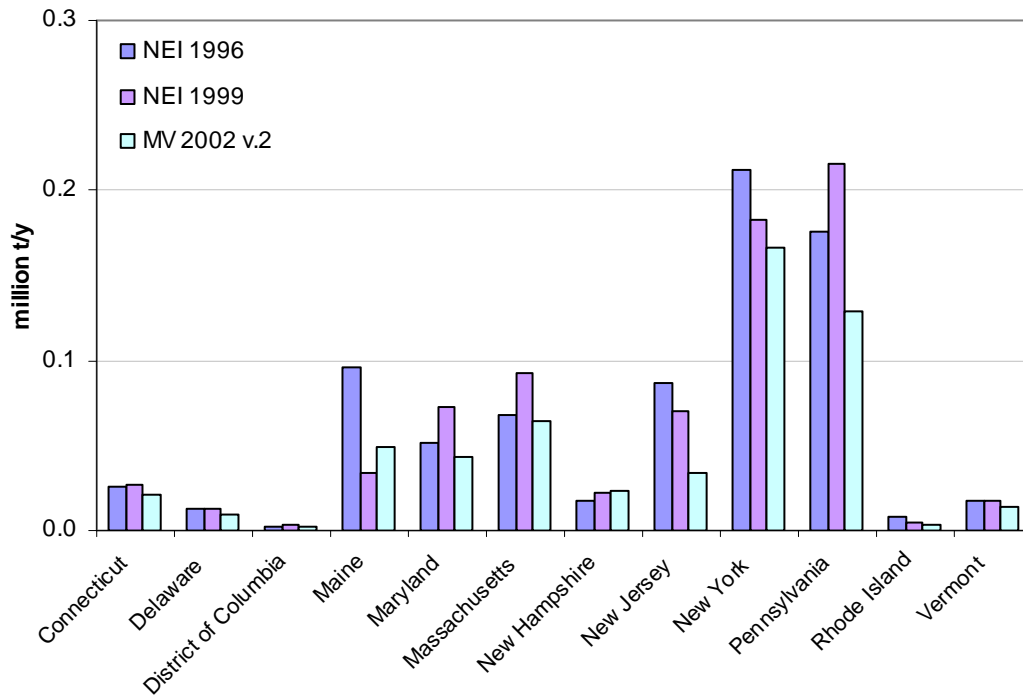
Figure 3-9 and Figure 3-10 show that area and mobile sources dominate primary PM emissions. (The NEI inventory categorizes residential wood combustion and some other combustion sources as area sources.) The relative contribution of point sources is

larger in the primary PM$_{2.5}$ inventory than in the primary PM$_{10}$ inventory since the crustal component (which consists mainly of larger or "coarse-mode" particles) contributes mostly to overall PM$_{10}$ levels. At the same time, pollution control equipment commonly installed at large point sources is usually more efficient at capturing coarse-mode particles.

Figure 3-9. 2002 MANE-VU state primary PM$_{10}$ inventories
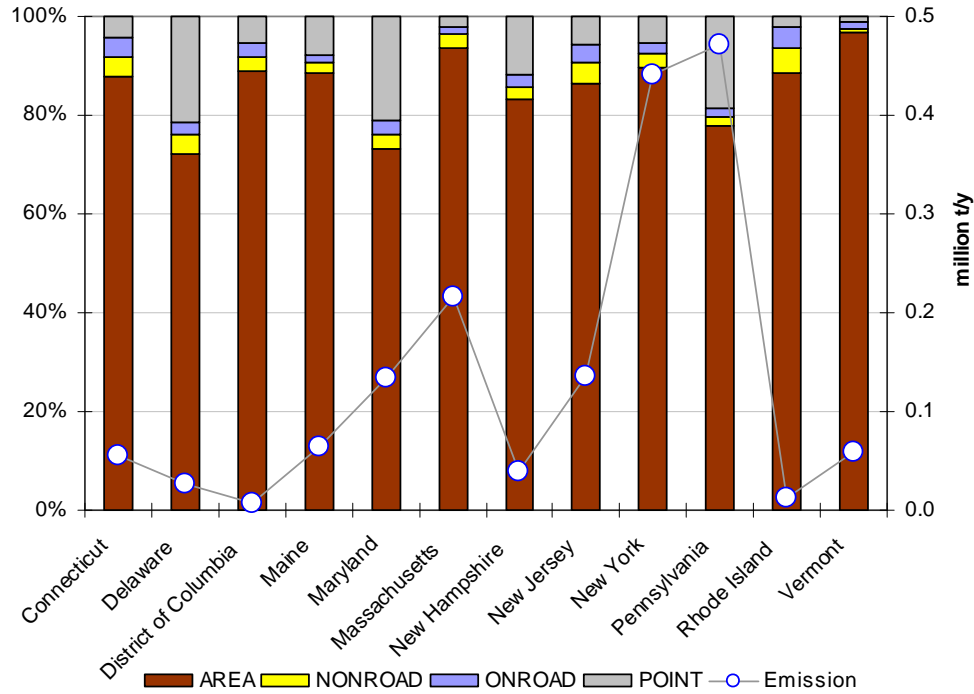
Figure 3-10. 2002 MANE-VU state primary PM$_{2.5}$ inventories

Figure key: Bars = Percentage fractions of four source categories; Circles = Annual emissions amount in 10^6 tons per year. Note that Version 2 of the MANE-VU inventory was used and the Virginia portion of the Washington, DC metropolitan area is not shown in the figure.

3.1.5. Ammonia emissions (NH$_3$)

Knowledge of ammonia emission sources will be necessary in developing effective regional haze reduction strategies because of the importance of ammonium sulfate and ammonium nitrate in determining overall fine particle mass and light scattering. According to 1998 estimates, livestock and agriculture fertilizer use accounted for approximately 85 percent of all ammonia emissions to the atmosphere (USEPA, 2000b). Intensive agricultural activities in the eastern United States have been identified as an important source of transported total ammonia (NH$_3$ + NH$_4^+$) sampled at a coastal location near the Maine-New Hampshire border during the summer of 2004 (Smith *et al.*, 2007). We need, however, better ammonia inventory data for the photochemical models used to simulate fine particle formation and transport in the eastern United States. Because the USEPA does not regulate ammonia as a criteria pollutant or as a criteria pollutant precursor, these data do not presently exist at the same level of detail or certainty as for NO$_X$ and SO$_2$.

Ammonium ion (formed from ammonia emissions to the atmosphere) is an important constituent of airborne particulate matter, typically accounting for 10–20 percent of total fine particle mass. Reductions in ammonium ion concentrations can be extremely beneficial because a more-than-proportional reduction in fine particle mass can result. Ansari and Pandis (1998) showed that a one $\mu g/m^3$ reduction in ammonium ion could result in up to a four $\mu g/m^3$ reduction in fine particulate matter. Decision makers, however, must weigh the benefits of ammonia reduction against the significant role it plays in neutralizing acidic aerosol. SO$_2$ reacts in the atmosphere to form sulfuric acid

(H$_2$SO$_4$). Ammonia can partially or fully neutralize this strong acid to form ammonium bisulfate or ammonium sulfate. If planners focus future control strategies on ammonia and do not achieve corresponding SO$_2$ reductions, fine particles formed in the atmosphere will be substantially more acidic than those presently observed.

To address the need for improved ammonia inventories, MARAMA, NESCAUM and USEPA funded researchers at Carnegie Mellon University (CMU) in Pittsburgh to develop a regional ammonia inventory system (Davidson *et al.*, 1999). This study focused on three issues with respect to current emissions estimates: (1) a wide range of ammonia emission factor values, (2) inadequate temporal and spatial resolution of ammonia emissions estimates, and (3) a lack of standardized ammonia source categories.

Figure 3-11 shows that estimated ammonia emissions were fairly stable in the 1996, 1999, and 2002 NEI for MANE-VU states, with some increases observed for Massachusetts, New Jersey and New York. Area and on-road mobile sources dominate the ammonia inventory, according to Figure 3-12. Specifically, emissions from agricultural sources and livestock production account for the largest share of estimated ammonia emissions in MANE-VU, except in the District of Columbia. The two remaining sources with a significant emissions contribution are wastewater treatment systems and gasoline exhaust from highway vehicles.

Figure 3-11. State level ammonia emissions

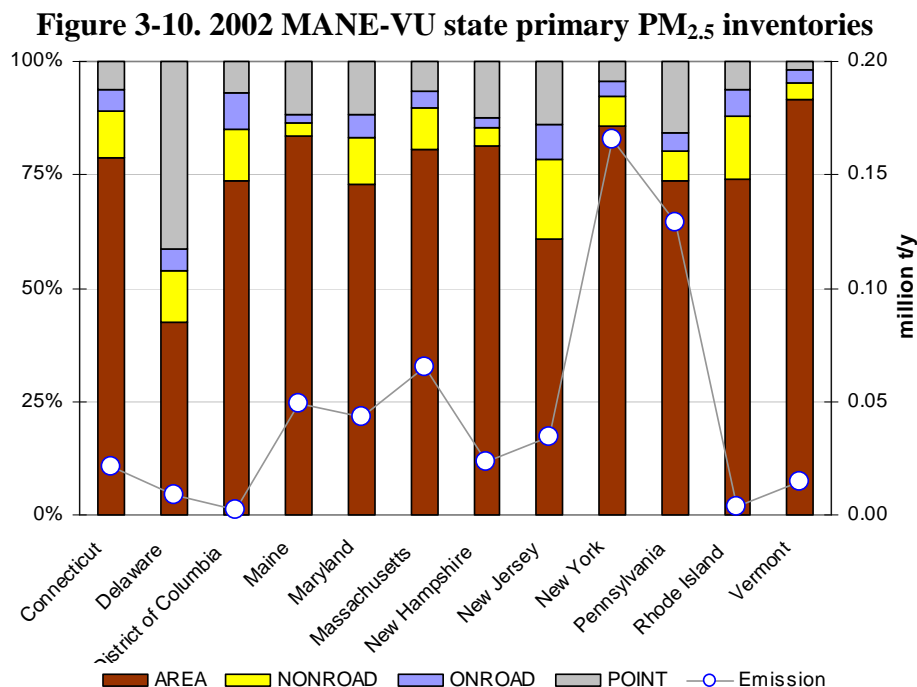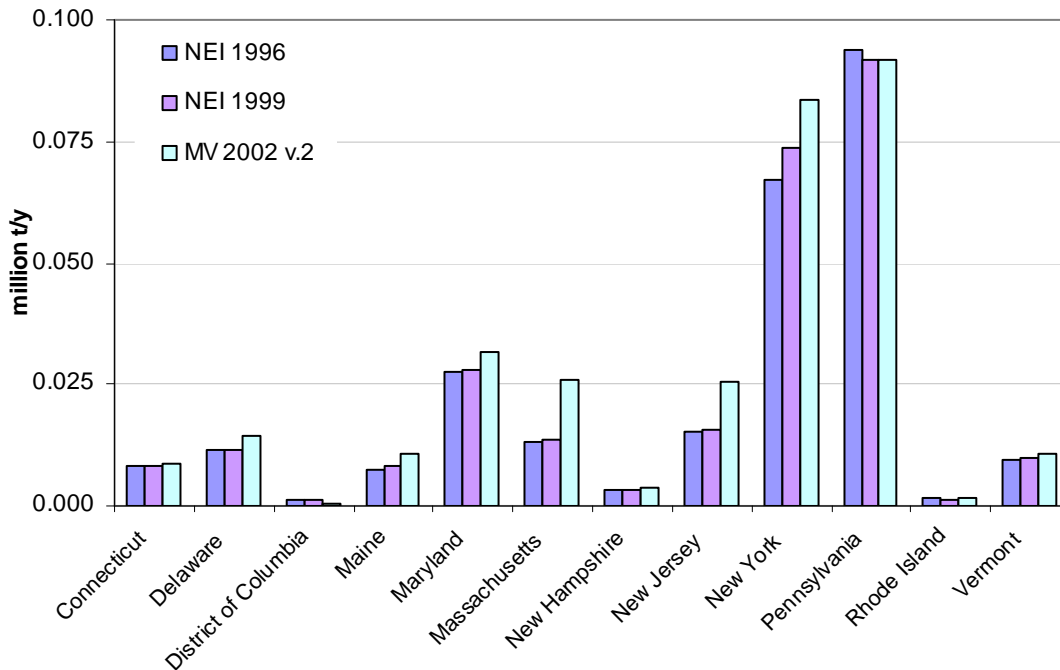**Figure 3-12. 2002 MANE-VU state NH₃ inventories**
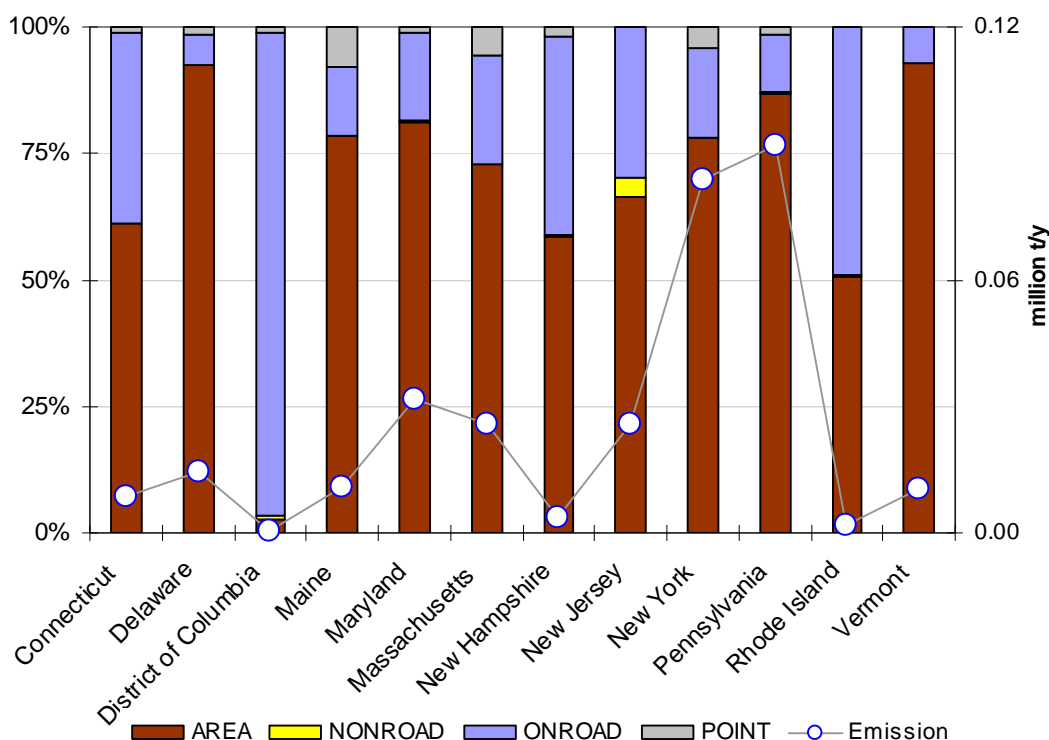
Figure key: Bars = Percentage fractions of four source categories; Circles = Annual emissions amount in 10⁶ tons per year. Note that Version 2 of the MANE-VU inventory was used and the Virginia portion of the Washington, DC metropolitan area is not shown in the figure.

3.2. Emissions inventory characteristics outside MANE-VU

Emissions of SO₂, NOₓ and VOCs from within MANE-VU are only one component of the emissions contributing to fine particles affecting the MANE-VU region. As regional modeling for the CAIR has shown, emission sources, primarily of SO₂ and NOₓ, located outside MANE-VU can significantly contribute to particle sulfate and nitrate transported into the MANE-VU region. Here we present regional emissions information grouped by the three eastern RPOs – MANE-VU, VISTAS (Visibility Improvement State and Tribal Association of the Southeast), and the MWRPO (Midwest RPO). Table 3-1 lists the states in each RPO.

The inventory information is extracted from the USEPA final 2002 National Emissions Inventory (NEI). For consistency, the MANE-VU information here also comes from the 2002 NEI rather than from the MANE-VU Version 2 regional haze emissions inventory described in Section 3.1. The differences between the inventories are not great, as the NEI and the MANE-VU Version 2 inventory are both based on the same inventory information provided by the states.

Table 3-1. Eastern U.S. RPOs and their state members

RPO	State
MWRPO	Illinois
MWRPO	Indiana
MWRPO	Michigan
MWRPO	Ohio
MWRPO	Wisconsin
MANE-VU	Connecticut
MANE-VU	Delaware
MANE-VU	District of Columbia
MANE-VU	Maine
MANE-VU	Maryland
MANE-VU	Massachusetts
MANE-VU	New Hampshire
MANE-VU	New Jersey
MANE-VU	New York
MANE-VU	Pennsylvania
MANE-VU	Rhode Island
MANE-VU	Vermont
VISTAS	Alabama
VISTAS	Florida
VISTAS	Georgia
VISTAS	Kentucky
VISTAS	Mississippi
VISTAS	North Carolina
VISTAS	South Carolina
VISTAS	Tennessee
VISTAS	Virginia
VISTAS	West Virginia

Table 3-2 presents SO$_2$ emissions by source sector and RPO for the eastern United States. The NO$_X$ emissions by source sector and RPO are presented in Table 3-3 and VOC emissions in Table 3-4. Regionally, SO$_2$ emissions are more important with respect to regional particle formation and transport. NO$_X$ emissions play an important role in determining the equilibrium between ammonium sulfate and ammonium nitrate formation, especially during winter. VOC emissions contribute to secondary organic aerosol formation.

Table 3-2. SO$_2$ emissions in eastern RPOs (tons/yr)

RPO	Point	Area	On-road	Non-road	Total
MWRPO	3,336,967	133,415	49,191	82,307	3,601,880
MANE-VU	1,924,573	353,176	39,368	74,566	2,391,683
VISTAS	4,349,437	448,023	83,001	91,307	4,971,769

Table 3-3. NO$_X$ emissions in eastern RPOs (tons/yr)

RPO	Point	Area	On-road	Non-road	Total
MWRPO	1,437,284	184,790	1,290,178	723,844	3,636,096
MANE-VU	680,975	268,997	1,297,357	534,454	2,781,783
VISTAS	2,094,228	266,848	2,160,601	812,615	5,334,293

Table 3-4. VOC emissions in eastern RPOs (tons/yr)

RPO	Point	Area	On-road	Non-road	Total
MWRPO	234,938	1,182,186	660,010	492,027	2,569,160
MANE-VU	93,691	1,798,158	793,541	494,115	3,179,504
VISTAS	458,740	2,047,359	1,314,979	609,539	4,430,617

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4. WHAT WILL IT TAKE TO CLEAN THE AIR?

In this chapter we build on the conceptual description of fine particle formation and impacts in the MANE-VU region by looking at a typical fine particle pollution event and the meteorological and chemical conditions which contributed to its formation. As an illustration of how the conceptual elements laid out in Chapter 2 and 3 contribute to a pollution event under real-world circumstances, we examine a pollution event from 2002. We examine this event from two perspectives: (1) the broad spatial patterns of the formation and transport of particle air pollution and (2) the chronological sequence of events at a few discrete points where high temporal resolution monitoring was in place. We then proceed to examine likely emission reduction strategies that should be considered in light of the conceptual understanding of fine particle formation and transport developed in this report.

4.1. Meteorological and Pollution Overview of August 8-16, 2002

Annual and seasonal statistics are useful for understanding the general patterns of air pollution in our region, but it is also instructive to review specific high $PM_{2.5}$ episodes in order to shed more light on the meteorological circumstances under which high ambient concentrations of $PM_{2.5}$ are able to form from emitted precursor pollutants. Here we present an analysis of the high $PM_{2.5}$ and regional haze episode of August 2002 by reviewing surface maps from the period to provide a synoptic overview of major weather systems that were influencing air quality across the Northeast U.S. during that time. Aircraft measurements on August 14 indicated that source regions in the Midwest and Mid-Atlantic urban corridor contributed to the observed pollution, with southerly transport up the urban corridor augmented by the Appalachian lee trough and nocturnal low-level jet (Taubman *et al*., 2004).

Figure 4-1 through Figure 4-3, respectively, show eight-panel displays of afternoon fine particle concentrations as well as surface weather maps and back trajectories from 12Z (8 a.m. EDT) each day. The following chronology of events combines the meteorological insights with $PM_{2.5}$ concentration information to provide a basic storyline for analysis.

A slow-moving high pressure system centered over the Great Lakes set up northerly flow over MANE-VU on August 8. The high drifted southeast-ward and became extended over several days bringing high temperatures to the region. Calm conditions west of MANE-VU on August 10 were pivotal in the formation of fine aerosol concentrations, which began building in the Ohio River Valley. Over the next four days, concentrations in MANE-VU climbed into the 60-90 µg/m^3 range over a wide area before being swept out to sea by a series of frontal passages beginning on August 15.

8/8 – A high pressure system over the Great Lakes produces NW-N prevailing surface winds (~4-8 mph) throughout the region. Maximum daily temperatures approach or exceed 80º F.

8/9 – Wind speeds fall off but direction remains NW-N as the high moves into the central portion of MANE-VU. Temperatures rise as cloud cover declines.

8/10 – The high reaches the East Coast and stalls. Temperatures (except in northern-most areas) reach 90º F while surface-level winds turn to more southerly directions. Calm conditions through the morning hours in the lower Ohio River Valley promote creation of haze noted in surface observations.

8/11 – Circulation around the high (now near Cape Hatteras) becomes well established. Peak temperatures are in the low to mid-90's. Morning winds are light-to-calm in the area east of the Mississippi – the area of haze now reaches from Michigan to northern Texas and eastward to West Virginia and eastern Tennessee. A surface-level trough descends from north of the Great Lakes during the day, passes eastward through the Ohio River Valley and stalls over the Allegheny Mountains and southward.

8/12 – Temperatures exceed 90º F throughout MANE-VU except in coastal ME. The area of concentrated haze has pushed eastward and now extends from central ME to central PA. Haze builds throughout the day as circulation forces it to channel NE between the stalled trough and a cold front approaching from the Midwest.

8/13 – Calm conditions prevail as the trough reaches coastal NJ by 8 a.m. Generally clear skies allow temperatures to reach the mid-90's everywhere except in coastal ME. Dew points, which had been rising since 8/8, reach the upper 60's. Peak hourly fine aerosol concentrations are greater than 40 µg/m³ everywhere in MANE-VU and exceed 90 µg/m³ in some locations. By 8 p.m., showers associated with the approaching cold front have reached into Ohio.

8/14 – By 8 a.m. the trough has dissipated and the high is moving offshore. Dew points remain in the upper 60's and peak temperatures reach into the 90's everywhere and top 100 in several locations. Increased ventilation causes aerosol concentrations to drop throughout the day everywhere except ME where some locations peak above 60 µg/m³ after midnight.

8/15 – The approaching cold front and associated showers fall apart during the morning hours. By 8 p.m., a new batch of moderate rain has intruded deeply into the region from the SW and has virtually pushed the haze out of the MANE-VU region.

8/16 – A new high building in over the upper Midwest pushes the remains of the showers out of the Northeast.

Figure 4-1. Spatially interpolated maps of fine particle concentrations
August 9 – 16, 2002
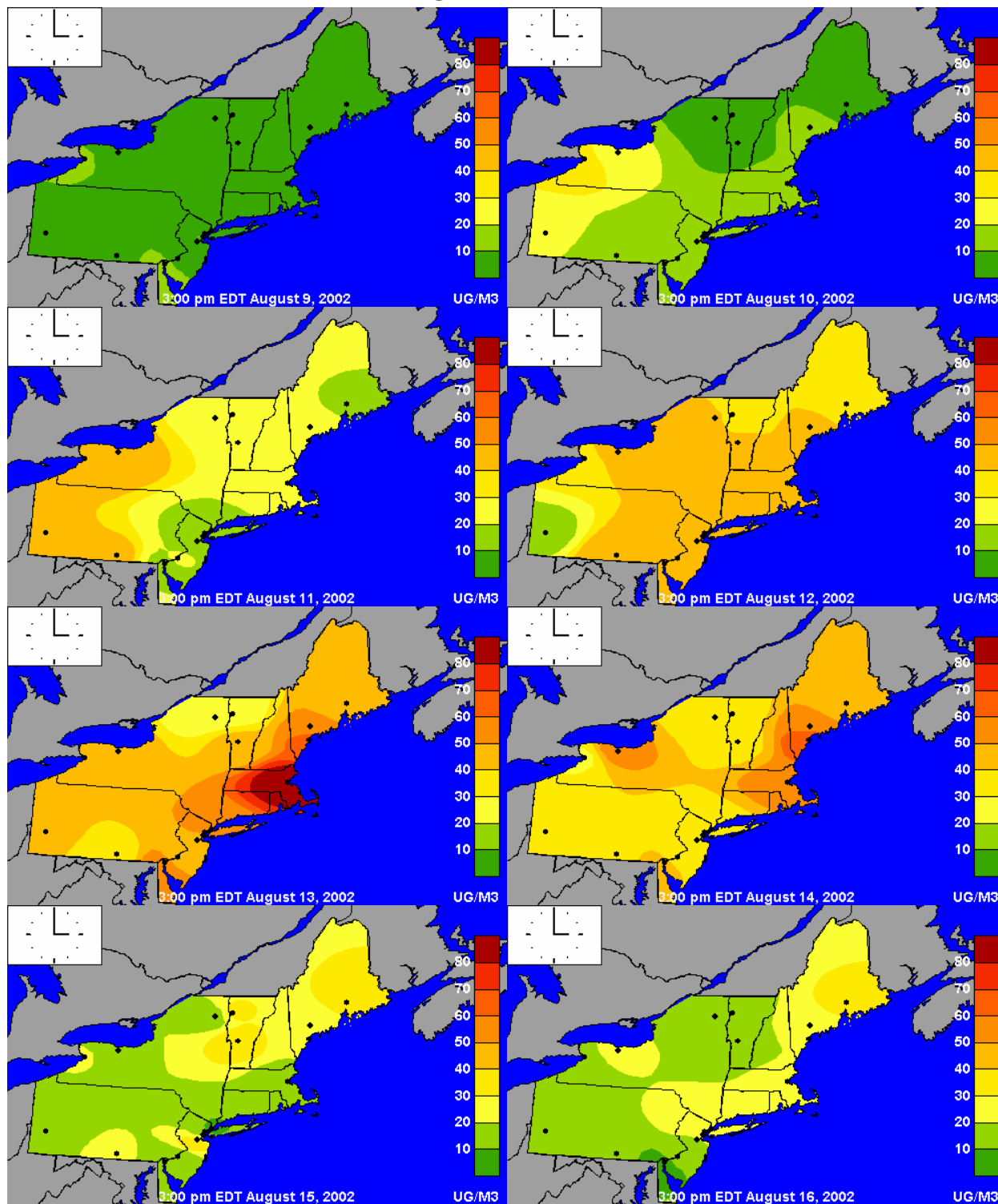
Figure 4-2. Surface weather maps for August 9-16, 2002

August 9, 8:00AM EDT

August 10, 8:00 AM EDT

August 11, 8:00 AM EDT

August 12, 8:00 AM EDT

August 13, 8:00 AM EDT

August 14, 8:00 AM EDT

August 15, 8:00 AM EDT

August 16, 8:00 AM EDT
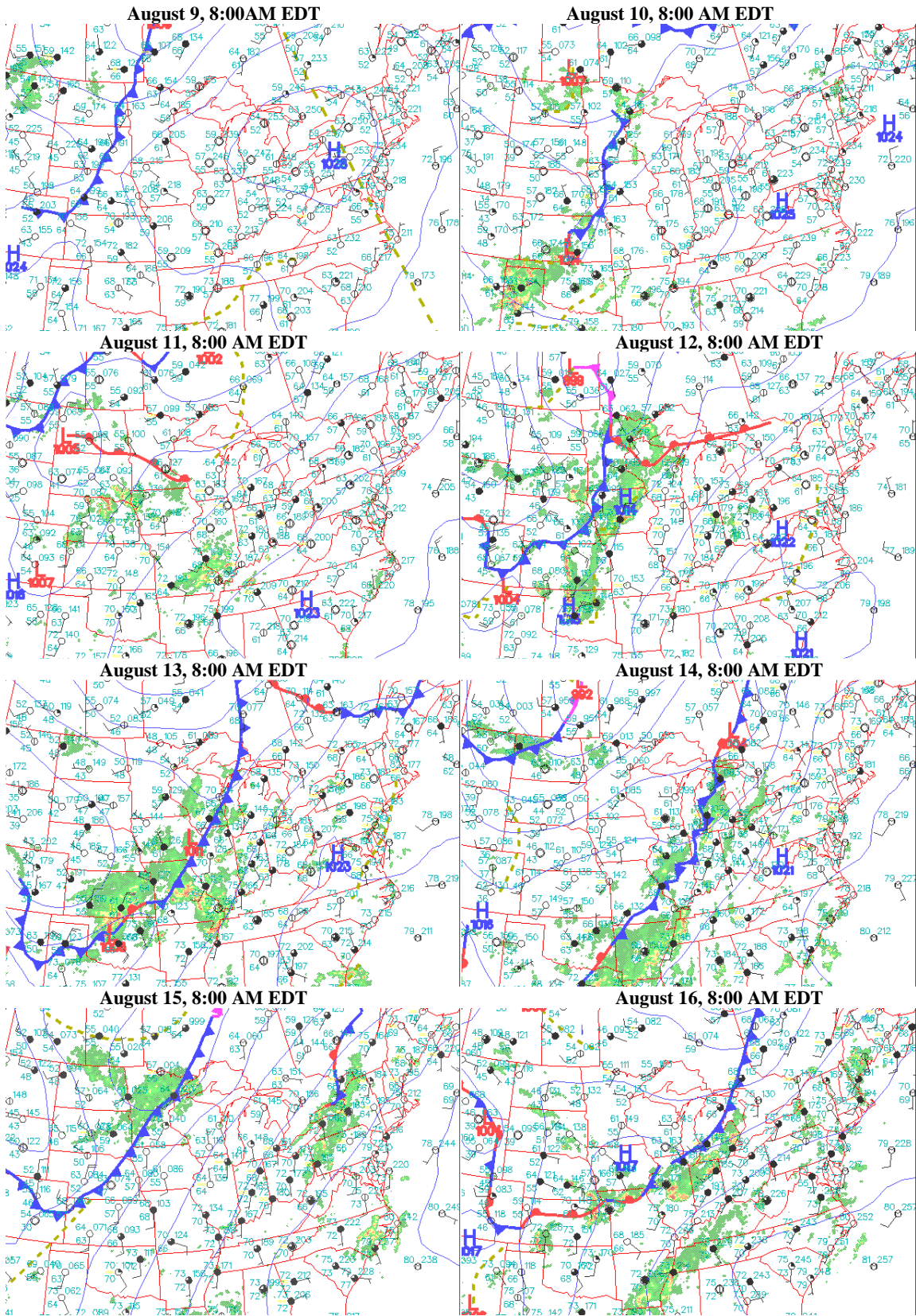
Figure 4-3. HYSPLIT 72-hour back trajectories for August 9-16, 2002

Aug 9, 2002 8 am EDT
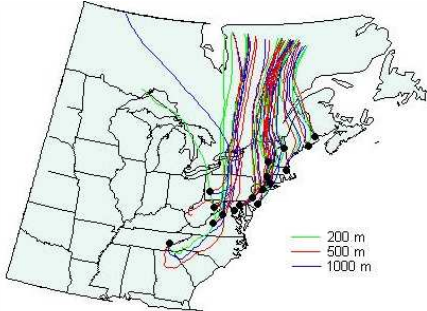
Aug 10, 2002 8 am EDT
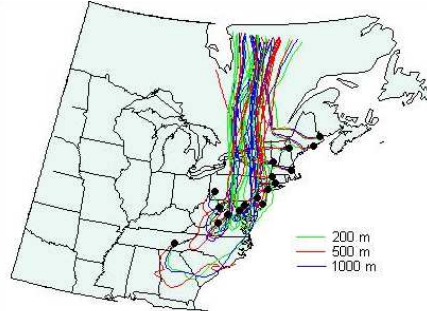
Aug 11, 2002 8 am EDT
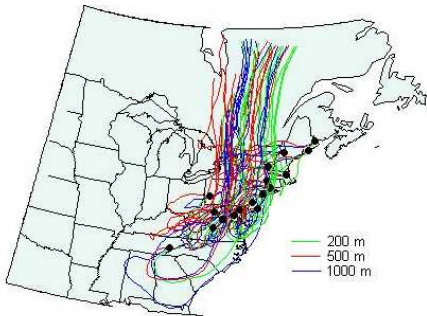
Aug 12, 2002 8 am EDT
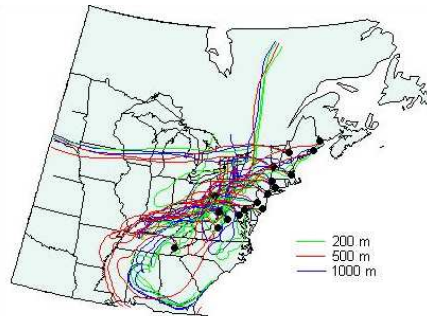
Aug 13, 2002 8 am EDT
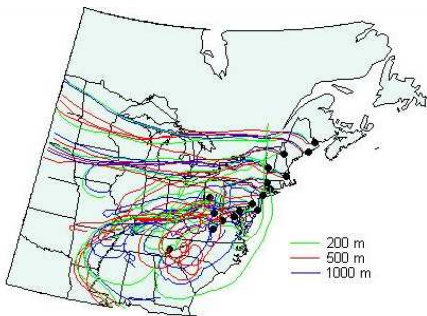
Aug 14, 2002 8 am EDT
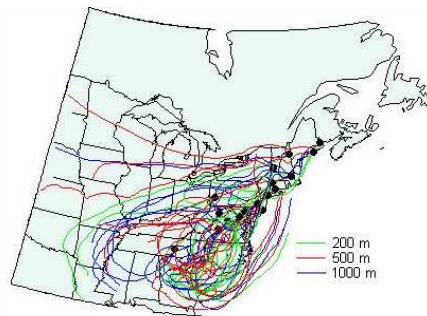
Aug 15, 2002 8 am EDT
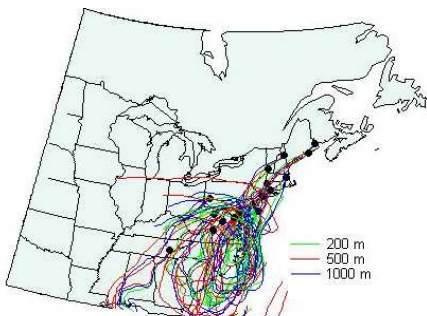
Aug 16, 2002 8 am EDT
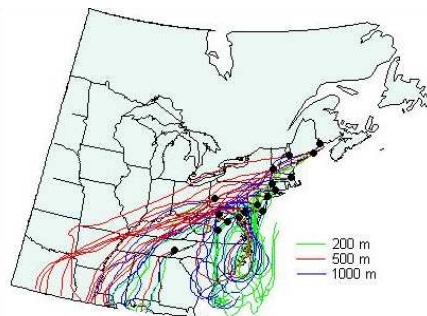
4.2.   Temporally and spatially resolved PM$_{2.5}$ measurements

Higher temporal resolution data provide insight into how the events played out in much more detail than can be captured by eight frames on a page; however the most complete picture is obtained when these high *temporal* resolution data can be presented in the context of the relatively greater *spatial* detail provided by maps such as we have seen in Figure 4-1 through Figure 4-3. In Figure 4-4 and Figure 4-5, we present continuous PM$_{2.5}$ data (hourly average and 24-hour rolling average filtered, respectively) for the August 8-16, 2002 time period.

Figure 4-4. Hourly average fine aerosol at 8 sites during the August 2002 episode
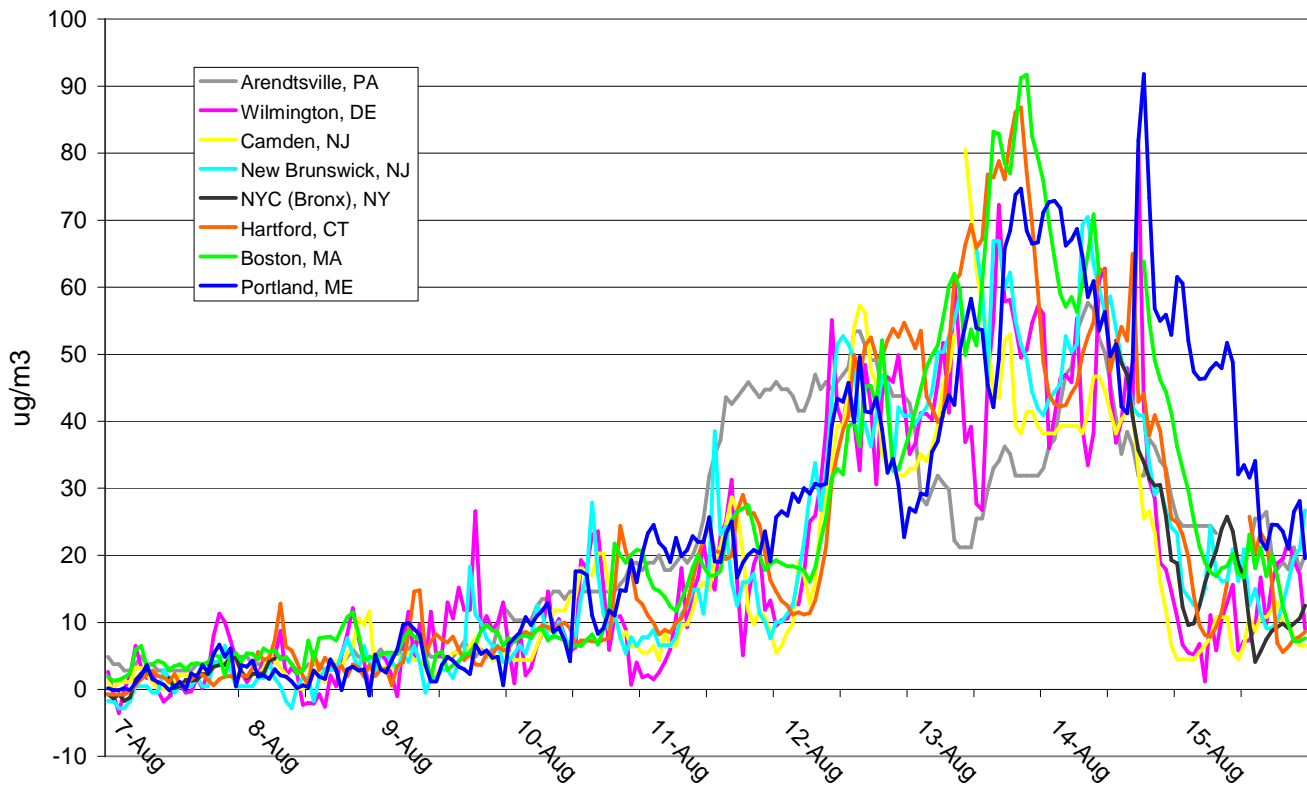
Looking at Figure 4-4 in the context of the maps presented in the earlier figures, it is interesting to note the rapid increase, first, in Arendtsville, PA at noon on the 11th, followed by a rise in concentrations along the East Coast around noon on the 12th. This is consistent with Figure 4-1, which shows high PM$_{2.5}$ levels covering western Pennsylvania by 3 p.m. on the 11th and that high PM$_{2.5}$ area has moved over to cover the East Coast by 3 p.m. the next day. This also makes sense with respect to Figure 4-2 and Figure 4-3, which show the high pressure system established on the East Coast by the 11th with surface level back trajectories having shifted from northerly flow to slow southwesterly flow in the western portion of the domain by the morning of the 11th and the coastal sites having switched by the morning of the 12th.

**Figure 4-5. 24-hour rolling average fine aerosol at
8 MANE-VU sites during the August 2002 episode**
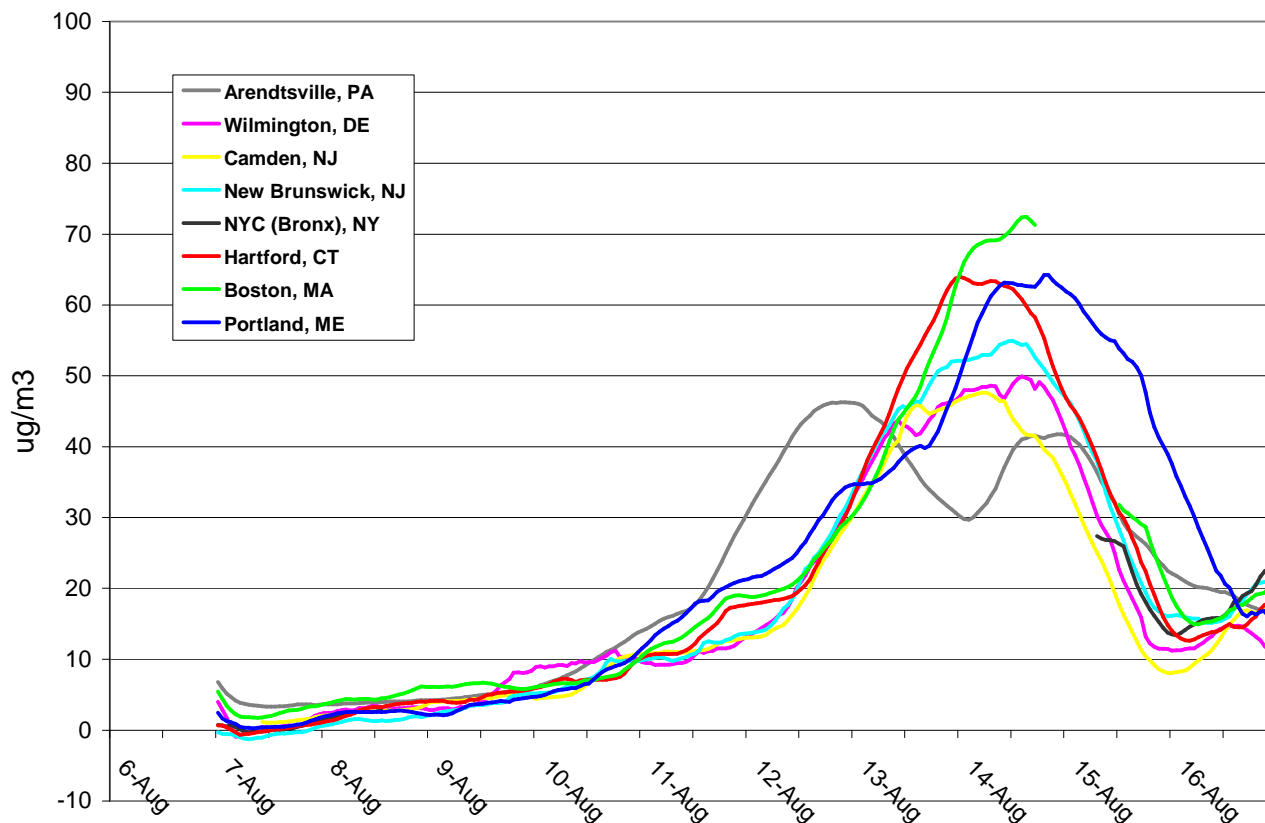
Also note the very high levels observed close to mid-day on the 13th at sites between New York City and Portland, Maine. This is consistent with the strong gradients shown for 3 p.m. on the 13th in Figure 4-1. These rapid increases in concentration are easily explained by the back trajectories of Figure 4-3 that show the advancing front (at this point over Lake Michigan) beginning to push, at upper levels of the atmosphere, an air mass from the upper Midwest due east across the northern half of MANE-VU. At lower levels (see 200 meter trajectories), it can be seen that closer to the surface, this air mass had spent the previous three to four days winding around the Tennessee and Ohio River Valleys before it was driven into the northern reaches of MANE-VU at the peak of the pollution event.

The following figures bring much of this information together in a single image. Figure 4-6 contains satellite photos from MODIS, a mosaic of two consecutive satellite passages on August 13, 2002 from NASA's TERRA satellite. Figure 4-7 shows the same image with geo-referenced activity data and inventory information layered on top to allow for simultaneous depiction of cities, roads, point source emissions, and back trajectories that play a role in the air pollution/haze that affected a large part of the Northeast during this episode.

**Figure 4-6. Composite images from NASA's TERRA Satellite on
August 13, 2002 showing fine particle pollution/haze**
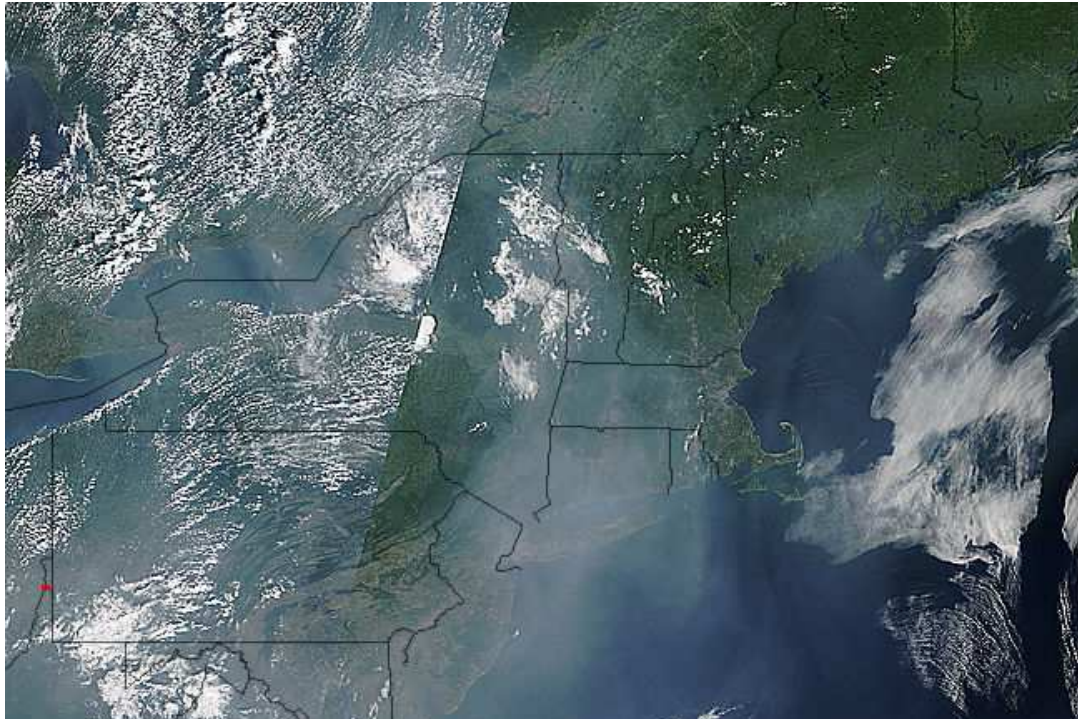
Note the milky/gray haze due to particle pollution as distinct from the puffy white clouds over broad
regions of southern New England and the eastern Mid-Atlantic region.

Figure 4-7. NASA MODIS Terra Satellite Image, Back Trajectories and NO$_X$ Inventory
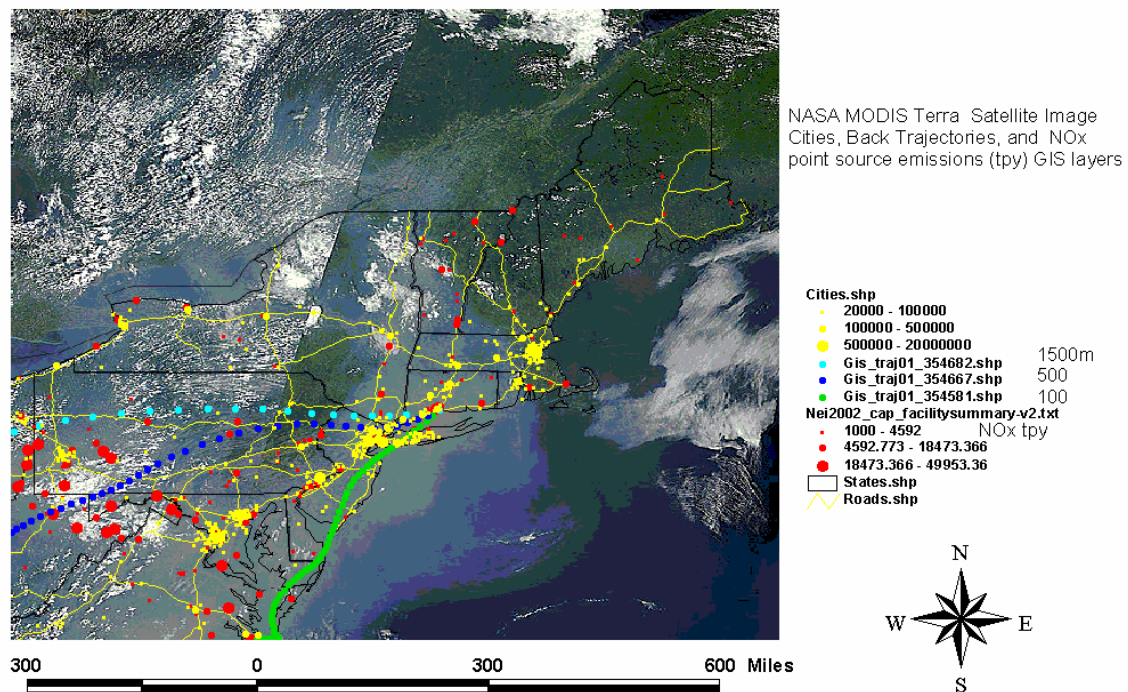
Geo-referenced activity and inventory data (on top of the satellite images presented above) demonstrating
the relationship between observed pollution and upper level winds (driving weather patterns from west to
east), mid-level winds (tracking back to major point sources), and lower level winds (tracking back to major
population centers along the East Coast).

4.3. Implications for control strategies

A 2003 assessment of fine particulate matter by NARSTO[18] states, "[c]urrent air-quality management approaches focusing on reductions of emissions of SO$_2$, NO$_X$, and VOCs are anticipated to be effective first steps towards reducing PM$_{2.5}$ across North America, noting that in parts of California and some eastern urban areas VOC (volatile organic compounds) emissions could be important to nitrate formation."

Sulfate is currently the major contributor to particulate matter and visibility impairment across much of the eastern United States, and the historical record documents a pronounced decline in particulate sulfate concentrations in this region during the 1990s. A review of several studies by Watson (2002) concluded that SO$_2$ emission reductions have in most cases been accompanied by statistically significant reductions in ambient sulfate concentrations. One study (Husar & Wilson, 1993) shows that regionally averaged light extinction closely tracks regionally averaged SO$_2$ emissions for the eastern United States from 1940 through the mid-1980s. Another study by Malm *et al.* (2002) shows that regionally averaged emissions and ambient concentrations decreased together from 1988 through 1999 over a broad region encompassing the states of Connecticut, Delaware, Illinois, Indiana, Kentucky, Maine, Massachusetts, Maryland, Michigan, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, Virginia, Wisconsin, and West Virginia (Watson, 2002).

The timing of this observed decline suggests that this is linked to reductions in SO$_2$ emissions resulting from controls implemented under the federal Acid Rain Program beginning in the early to mid-1990s. From 1989 to 1998, SO$_2$ emissions in the eastern half of the country – that is, including all states within a region defined by the western borders of Minnesota and Louisiana – declined by about 25 percent. This decline in SO$_2$ emissions correlated with a decline of about 40 percent in average SO$_2$ and sulfate concentrations, as measured at Clean Air States and Trend Networks (CASTNet) monitoring sites in the same region over the same time period. In fact, at prevailing levels of atmospheric SO$_2$ loading, the magnitudes of the emissions and concentration changes were not statistically different.

This finding suggests that regional reductions in SO$_2$ emissions have produced near-proportional reductions of particulate sulfate in the eastern United States (NARSTO, 2003). Reductions since 1990 in precursor SO$_2$ emissions are likely also responsible for a continued decline in median sulfate concentrations in the northeastern United States. Nevertheless, episodes of high ambient sulfate concentrations (with peak levels well above the regional median or average) continue to occur, especially during the summertime when regional transport from the Ohio River Valley is also at its peak. This suggests that further reductions in regional and local SO$_2$ emissions would provide significant further air quality and visibility benefits (NARSTO, 2003). Collectively, these

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[18] NARSTO was formerly an acronym for the "North American Research Strategy for Tropospheric Ozone." More recently, the term NARSTO became simply a wordmark signifying a tri-national, public-private partnership for dealing with multiple features of tropospheric pollution, including ozone and suspended particulate matter. For more information on NARSTO, *see* http://www.cgenv.com/Narsto/.

studies provide strong evidence that regional SO$_2$ reductions have yielded, and will continue to yield, reductions in ambient secondary sulfate levels with subsequent reductions in regional haze and associated light extinction.

For urban areas of the eastern United States, an effective multipollutant management approach may be to combine regional SO$_2$ control efforts aimed at reducing summertime PM$_{2.5}$ concentrations with local SO$_2$ and OC control efforts. Local SO$_2$ reductions would help reduce wintertime PM$_{2.5}$ concentrations, while OC reductions can help reduce overall PM$_{2.5}$ concentrations year-round. For areas with high wintertime PM$_{2.5}$ levels, strategies that involve NO$_X$ reductions may also be effective (NARSTO, 2003). Katzman *et al*. (2010) observed that in Midwest urban areas at more northern latitudes, wintertime PM$_{2.5}$ exceedances were driven by nitrates, therefore winter NOx controls on mobile and stationary sources, along with SO$_2$, OC and NH$_3$ controls, should be considered to address wintertime PM$_{2.5}$ levels. Tsimpidi *et al.* (2008) concluded from a modeling study that a coupled decrease in SO$_2$ and NO$_X$ emissions in winter and summer was more effective at lowering total PM$_{2.5}$ mass than SO$_2$ reductions alone.

Emission reductions in local and regional NO$_X$ and primary carbonaceous matter may also lead to reductions in secondary organic aerosols previously thought to be solely biogenic in origin, thus considered not controllable. Modeling and experimental studies indicate that anthropogenic NO$_X$ and primary carbonaceous emissions can affect formation of biogenic secondary organic aerosols through various mechanisms (Carlton *et al*., 2010; Zaveri *et al.*, 2010; Hogrefe *et al.*, 2011; Perraud, *et al.*, 2012).

An analysis of monitoring data from the IMPROVE network suggests that local and regional multipollutant measures in the eastern United States are leading to downward trends across most components of visibility-impairing aerosols at MANE-VU Class I sites. Decreasing trends were seen in sulfate, nitrate, and total carbon concentrations across all MANE-VU sites when analyzing IMPROVE data over periods as long as twenty years (1989-2008) (Hand *et al.*, 2011). The long-term trends in nitrate and sulfate were also seen in a multi-year (1988-2005) study of modeled wet and dry deposition compared with observations in the northeastern United States. These trends were attributed to a number of SO$_2$ and NO$_X$ reduction programs during the study period, including the Clean Air Act Acid Rain Program, the OTC NO$_X$ Budget Program, the NO$_X$ SIP Call, and mobile source measures (Civerolo *et al.*, 2010).

Watson (2002) notes that during the 65 years in which the regional haze program aims to reach its final visibility goals, several opportunities will arise through the decadal SIP cycle to revise the basic control approach. This enables new scientific results to continue to exert a positive influence as states implement new regulatory control programs for multiple pollutants, and as ambient concentrations of these pollutants change relative to each other and relative to ambient ammonia levels. As these relationships between species change, atmospheric chemistry may dictate a revised control approach to those previously described. Further research on these issues should be a priority for supporting 2018 SIP submissions. They include the possibility that:

- Reduction of sulfate in a fully neutralized atmosphere (excess ammonia) could encourage ammonium nitrate formation.

- Ever-greater emissions reductions could be required to produce a given level of improvement in ambient pollutant concentrations because of non-linearities in the atmospheric formation of sulfate.

- Changes in ambient conditions favoring the aqueous oxidation of sulfate (this pathway largely accounts for the non-linearity noted above) may have implications for future emissions control programs. Causes of changing ambient conditions could include, for example, climate change.

West *et al.* (1999) examined a scenario for the eastern United States where $PM_{2.5}$ mass decreases linearly with ammonium sulfate until the latter is fully neutralized by ammonia. Further reductions would free ammonia for combination with gaseous nitric acid that, in turn, would slightly increase $PM_{2.5}$ until all of the nitric acid is neutralized and further sulfate reductions are reflected in lower $PM_{2.5}$ mass. This is an extreme case that is more relevant to source areas (e.g., Ohio) where nitric acid (HNO_3) is more abundant than in areas with lower emissions (e.g., Vermont) (Watson, 2002).

In most situations with non-neutralized sulfate (typical of the eastern United States), ammonia is a limiting agent for the formation of nitrate but will not make any difference until sulfate is reduced to the point where it is completely neutralized. At that point, identifying large sources of ammonia emissions will be important. This point is likely to be many years in the future, however (Watson, 2002).

Based on analyses using the Community Multi-Scale Air Quality (CMAQ) model, the aqueous phase production of sulfate in the Northeast appears to be very oxidant limited and hence non-linear. Thus, conditions that are conducive to a dominance of the gas-phase production pathway drive the summer peaks in ambient sulfate levels. Nonetheless, the expected reduction in ambient sulfate levels resulting from a given reduction in SO_2 emissions is less than proportional overall due to the non-linearity introduced by the aqueous pathway for sulfate formation (NARSTO, 2003). These non-linearity effects are more pronounced for haze than for sulfate deposition, especially at higher sulfate air concentrations (USNPS, 2003).

Finally, we note that because visibility in the clearest areas is sensitive to even minute increases in particle concentrations, strategies to preserve visibility on the clearest days may require stringent limits on emissions growth. In this context, even the dilute emissions from distant sources can be important (NARSTO, 2003). International transport from sources in Canada, Mexico, and Asia may contribute to background sulfate and nitrate in the eastern U.S., so that achieving natural visibility conditions may ultimately require international measures (Park *et al.*, 2004).

4.4. Future $PM_{2.5}$ standards

While the OTR states have made demonstrable progress towards meeting the current $PM_{2.5}$ NAAQS, evolving understanding of fine particles' impacts on human health and welfare has led to the need for a newer, more protective set of standards. In 2005, the Clean Air Scientific Advisory Committee (CASAC) – an independent review committee of expert scientists – made a consensus recommendation to the USEPA to revise the annual primary $PM_{2.5}$ standard from the existing 15 µg/m^3 level down to a level

within a range of 14 to 13 µg/m^3 in concert with strengthening the 24-hour NAAQS from 65 µg/m^3 to within the range of 35 to 30 µg/m^3, 98th percentile form (CASAC, 2005). The USEPA revised the PM$_{2.5}$ NAAQS in 2006 (71 Fed. Reg. 61144) by lowering the 24-hour primary NAAQS to 35 µg/m^3, 98th percentile form, which was at the upper end of the CASAC consensus recommendation. The USEPA, however, retained the annual primary NAAQS at 15 µg/m^3, outside and above the CASAC-recommended range. Legal challenges against the USEPA's decision were filed, resulting in a court ruling in 2009 to remand (without vacating) the annual health standard back to the USEPA for reconsideration on the basis that the USEPA failed to adequately explain its basis for retaining rather than revising the annual standard.[19] The USEPA was to respond to this and other aspects of the court's decision during its next 5-year review of the PM$_{2.5}$ NAAQS,[20] however, the agency missed the deadline of October 17, 2011 for completing its required 5-year review. As the result of a federal district court order to compel the USEPA to act,[21] the agency agreed to propose revised standards by June 14, 2012, and issue final standards by December 14, 2012.

4.5. Future climate change and PM$_{2.5}$/regional haze

In a review of studies projecting the impact of climate change on future air quality in the eastern U.S., Jacob and Winner (2009) noted that the effect of climate change on particulate matter was more complicated and uncertain than for ozone. Important factors are future precipitation frequency and mixing depth, but model projections for these are often unreliable.

Dawson *et al*. (2009) studied the sensitivity of PM$_{2.5}$ to changes in a suite of meteorological variables in the eastern U.S. while holding biogenic and anthropogenic precursor emissions constant. Their results for the portion of the eastern U.S. encompassing the MANE-VU region projected no statistically significant changes in PM$_{2.5}$ during future Januaries and Julies *circa* 2050.

In a modeling study by Pye *et al*. (2009), sulfate aerosol levels due to climate change alone were projected to increase by about 0.3 µg/m^3 on an annual basis in the Northeast by *circa* 2050, with nitrate aerosols projected to decrease region wide by up to 0.24 µg/m^3 due in part to greater thermal decomposition of ammonium nitrate with higher future temperatures. According to the model results, present-day emissions and climate change in the Northeast would result in degraded air quality. Accounting for future domestic emission reductions in SO$_2$ and NO$_X$ reduces the amount of projected air quality degradation. The combined effect of SO$_2$ reductions and climate change is predicted to result in sulfate aerosol reductions across the eastern U.S., including MANE-VU, by up to 3.2 µg/m^3 on an annual basis *circa* 2050. In contrast, nitrate aerosols were projected to increase in much of the OTR, particularly during the colder months, by about 1-2 µg/m^3, depending on the season and location. The reasons given for the projected

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[19] *American Farm Bureau Federation v. EPA*, 559 F. 3d 512 (D.C. Cir. 2009).

[20] In addition to remanding the health-based annual PM$_{2.5}$ NAAQS, the D.C. Circuit also remanded the secondary PM$_{2.5}$ NAAQS intended to protect urban visibility as a welfare value, and originally set at the same levels as the health-based standards (24-hour and annual).

[21] *American Lung Association et al. v. EPA*, U.S. Dist. Ct. D.C., Civil Action No. 1:12-cv-00243-RLW (June 1, 2012).

increase in nitrate aerosols are the decrease in sulfate that would have efficiently competed for the available ammonia along with more total ammonia in the atmosphere due to a projected increase in ammonia emissions.

A modeling study by Tagaris *et al*. (2007) predicted that the impact of climate change alone on regional PM$_{2.5}$ levels *circa* 2050 was small compared to the impacts of emission controls. A mean annual reduction of 23 percent in PM$_{2.5}$ *circa* 2050 was predicted to occur as the result of major reductions in sulfate, nitrate, and ammonium components. More limited reductions in organic carbon suggested that organic carbon would become the dominant PM$_{2.5}$ mass component in the future.

4.6. Conclusion: Simplifying a complex problem

A conceptual understanding of fine particles from a regional perspective across MANE-VU and throughout the eastern U.S. is well understood, yet remains complex due to the multiplicity of source regions (both regional and local), pollutants (SO$_2$, NO$_X$, organic carbon, and primary PM$_{2.5}$), and seasons (summer and winter) that are involved in fine particle formation.

Regional approaches to the control of precursor SO$_2$ and NO$_X$ emissions have been started through Title IV of the Clean Air Act, the NO$_X$ SIP Call, the CAIR, and the establishment and support of Regional Planning Organizations to assist with Regional Haze Rule compliance. With the modeling foundation developed for the CAIR program, the USEPA has presented a compelling technical case on the need for additional regional SO$_2$ and NO$_X$ reductions in the eastern U.S. to reduce particulate levels and protect public health. While states in the Northeast disagree with the extent of SO$_2$ and NO$_X$ reductions and the timeline for those reductions to occur, the program is an excellent next step toward reducing fine particles in MANE-VU. It is tempting to suggest that the regional control of SO$_2$ and NO$_X$ addresses the extent of the problem facing MANE-VU, but as the conceptual description contained in this report demonstrates, the reduction of fine particles in the eastern U.S. requires a careful balance of regional and local controls for SO$_2$, NO$_X$, organic carbon, and primary PM$_{2.5}$ over the course of a year.

The (relatively) higher emissions of SO$_2$ and NO$_X$ from regions upwind of MANE-VU as well as the long "reach" of sulfate pollution requires continued regional control of these fine particle precursors. However, local accumulation of SO$_2$-derived sulfate, NO$_X$-derived nitrate, and primary PM (mostly in the form of black carbon/diesel exhaust) can significantly boost urban PM$_{2.5}$ levels. Residential wood combustion in rural river valleys can significantly raise PM levels as well and affect rural visibility in areas near to Class I areas.

The balance between regional and local controls parallels the balance that needs to be achieved between pollutants. The regional contribution to fine particle pollution is driven by sulfates and organic carbon, whereas the local contribution to PM$_{2.5}$ is derived from SO$_2$, NO$_X$, organic carbon, and primary PM$_{2.5}$ (including black carbon/diesel exhaust).

Finally, control strategies which focus on regional SO$_2$ emissions reductions are needed throughout the summer and winter months, suggesting that a year-round approach to control is needed. Urban nonattainment counties with local emissions of NO$_X$ and

VOC will be driven to reduce these emissions during the summer for ozone benefits, but these same pollutants – as well as primary particulate emissions – contribute to high PM$_{2.5}$ levels in winter, suggesting that annual controls for all of these pollutants make sense in a multi-pollutant context. Finally, residential wood smoke near Class I areas is clearly a winter issue, and further controls may be desirable near specific Class I sites where organic carbon is a contributor on the 20 percent worst visibility days that occur in winter months.

To bring attainment to the current fine particle nonattainment counties and meet reasonable progress goals toward national visibility goals, there continues to be a need for more regional SO$_2$ and NO$_X$ reductions coupled with appropriate local SO$_2$, NO$_X$, VOC, and primary PM$_{2.5}$ (including diesel exhaust) controls where local accumulation is shown to add to the regional burden of sulfate and nitrate PM$_{2.5}$ (primarily in winter). These local controls will vary by location and by season, but the regional control of SO$_2$ and NO$_X$ should be maintained on an annual basis given the contribution of regional sulfate and nitrate to fine particle peaks during both summer and winter months. The need for maintaining progress in reducing PM$_{2.5}$ precursor and primary emissions is further reinforced by a potential future strengthening of the PM$_{2.5}$ NAAQS in response to a court order. Potential future climate change is also projected to have an impact, although current emission control measures may help ameliorate the increase to some extent.

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Appendix A: USEPA Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, PM₂.₅, and Regional Haze

From "Guidance on the Use of Models and Other Analyses for Demonstrating Attainment of Air Quality Goals for Ozone, PM₂.₅, and Regional Haze," U.S. Environmental Protection Agency, EPA-454/B-07-002, Chapter 11, April 2007.

APPENDIX A: EPA GUIDANCE DOCUMENT EXCERPT

11.0 How Do I Get Started? - A "Conceptual Description"

A State/Tribe should start developing information to support a modeled attainment demonstration by assembling and reviewing available air quality, emissions and meteorological data. Baseline design values should be calculated at each monitoring site, as described in Section 3. For PM applications, speciated data should be reviewed to get a sense of what component(s) might be contributing most significantly to nonattainment or light extinction. If past modeling has been performed, the emission scenarios examined and air quality predictions may also be useful. Readily available information should be used by a State/Tribe to develop an initial conceptual description of the nonattainment or reasonable haze problem in the area which is the focus of a modeled demonstration. A conceptual description is instrumental for identifying potential stakeholders and for developing a modeling/analysis protocol. It may also influence a State's choice of air quality model, modeling domain, grid cell size, priorities for quality assuring and refining emissions estimates, and the choice of initial diagnostic tests to identify potentially effective control strategies. In general, a conceptual description is useful for helping a State/Tribe identify priorities and allocate resources in performing a modeled demonstration.

In this Section, we identify key parts of a conceptual description. We then present examples of analyses which could be used to describe each of these parts. We note that initial analyses may be complemented later by additional efforts performed by those implementing the protocol.

11.1 What Is A "Conceptual Description"?

A "conceptual description" is a qualitative way of characterizing the nature of an area's nonattainment or regional haze problem. It is best described by identifying key components of a description. Examples are listed below. There are 3 different examples. One each for ozone, annual PM₂.₅, and regional haze. The examples are not necessarily comprehensive. There could be other features of an area's problem which are important in particular cases. For purposes of illustration later in the discussion, we have answered each of the questions posed below. Our responses appear in parentheses.

11.1.1 8-Hour Ozone NAAQS

1. Is the nonattainment problem primarily a local one, or are regional factors important?

   (Surface measurements suggest transport of ozone close to 84 ppb is likely. There are some other nonattainment areas not too far distant.)

2. Are ozone and/or precursor concentrations aloft also high?

   (There are no such measurements.)

3. Do violations of the NAAQS occur at several monitoring sites throughout the nonattainment area, or are they confined to one or a small number of sites in proximity to one another?

   (Violations occur at a limited number of sites, located throughout the area.)

4. Do observed 8-hour daily maximum ozone concentrations exceed 84 ppb frequently or just on a few occasions?

   (This varies among the monitors from 4 times up to 12 times per year.)

5. When 8-hour daily maxima in excess of 84 ppb occur, is there an accompanying characteristic spatial pattern, or is there a variety of spatial patterns?

   (A variety of patterns is seen.)

6. Do monitored violations occur at locations subject to mesoscale wind patterns (e.g., at a coastline) which may differ from the general wind flow?

   (No.)

7. Have there been any recent major changes in emissions of VOC or NO$_X$ in or near the nonattainment area? If so, what changes have occurred?

   (Yes, several local measures [include a list] believed to result in major reductions in VOC [quantify in tons per summer day] have been implemented in the last five years. Additionally, the area has seen large regional NO$_X$ reductions from the NO$_X$ SIP call.)

8. Are there discernible trends in design values or other air quality indicators which have accompanied a change in emissions?

   (Yes, design values have decreased by about 10% at four sites over the past [x] years. Smaller or no reductions are seen at three other sites.)

9. Is there any apparent spatial pattern to the trends in design values?

   (No.)

10. Have ambient precursor concentrations or measured VOC species profiles changed?

(There are no measurements.)

11. What past modeling has been performed and what do the results suggest?

(A regional modeling analysis has been performed. Two emission scenarios were modeled: current emissions and a substantial reduction in NO$_X$ emissions throughout the regional domain. Reduced NO$_X$ emissions led to substantial predicted reductions in 8-hour daily maximum ozone in most locations, but changes near the most populated area in the nonattainment area in question were small or nonexistent.)

12. Are there any distinctive meteorological measurements at the surface or aloft which appear to coincide with occasions with 8-hour daily maxima greater than 84 ppb?

(Other than routine soundings taken twice per day, there are no measurements aloft. There is no obvious correspondence with meteorological measurements other than daily maximum temperatures are always > 85 F on these days.)

Using responses to the preceding questions in this example, it is possible to construct an initial conceptual description of the nonattainment area's ozone problem. First, responses to questions 1 and 11 suggest there is a significant regional component to the area's nonattainment problem. Second, responses to questions 3, 4, 7, 8, and 11 indicate there is an important local component to the area's nonattainment problem. The responses to questions 4, 5 and 12 indicate that high ozone concentrations may be observed under several sets of meteorological conditions. The responses to questions 7, 8, and 11 suggest that ozone in and near the nonattainment area may be responsive to both VOC and NO$_X$ controls and that the extent of this response may vary spatially. The response to question 6 suggests that it may be appropriate to develop a strategy using a model with 12 km grid cells.

The preceding conceptual description implies that the State/Tribe containing the nonattainment area in this example will need to involve stakeholders from other, nearby States/Tribes to develop and implement a modeling/analysis protocol. It also suggests that a nested regional modeling analysis will be needed to address the problem. Further, it may be necessary to model at least several distinctive types of episodes and additional analyses will be needed to select episodes. Finally, sensitivity (i.e., diagnostic) tests, or other modeling probing tools, will be needed to assess the effects of reducing VOC and NO$_X$ emissions separately and at the same time.

11.1.2 Annual PM$_{2.5}$ NAAQS

1. Is the nonattainment problem primarily a local one, or are regional factors important?

(Surface measurements suggest that only design values in or immediately downwind of the city violate the NAAQS. However, other nearby design values come close to the concentration specified in the NAAQS.)

2. What is the relative importance of measured primary and secondary components of $PM_{2.5}$ measured at sites violating the NAAQS?

(Secondary components (i.e., SO_4, NO_3, OC) constitute about 80% of the measured mass of $PM_{2.5}$. There are higher concentrations of primary $PM_{2.5}$ in the core urban area compared to the suburbs and more rural areas.)

3. What are the most prevalent components of measured $PM_{2.5}$?

(The most important components in ranked order are mass associated with SO_4, OC and inorganic primary particulate matter (IP).)

4. Does the measured mix of PM components appear to roughly agree with mix of emission categories surrounding the monitoring sites?

(No. Relative importance of measured crustal material (IP) appears less than what might be inferred from the inventory.)

5. Do there appear to be any areas with large gradients of primary $PM_{2.5}$ in monitored or unmonitored areas?

(Cannot really tell for sources of crustal material until we resolve the preceding inventory/monitoring discrepancy. There are no other obvious major sources of primary particulate matter.)

6. Is there any indication of what precursor might be limiting formation of secondary particulate matter?

(No indicator species analyses have been performed. Past analyses performed for ozone-related SIP revisions suggest that ozone in this area may be limited by availability of VOC.)

7. Do monitored violations occur at locations subject to mesoscale wind patterns (e.g., at a coastline) which may differ from the general wind flow?

(No.)

8. Have there been any recent major changes in emissions of PM or its precursors in or near the nonattainment area? What?

(Yes, measures believed to result in major reductions in VOC and NO_X have been implemented in the last 5 years. Reductions in power plant NO_X have resulted from

the NO$_X$ SIP call and SO$_2$ emissions reductions have resulted from the national program to reduce acid deposition.)

9. Are there discernible trends in design values or other air quality indicators which have accompanied a change in emissions?

(The trend appears to be downward, but the most recent air quality data has been higher. Overall, the period of record is insufficiently long to tell.)

10. Is there any apparent spatial pattern to the trends in design values?

(No.)

11. What past modeling has been performed and what do the results suggest?

(A regional modeling analysis has been performed for ozone and PM$_{2.5}$. Two emission scenarios were modeled: current emissions and a substantial reduction in NO$_X$ and SO$_2$ emissions throughout a regional domain. Reduced NO$_X$ emissions led to substantial predicted reductions in 8-hour daily maximum ozone in most locations. Modeled SO$_2$ reductions from the CAIR rule had a strong impact on sulfate concentrations.)

12. Are there any distinctive meteorological measurements at the surface or aloft which appear to coincide with occasions with PM$_{2.5}$ concentrations in excess of 15.0 µg/m^3?

(Other than routine soundings taken twice per day, there are no measurements aloft. There is no obvious correspondence with meteorological measurements other than daily maximum temperatures are often > 85 F on days with the highest PM$_{2.5}$ observations.)

13. Do periods with high measured particulate matter or components of particulate matter appear to track each other or any other measured pollutant?

(There appears to be some correspondence between measured high concentrations of SO$_4$ and ozone.)

Using responses to the preceding questions in this example, it is possible to construct an initial conceptual description of the nonattainment area's ozone problem. First, responses to questions 1, 2 and 3 suggest there is a significant regional component to the area's nonattainment problem. Second, responses to questions 1 and 3 indicate there is a local component to the problem. The responses to questions 11, 12 and 13 suggest that there may be a link between reducing ozone and reducing particulate matter. Thus, it may be appropriate to assess effects of previously committed to strategies to reduce ozone and national PM control measures before simulating additional control measures. The responses to questions 4 and 5 suggest that it is premature to determine whether a "local

area analysis" will be needed. The response to question 7 suggests that it may not be necessary to model with very small grid cells, at least for the secondary components of PM$_{2.5}$.

The preceding conceptual description implies that the State containing the nonattainment area in this example will need to involve stakeholders from other, nearby States to develop and implement a modeling/analysis protocol. It also suggests that a nested regional modeling analysis will be needed to address the problem.

11.1.3 Example Regional Haze Application

1. What components of particulate matter appear to have high concentrations on days with poor visibility?

   (Mass associated with SO$_4$ and coarse particulate matter (CM) seem to have the highest concentrations on most such days.)

2. What are typical values for the humidity adjustment factor during the times of year when most of the days with poor visibility occur?

   (Typical values appear to be about "4.0".)

3. Does visibility appear to track well among nearby Class I areas?

   (Yes, but not always.)

4. Does poor visibility seem to occur under any specific meteorological conditions?

   (This information is not readily available.)

5. Does poor visibility seem to coincide with high observed concentrations of any particular other pollutant?

   (There seems to be some correspondence with high regional ozone concentrations.)

6. What components of particulate matter appear to have relatively high concentrations on days with good visibility?

   (Coarse particulate matter and OC.)

7. What are typical values for the humidity adjustment factor during times of year when most of the days with good visibility occur?

   (About "2.3".)

8. Does good visibility appear to occur under any specific meteorological conditions?

(Don't know.)

Answers to the preceding questions suggest that strategies to reduce sulfate concentrations and, perhaps, regional ozone concentrations might be effective in reducing light extinction on days when visibility is currently poor. The responses suggest that a strategy which focuses on this alone should first be tried for the days with good visibility as well. Even though sulfate concentrations appear low on such days, the fact that sulfates scatter light efficiently (see Equation (6.1)) and relative humidity is still high enough to enhance this effect is worth considering. Responses suggest that further meteorological analyses would be worthwhile prior to selecting strategies to simulate with a resource intensive regional model.

It should be clear from the preceding examples that the initial conceptual description of an area's nonattainment problem draws on readily available information and need not be detailed. It is intended to help launch development and implementation of a modeling/analysis protocol in a productive direction. It will likely be supplemented by subsequent, more extensive modeling and ambient analyses performed by or for those implementing the modeling/analysis protocol discussed in Section 12.0.

Questions like those posed in Section 11.1 can be addressed using a variety of analyses ranging in complexity from an inspection of air quality data to sophisticated mathematical analyses. We anticipate the simpler analyses will often be used to develop the initial conceptual description. These will be followed by more complex approaches or by approaches requiring more extensive data bases as the need later becomes apparent. These analyses are intended to channel resources available to support modeled attainment demonstrations onto the most productive paths possible. They will also provide other pieces of information which can be used to reinforce conclusions reached with an air quality model, or cause a reassessment of assumptions made previously in applying the model. As noted in Section 7, corroboratory analyses should be used to help assess whether a simulated control strategy is sufficient to meet the NAAQS.

Appendix B: Monitoring Data from Class I sites in MANE-VU

Below are figures presenting baseline monitoring data for the Class I sites (and Washington, DC) based on IMPROVE monitoring network data using the alternative methodology for the reconstructed extinction equation approved by the IMPROVE Steering Committee and adopted by the MANE-VU states. This alternative methodology was used to calculate natural background and baseline visibility conditions as well as tracking progress relative to the derived uniform rate of progress. Graphs were created from data downloaded from the VIEWS website (http://views.cira.colostate.edu/web/).

APPENDIX B: MONITORING DATA FROM CLASS I SITES IN MANE-VU

Figure B-1. Monitoring Data from Acadia National Park, ME

Acadia Deciview (dv) Trends
Worst 20%

Baseline (2000-2004 Average) = 22.89
2004-2008 Average = 22.06
Natural Conditions = 12.43

Acadia Extinction Trends
Best 20% and Worst 20%

Legend:
- SS bext
- SOILf_bext
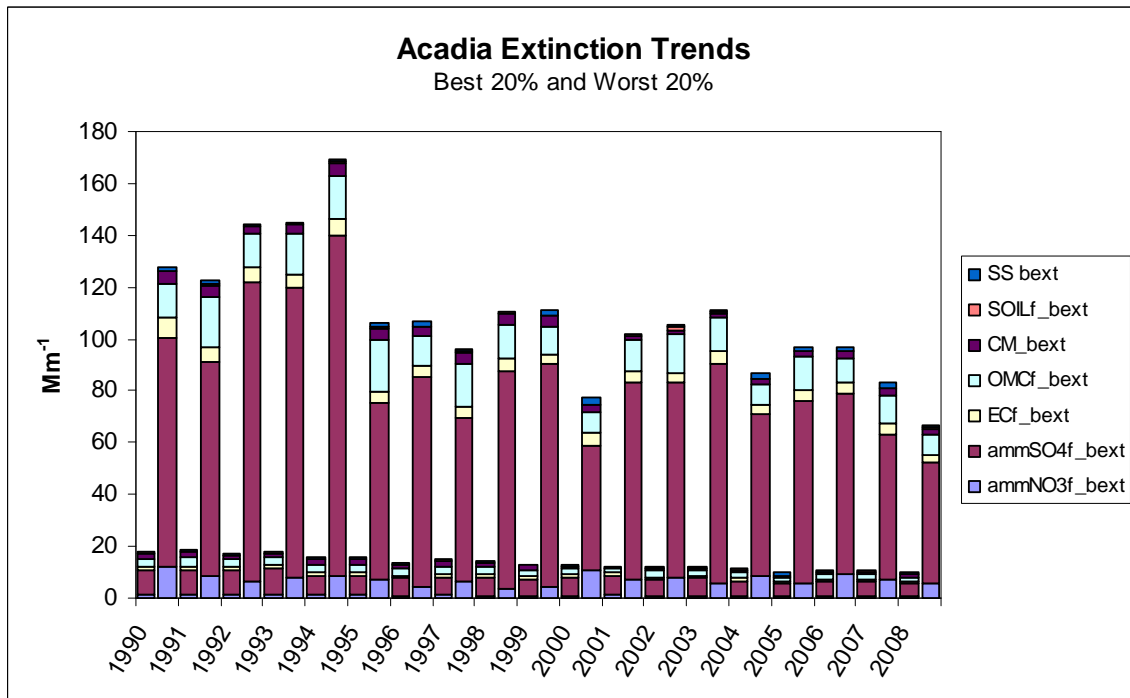- CM_bext
- OMCf_bext
- ECf_bext
- ammSO4f_bext
- ammNO3f_bext
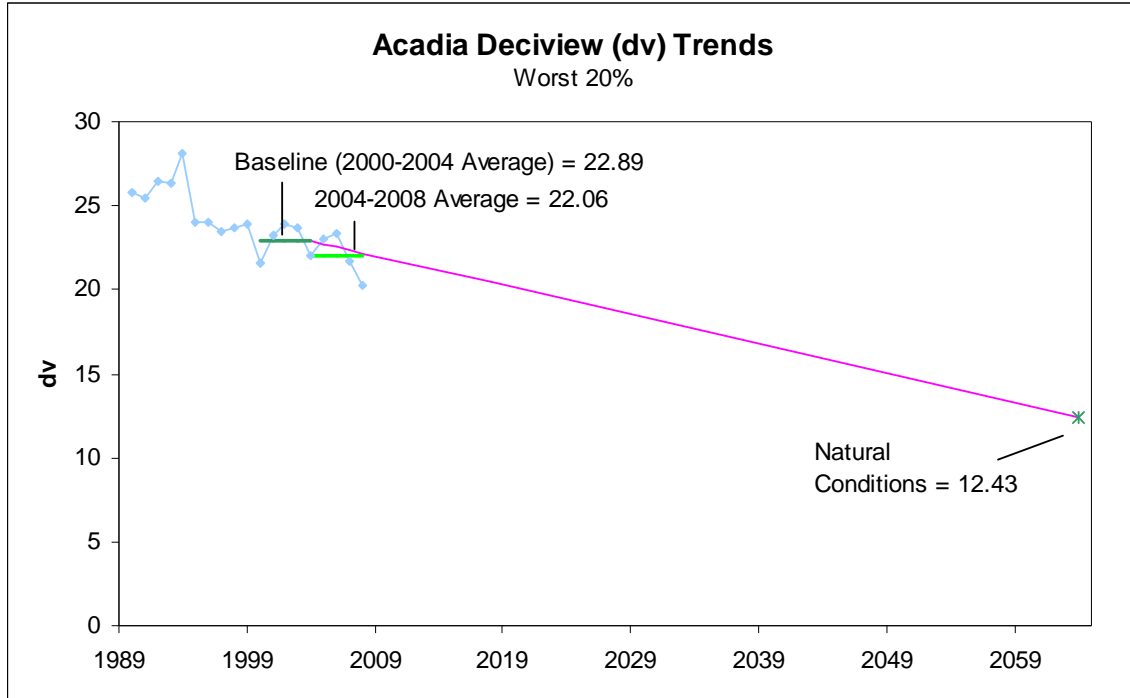
Figure B-2. Monitoring Data from Brigantine, NJ
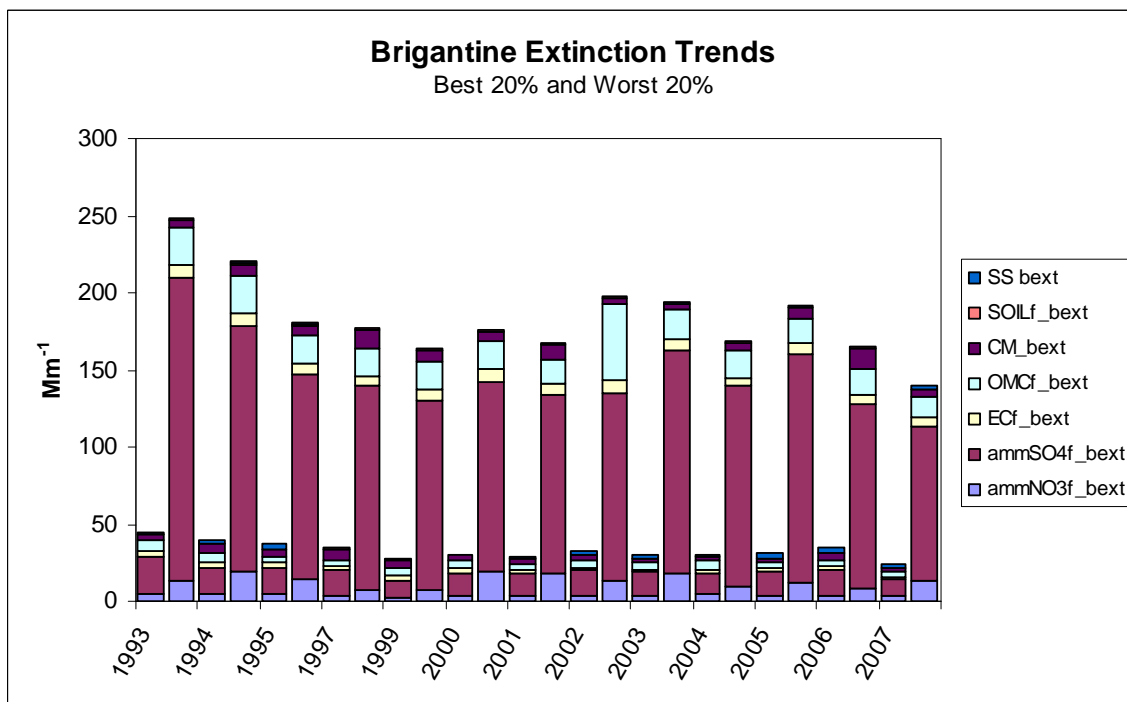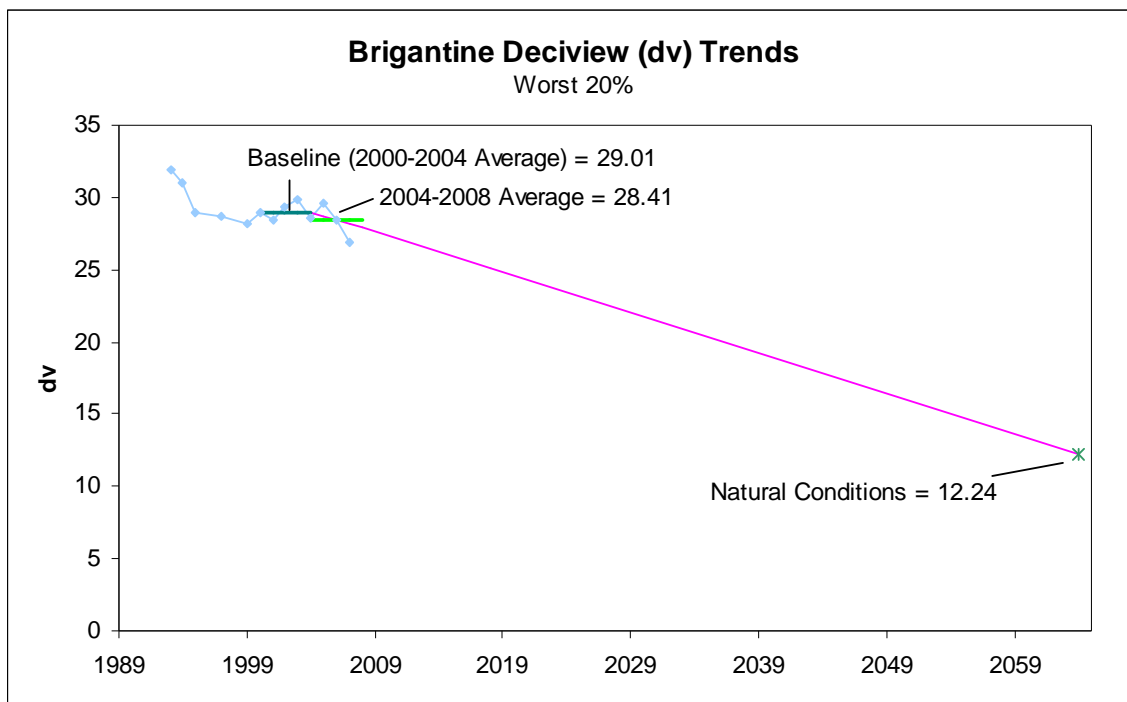
Figure B-3. Monitoring Data from Great Gulf, NH
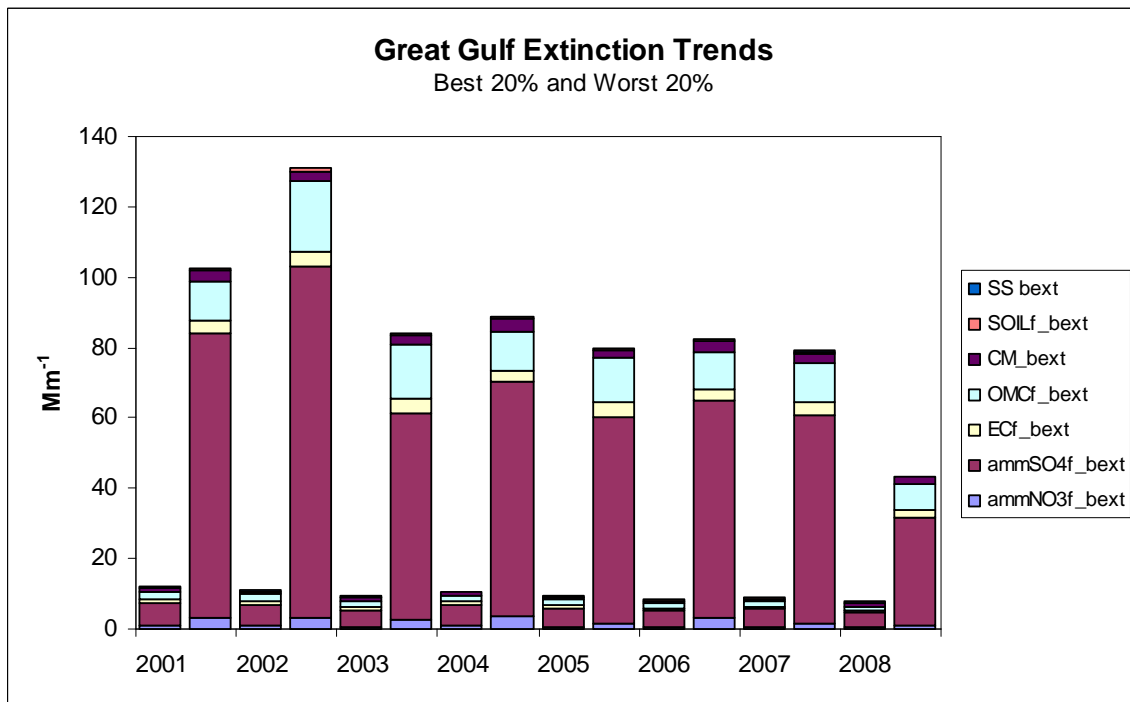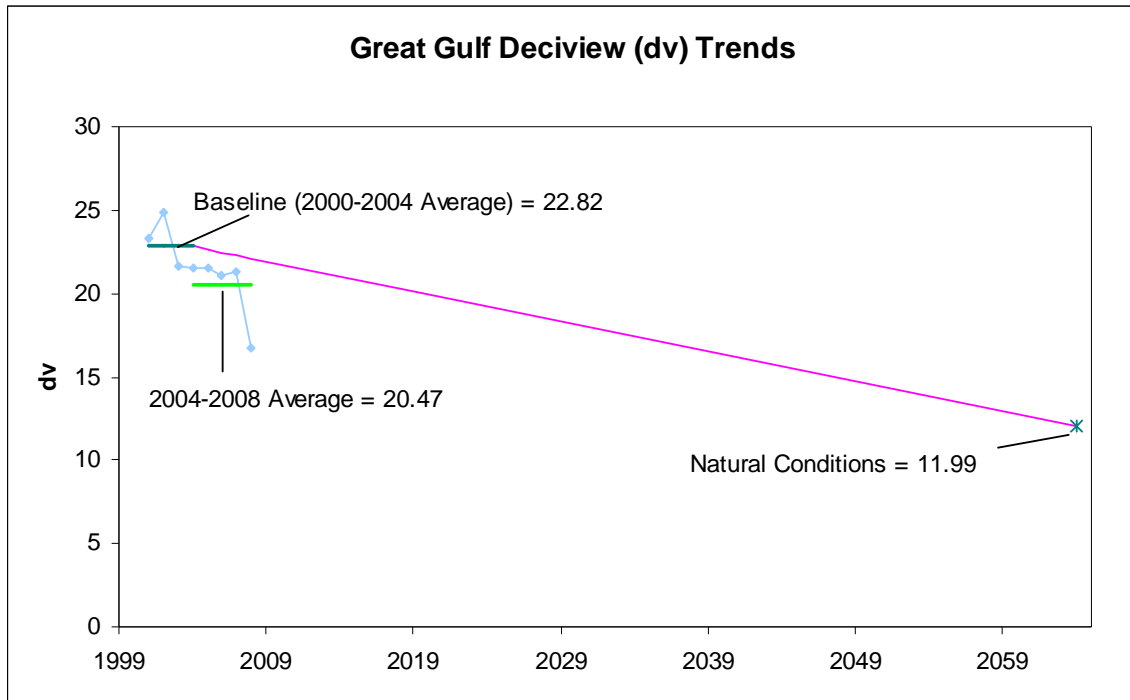
Figure B-4. Monitoring Data from Lye Brook, VT
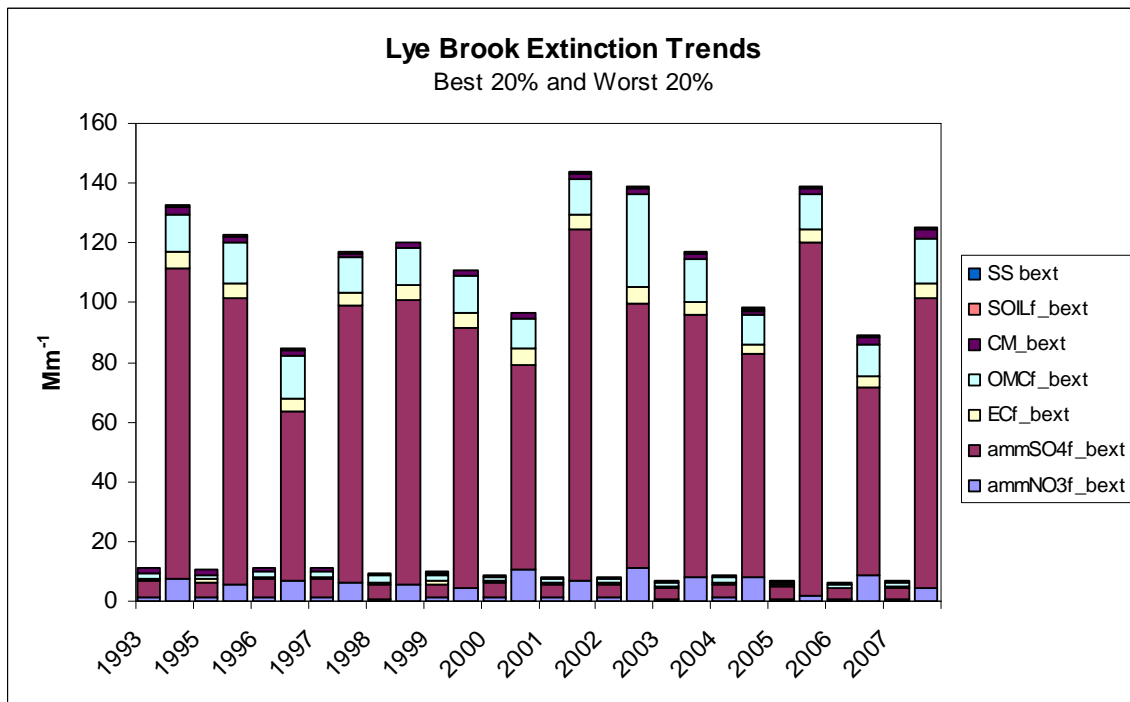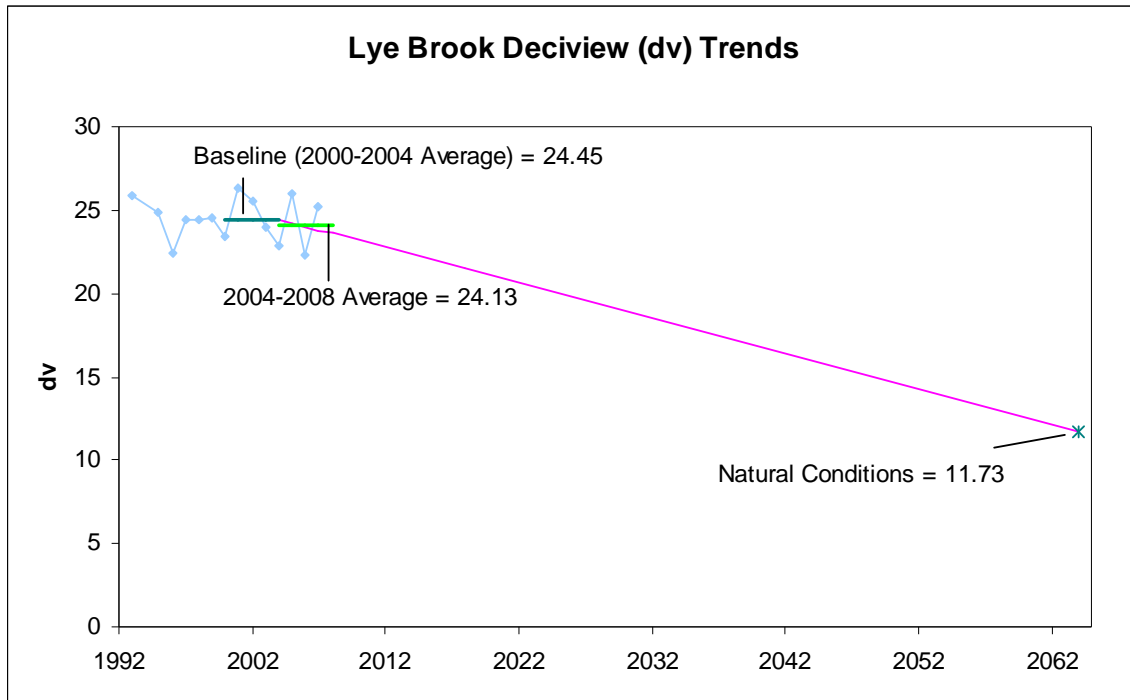
Figure B-5. Monitoring Data from Moosehorn, ME

Moosehorn Deciview (dv) Trends

Baseline (2000-2004 Average) = 21.72

2004-2008 Average = 20.43

Natural Conditions = 12.01

Moosehorn Extinction Trends
Best 20% and Worst 20%

Legend:
- SS bext
- SOILf_bext
- CM_bext
- OMCf_bext
- ECf_bext
- ammSO4f_bext
- ammNO3f_bext
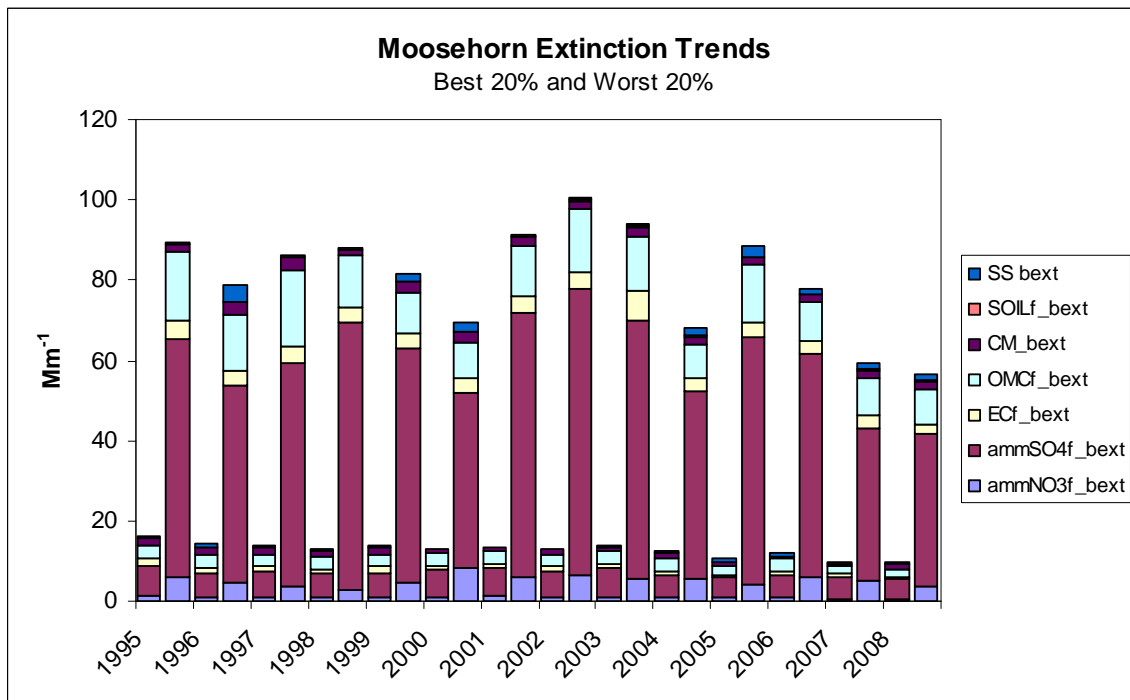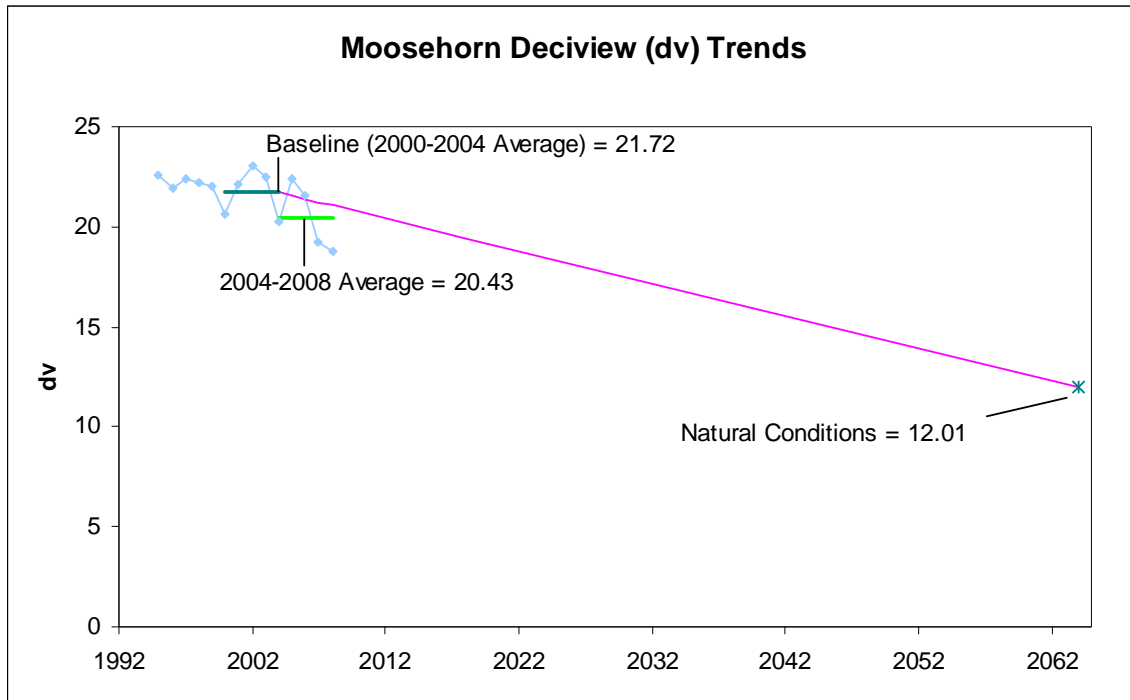
Figure B-6. Monitoring Data from Washington, DC

Washington, D.C. Deciview (dv) Trends

Baseline (2000-2004 Average) = 30.47

2004-2008 Average = 29.70

Natural Conditions = 11.86

Washington, D.C. Extinction Trends
Best 20% and Worst 20%

Legend:
- SS bext
- SOILf_bext
- CM_bext
- OMCf_bext
- ECf_bext
- ammSO4f_bext
- ammNO3f_bext
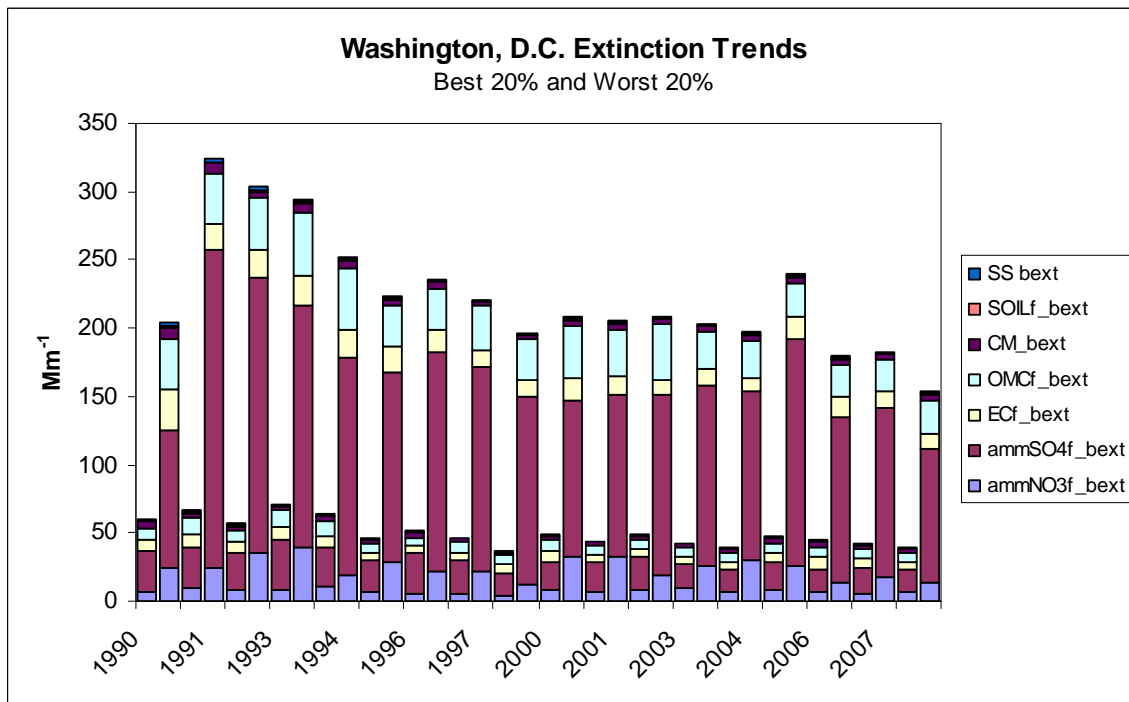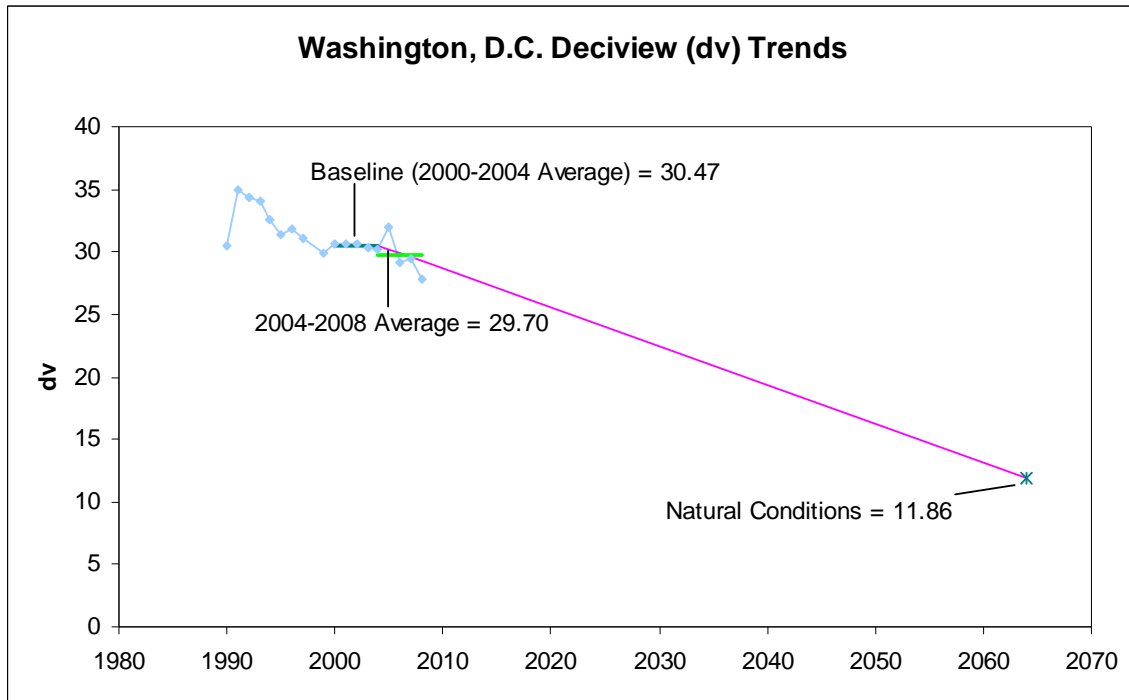
Figure B-7. 20% Worst and Best 2004-2008 Visibility Days at Acadia NP, ME
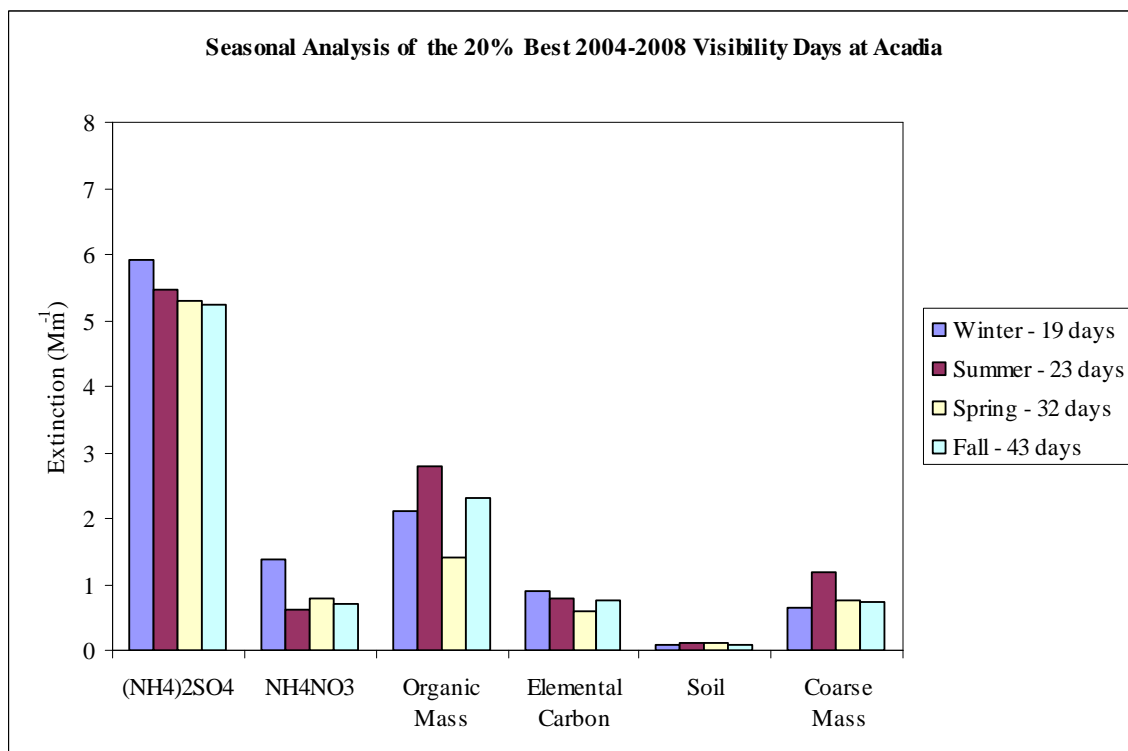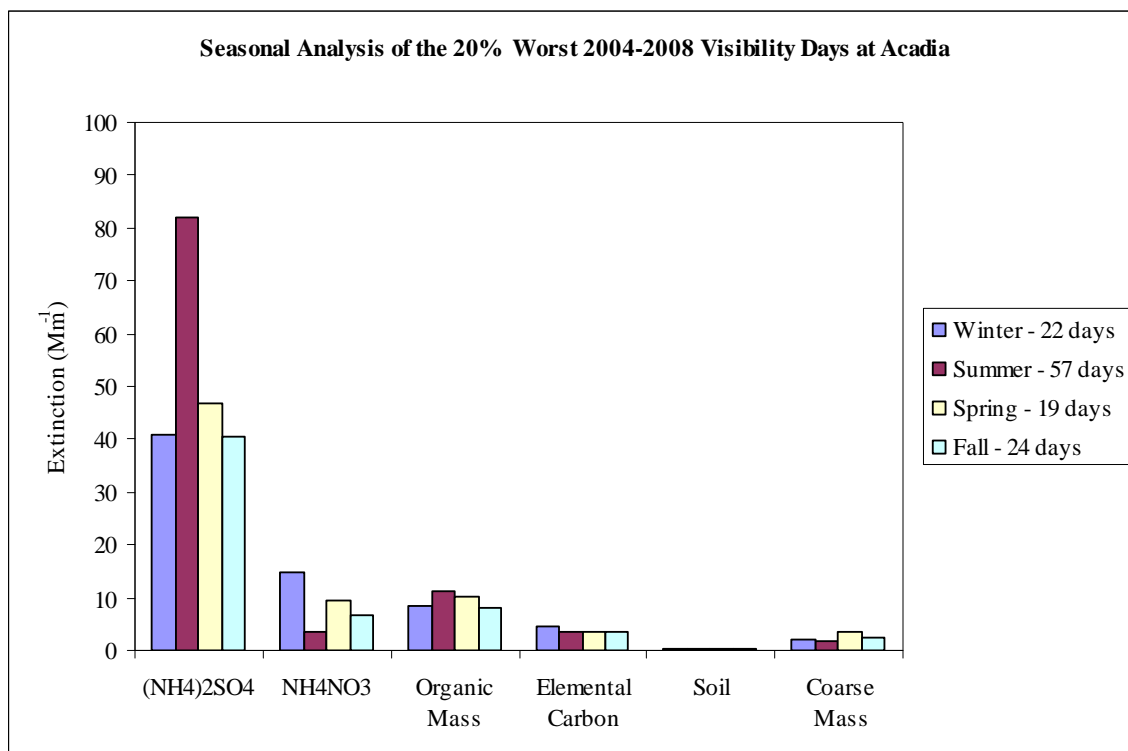
Figure B-8. 20% Worst and Best 2004-2008 Visibility Days at Brigantine, NJ

Seasonal Analysis of the 20% Worst 2004-2007 Visibility Days at Brigantine

Seasonal Analysis of the 20% Best 2004-2007 Visibility Days at Brigantine
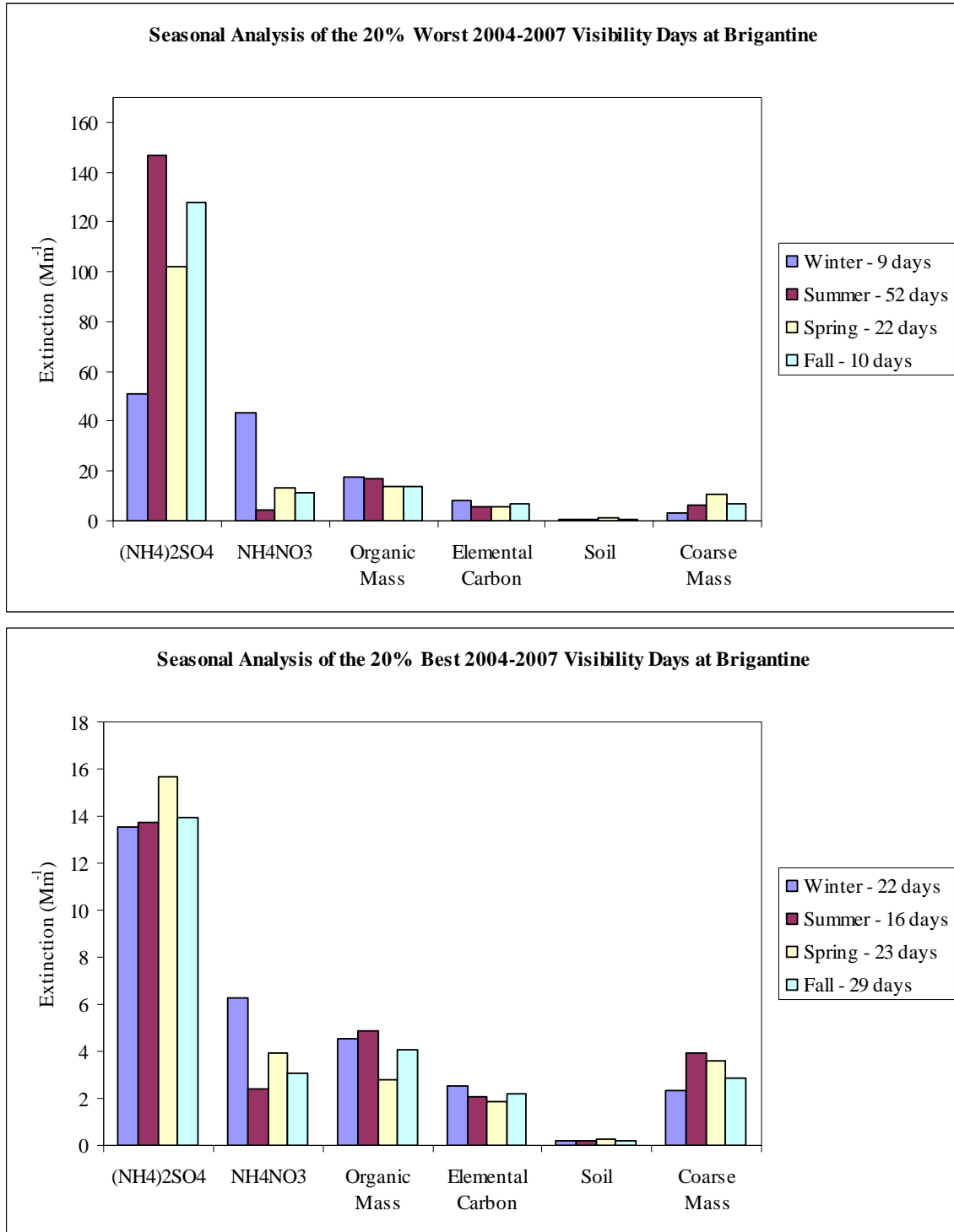
Figure B-9. 20% Worst and Best 2004-2008 Visibility Days at Great Gulf, NH

Seasonal Analysis of the 20% Worst 2004-2008 Visibility Days at Great Gulf

Seasonal Analysis of the 20% Best 2004-2008 Visibility Days at Great Gulf
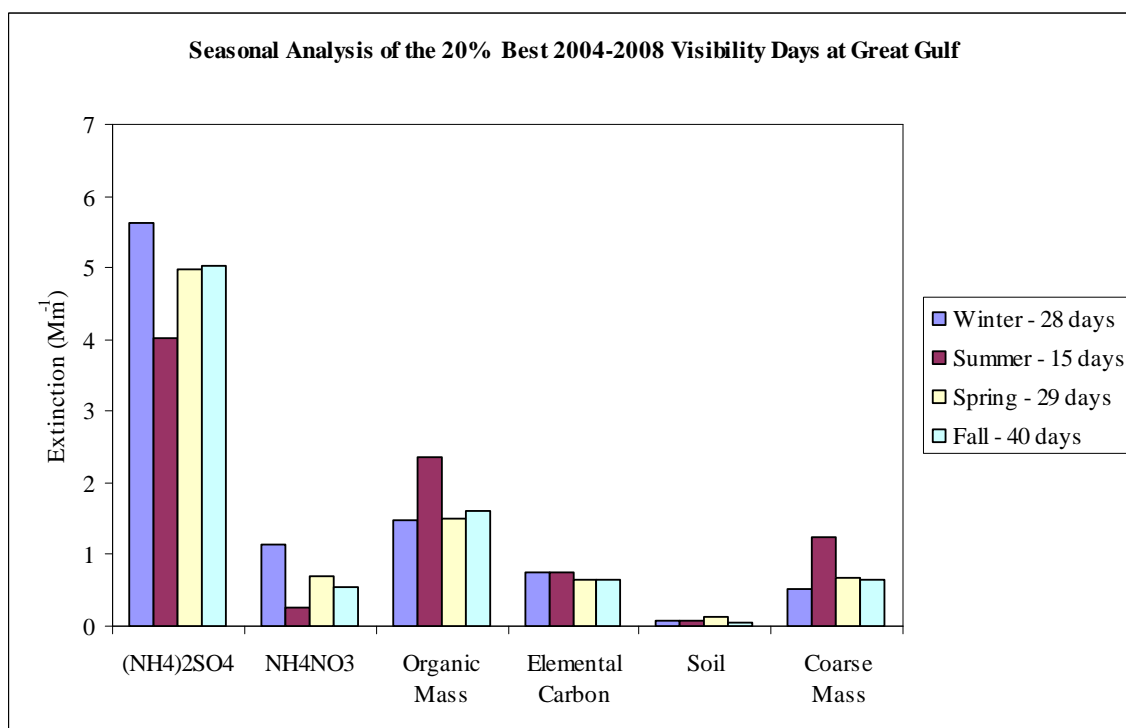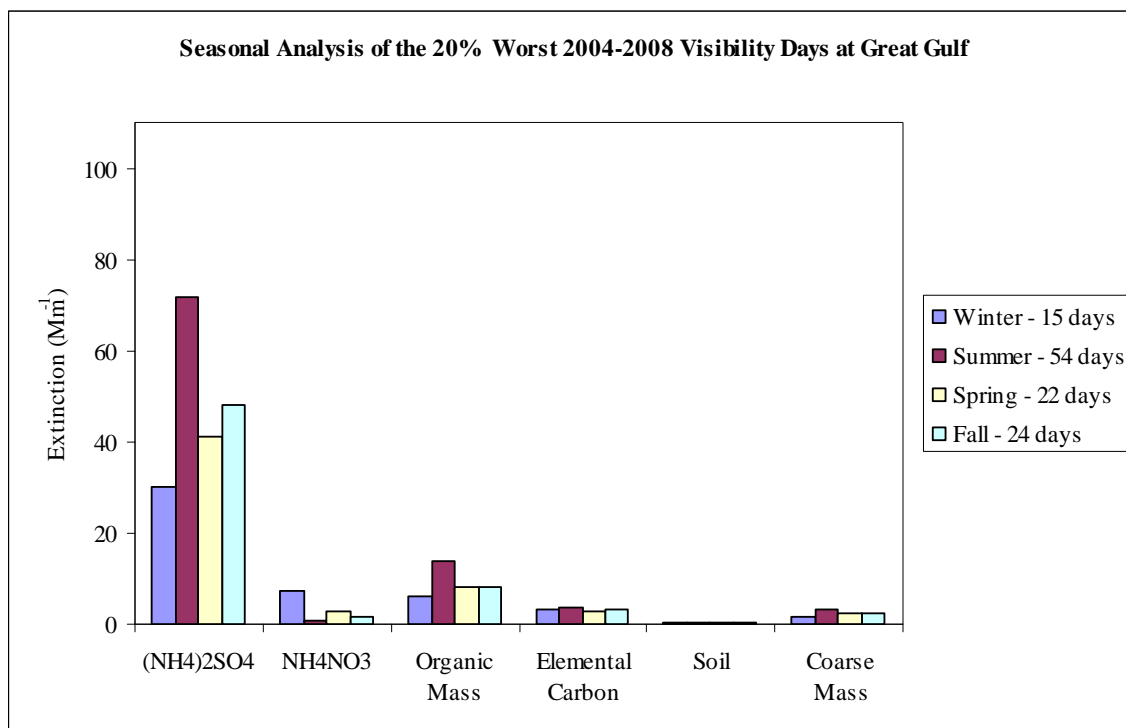
Figure B-10. 20% Worst and Best 2004-2008 Visibility Days at Lye Brook, VT

Seasonal Analysis of the 20% Best 2004-2007 Visibility Days at Lye Brook

Seasonal Analysis of the 20% Worst 2004-2007 Visibility Days at Lye Brook
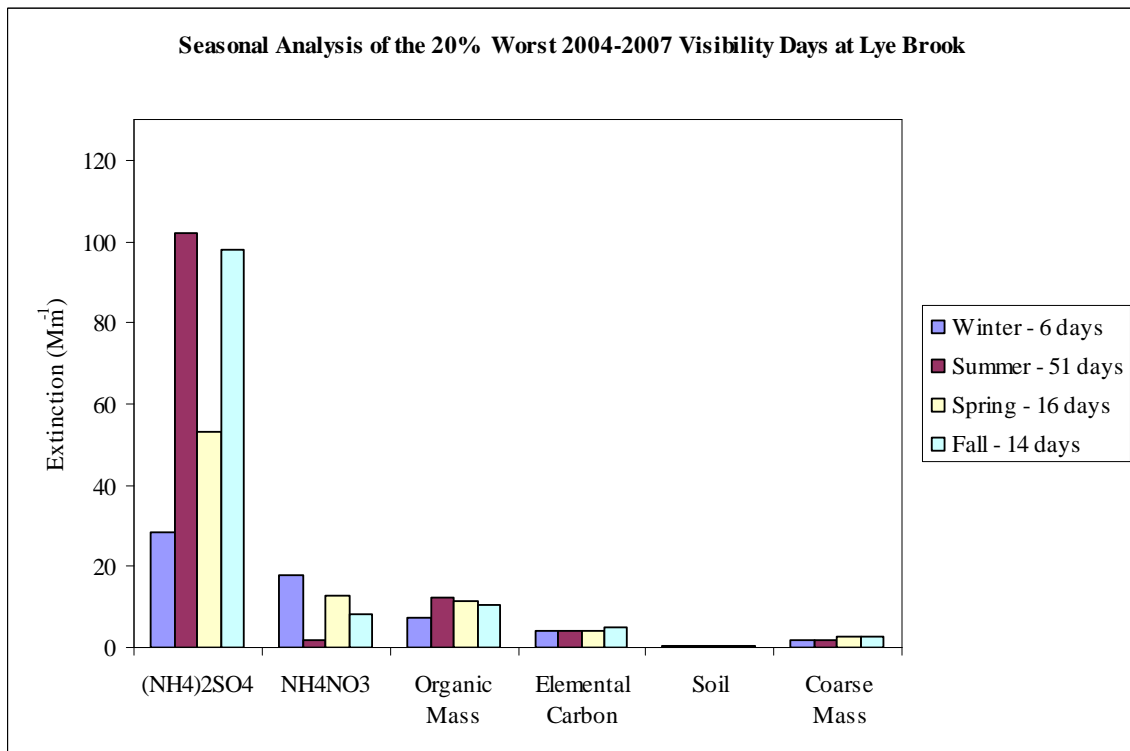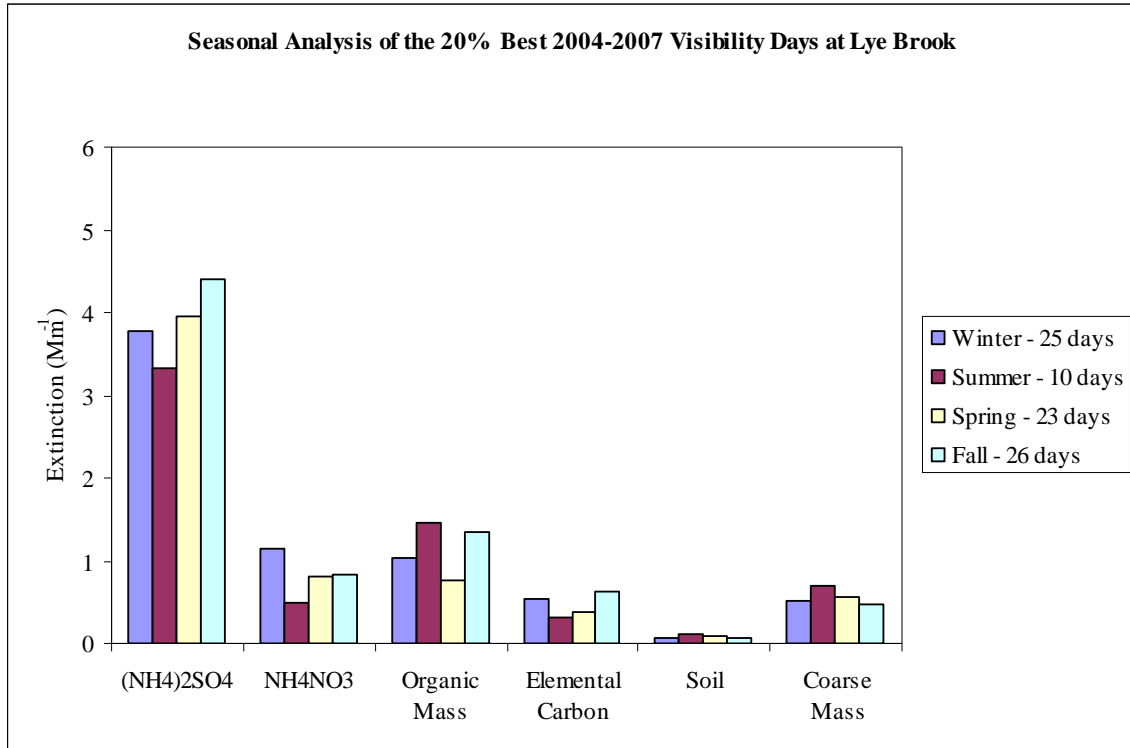
Figure B-11. 20% Worst and Best 2004-2008 Visibility Days at Moosehorn, ME

Seasonal Analysis of the 20% Worst 2004-2008 Visibility Days at Moosehorn

Seasonal Analysis of the 20% Best 2004-2008 Visibility Days at Moosehorn
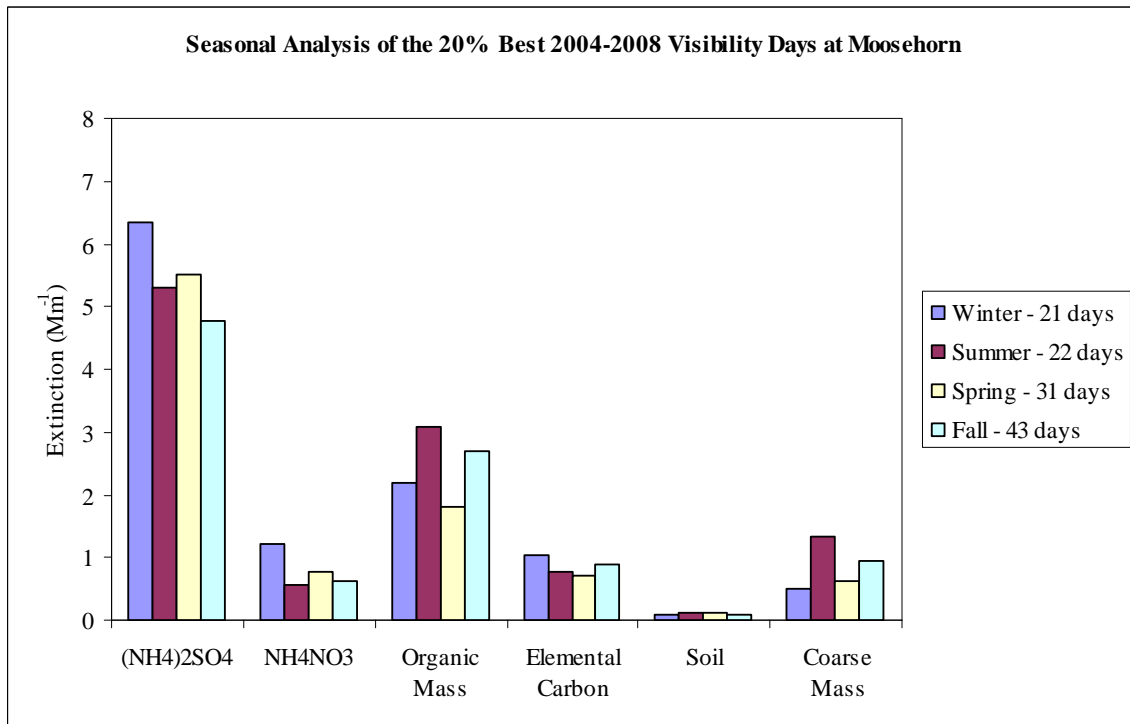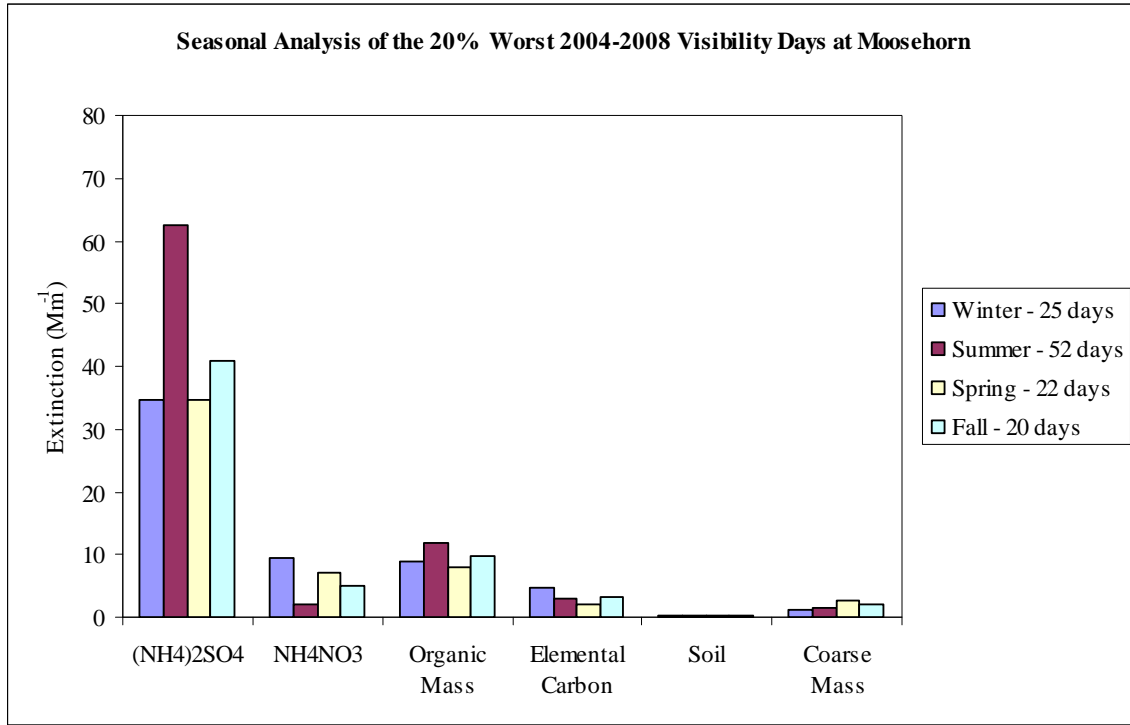
Figure B-12. 20% Worst and Best 2004-2008 Visibility Days at Washington, D.C.

Seasonal Analysis of the 20% Worst 2004-2008 Visibility Days at Washington, D.C.

Seasonal Analysis of the 20% Best 2004-2008 Visibility Days at Washington, D.C.
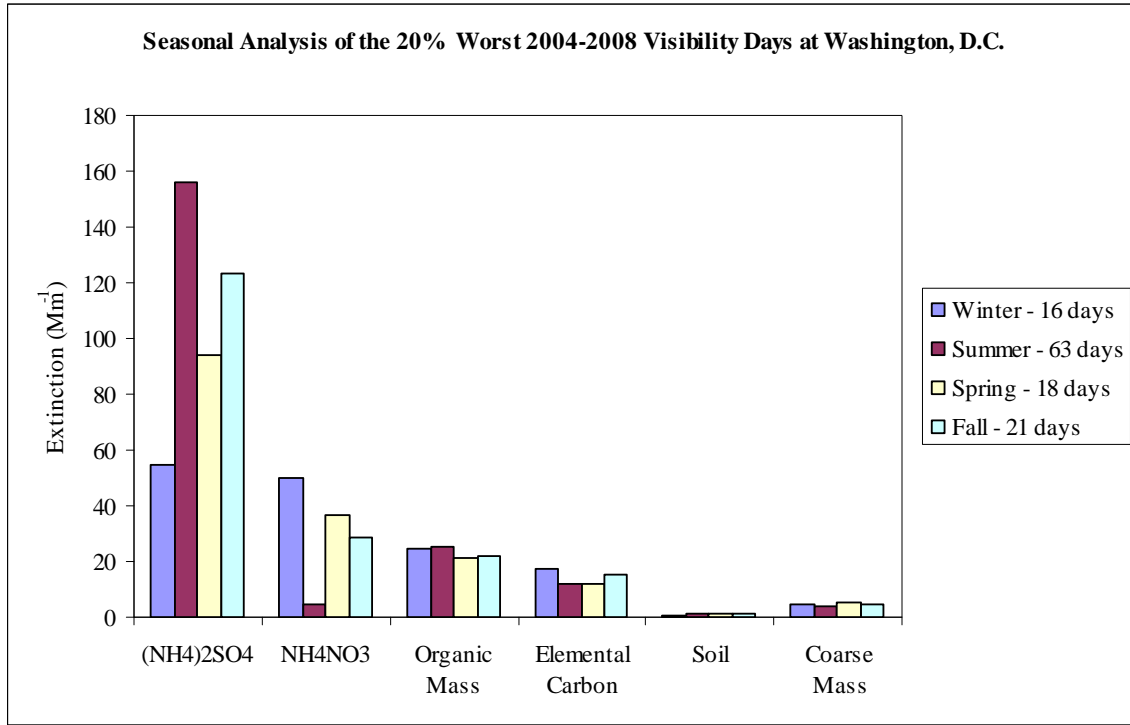
Figure B-13. 20% Best 2004-2008 Visibility Days Speciated Contributions to Extinction

Extinction on 20% Best Days (Mm^{-1})   **2004-2008**

Site	Total	NH$_4$NO$_3$	(NH$_4$)$_2$SO$_4$	Elemental Carbon	Organic Mass	Coarse Mass	Soil	Sea Salt
Acadia	10.6	0.8	5.4	0.7	2.1	0.8	0.1	0.6
Brigantine	30.1	3.9	14.3	2.2	4.0	3.2	0.2	2.3
Great Gulf	9.0	0.7	5.0	0.7	1.6	0.7	0.1	0.2
Lye Brook	7.2	0.9	4.0	0.5	1.1	0.5	0.1	0.1
Moosehorn	11.0	0.8	5.3	0.8	2.4	0.9	0.1	0.7
Washington, D.C.	42.8	6.6	17.8	7.0	6.8	3.2	0.5	0.8

Legend:
- NH$_4$NO$_3$
- (NH$_4$)$_2$SO$_4$
- EC
- OM
- Coarse Material
- Soil
- Sea Salt

Great Gulf

Moosehorn

Acadia

Lye Brook

20% Best Visibility

Speciated Contributions to Extinction (2004-2008 except for Lye Brook and Brigantine 2004-2007)

Brigantine

Washington, D.C.

$$b_{ext} \approx 2.2 \times f_s(RH) \times [Small\_Sulfate] + 4.8 \times f_L(RH) \times [Large\_Sulfate] + 2.4 \times f_s(RH) \times [Small\_Nitrate] + 5.1 \times f_L(RH) \times [Large\_Nitrate] + 2.8 \times [Small\_Organic\_Mass] + 6.1 \times [Large\_Organic\_Mass] + 10 \times [Elemental\_Carbon] + 1 \times [Fine\_Soil] + 1.7 \times f_{ss}(RH) \times [Sea\_Salt] + 0.6 \times [Coarse\_Mass] + Rayleigh\_Scattering(Site\_Specific) + 0.33 \times [NO_2(ppb)]$$

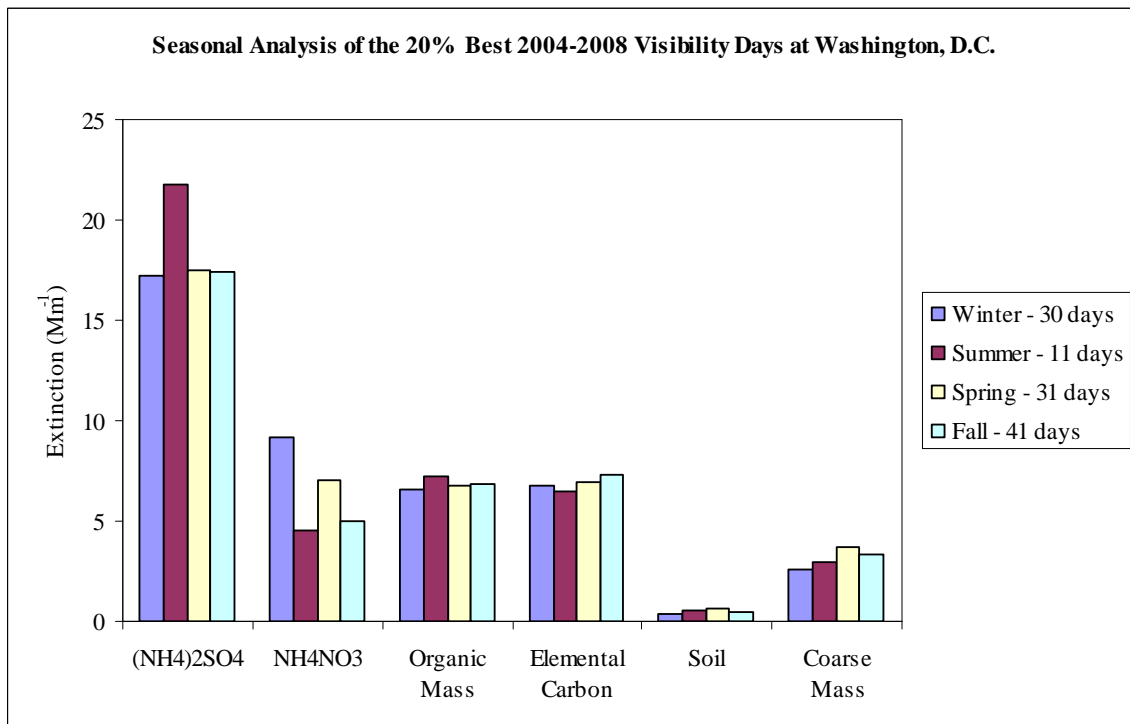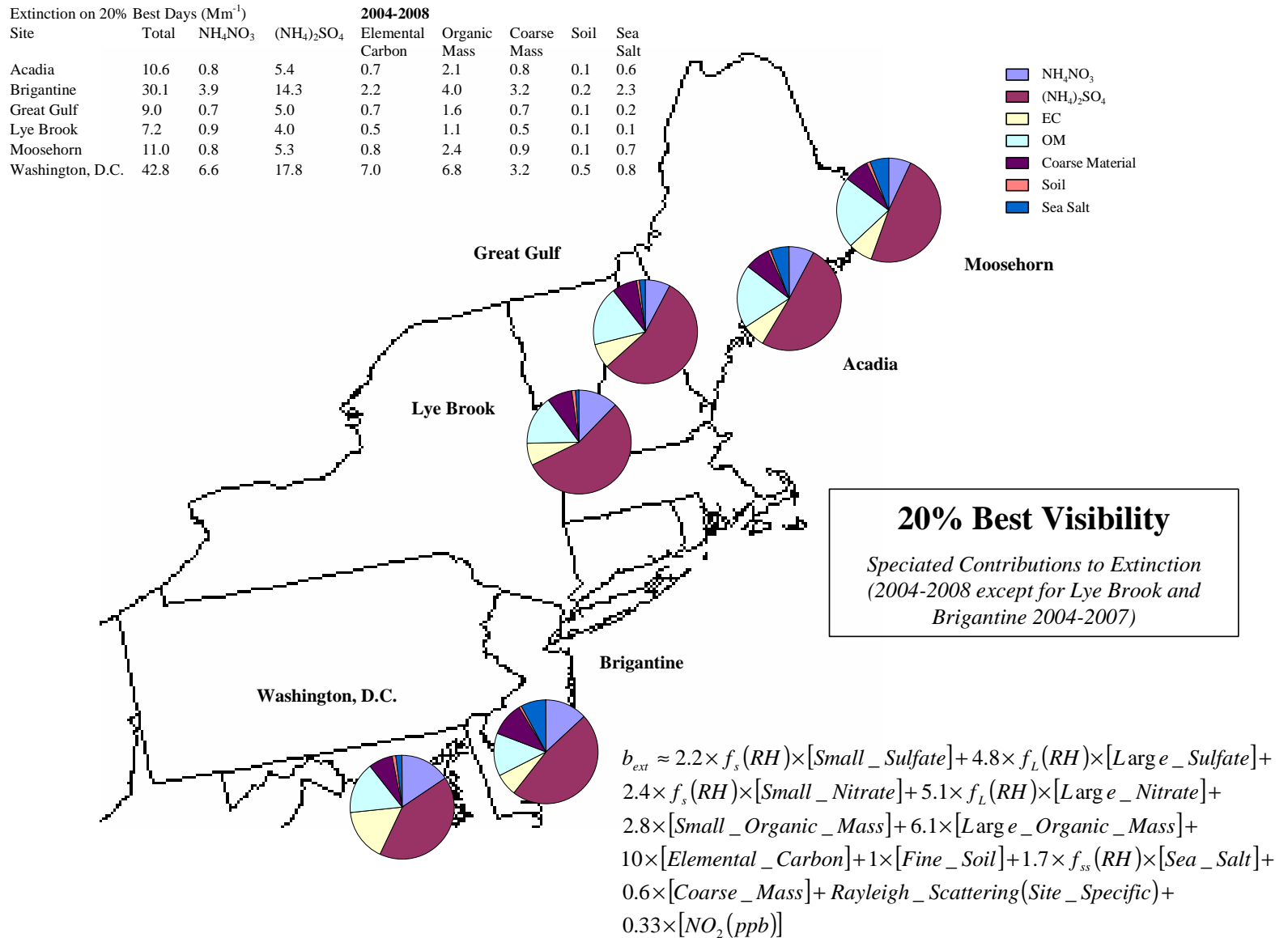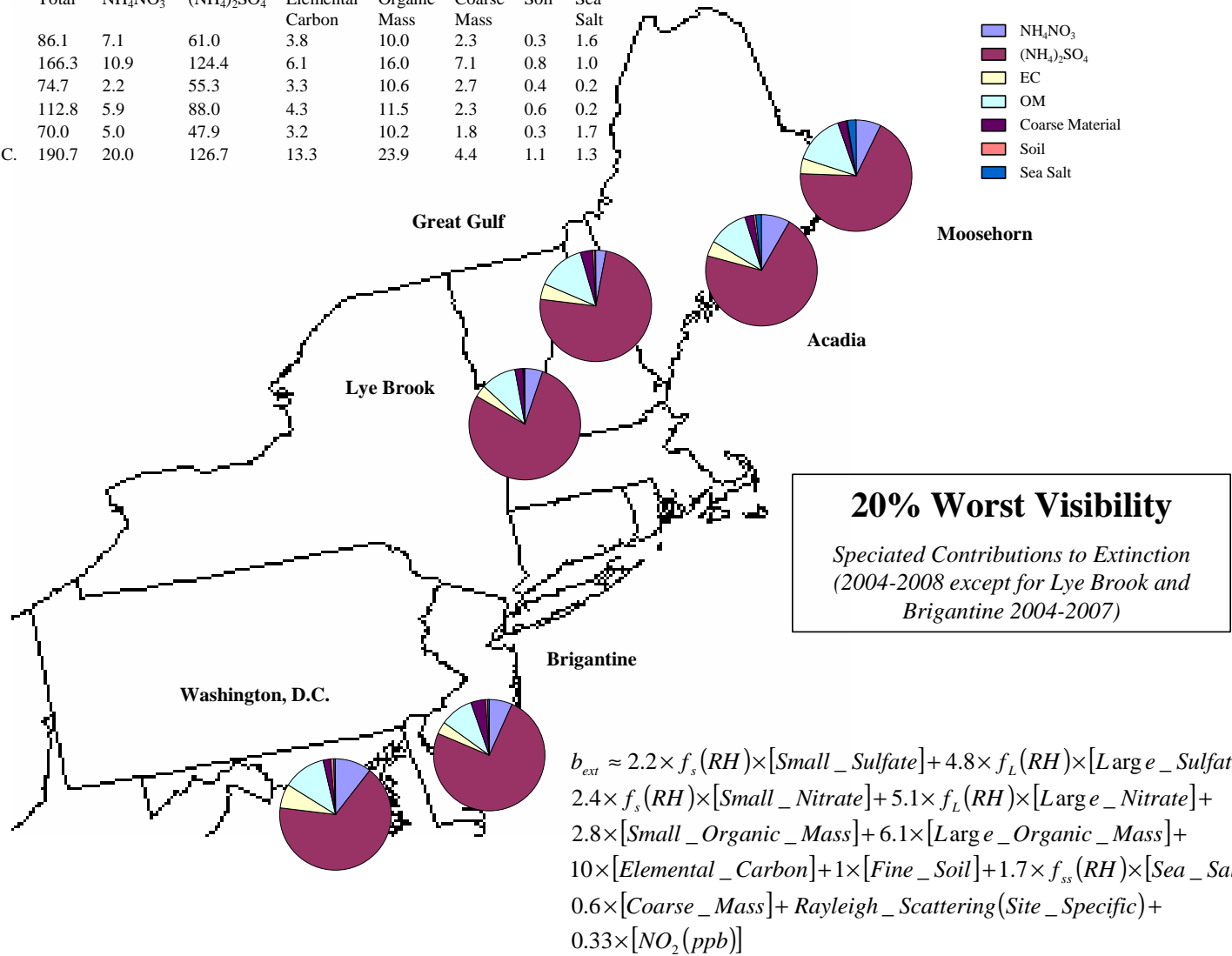## Figure B-14. 20% Best 2004-2008 Visibility Days Speciated Contributions to Extinction

Extinction on 20% Worst Days (Mm^{-1})

Site	Total	NH$_4$NO$_3$	(NH$_4$)$_2$SO$_4$	Elemental Carbon	Organic Mass	Coarse Mass	Soil	Sea Salt
				2004-2008				
Acadia	86.1	7.1	61.0	3.8	10.0	2.3	0.3	1.6
Brigantine	166.3	10.9	124.4	6.1	16.0	7.1	0.8	1.0
Great Gulf	74.7	2.2	55.3	3.3	10.6	2.7	0.4	0.2
Lye Brook	112.8	5.9	88.0	4.3	11.5	2.3	0.6	0.2
Moosehorn	70.0	5.0	47.9	3.2	10.2	1.8	0.3	1.7
Washington, D.C.	190.7	20.0	126.7	13.3	23.9	4.4	1.1	1.3

Legend:
- NH$_4$NO$_3$
- (NH$_4$)$_2$SO$_4$
- EC
- OM
- Coarse Material
- Soil
- Sea Salt

Great Gulf

Moosehorn

Acadia

Lye Brook

20% Worst Visibility

Speciated Contributions to Extinction (2004-2008 except for Lye Brook and Brigantine 2004-2007)

Washington, D.C.

Brigantine

$$b_{ext} \approx 2.2 \times f_s(RH) \times [Small\_Sulfate] + 4.8 \times f_L(RH) \times [Large\_Sulfate] +$$
$$2.4 \times f_s(RH) \times [Small\_Nitrate] + 5.1 \times f_L(RH) \times [Large\_Nitrate] +$$
$$2.8 \times [Small\_Organic\_Mass] + 6.1 \times [Large\_Organic\_Mass] +$$
$$10 \times [Elemental\_Carbon] + 1 \times [Fine\_Soil] + 1.7 \times f_{ss}(RH) \times [Sea\_Salt] +$$
$$0.6 \times [Coarse\_Mass] + Rayleigh\_Scattering(Site\_Specific) +$$
$$0.33 \times [NO_2(ppb)]$$

Appendix C: Additional Considerations for PM$_{2.5}$ Air Quality Management

APPENDIX C: ADDITIONAL CONSIDERATIONS FOR PM$_{2.5}$ AIR QUALITY MANAGEMENT

C.1. Averaging times and data interpretation

In analyzing the chemical data available for interpreting the air quality event of August 2002, it is important to point out that the use of different averaging times can have a profound effect on our understanding of the progression of any specific episode. Many subtleties of synoptic-scale meteorology and atmospheric chemistry are "aliased out" of data sets with temporal resolution greater than 3-6 hours. These effects are demonstrated in Figure C-1 which show fine aerosol TEOM data from New Haven for the "episode" period August 10-16, 2002. In these figures, the hourly TEOM values have been aggregated into 3-, 6- and 24-hour mean values. Average concentrations are inversely proportional to the length of the averaging period and the ratio of peak hourly concentration within a daily average ranges from about 1.5 to 1.75 for this episode.

Figure C-1. Effects of averaging times (or temporal resolution) on time series information
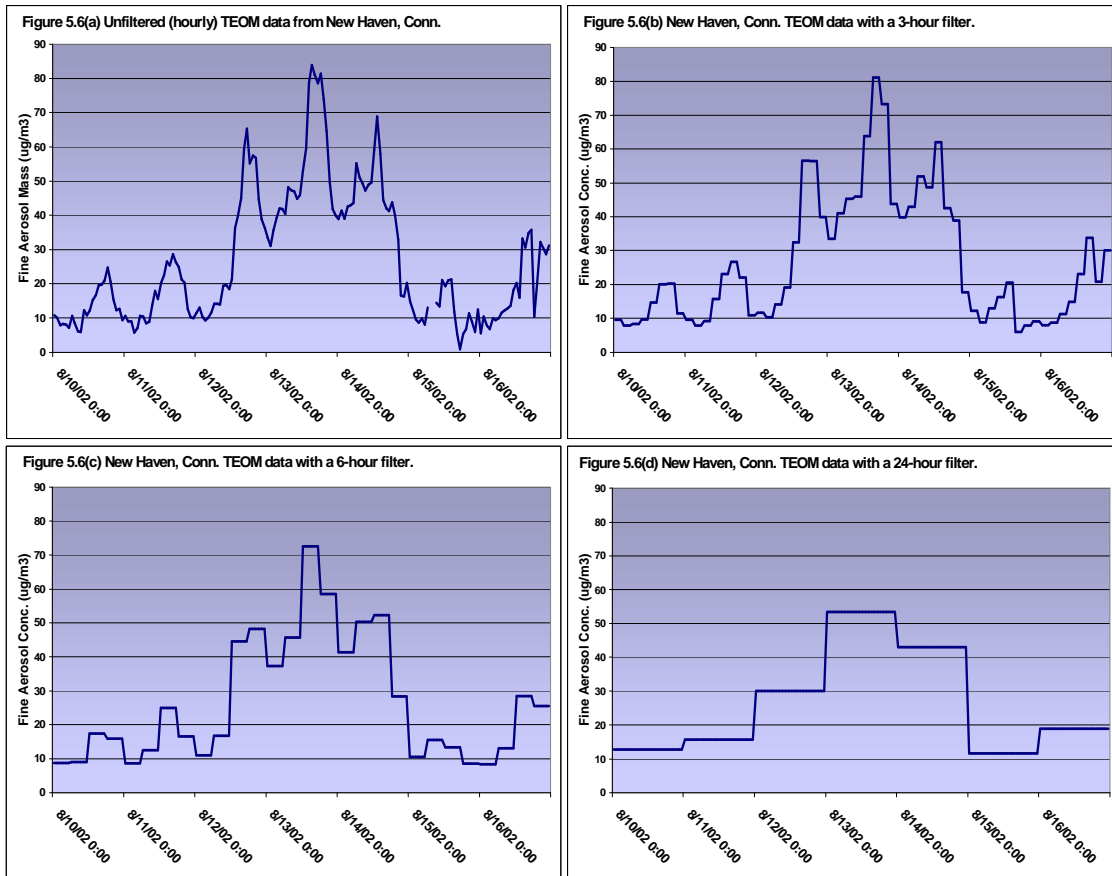
C.2. Rural versus urban PM$_{2.5}$ mass

Comparison of PM$_{2.5}$ concentrations from rural areas with those from urban/suburban areas can add significantly to our understanding of the impact on air quality of both urban sources and of medium to long-range fine aerosol transport. To assist with this approach, data from 10 pairs of rural and urban/suburban FRM sites throughout the MANE-VU region were selected and analyzed. Table C-1 shows basic site description information including the approximate, straight-line distance between the site pairs.

Due to the difficulty in finding a significant number of urban-rural site pairs that operated on the same sampling schedule, sites with a mixture of schedules were used to insure samples representative of the entire MANE-VU region. As a result, 3 of the 20 sites employed an everyday schedule while 2 sites sampled every sixth day (the remainder sampled every third day). Data from the three everyday sites were edited so as to include data from the 1-in-3 schedule only. In all, a total of 1098 data points were possible from the 10 site pairs for 2002. Of the 1098 possible point-pairs, 951 (87%) were valid and were used in this analysis.

**Table C-1. MANE-VU urban-rural site pair information**

State	Site No	City	Land use	Location type	Longitude	Latitude	Inter-site Distance (mi)
DE	100051002		Agricultural	Rural	-75.55560	38.98470	
DE	100010002	Seaford	Residential	Suburban	-75.61310	38.64440	24.0
MA	250154002	Ware	Forest	Rural	-72.33472	42.29833	
MA	250130016	Springfield	Commercial	Urban & Center City	-72.59140	42.10890	17.6
MD	240030014		Agricultural	Rural	-76.65310	38.90250	
MD	245100049	Baltimore	Residential	Urban & Center City	-76.63750	39.26170	25.2
ME	230052003	Cape Elizabeth	Residential	Rural	-70.20778	43.56083	
ME	230010011	Lewiston	Commercial	Urban & Center City	-70.21500	44.08940	37.0
NJ	340218001		Agricultural	Rural	-74.85470	40.31500	
NJ	340210008	Trenton	Residential	Urban & Center City	-74.76360	40.22220	7.7
NY	360010012	Albany	Agricultural	Rural	-73.75690	42.68070	
NY	360930003	Schenectady	Residential	Suburban	-73.94020	42.79960	11.7
NY	361030001	Babylon	Commercial	Rural	-73.42030	40.74580	
NY	360590013	Bethpage	Residential	Suburban	-73.49060	40.76080	3.3
NY	360130011	Westfield	Agricultural	Rural	-79.60250	42.29080	
PA	420490003	Erie	Commercial	Suburban	-80.03860	42.14180	22.2
PA	420030093		Residential	Rural	-80.02080	40.60720	
PA	420030021	Pittsburgh	Residential	Suburban	-79.94140	40.41360	14.0
PA	420290100		Commercial	Rural	-75.76860	39.83440	
DE	100031012	Newark	Residential	Suburban	-75.76170	39.69190	10.0

As expected, urban/suburban areas, with their rich supply of emission sources, almost always reported higher concentrations than their nearby sister sites in rural areas.

Of the 951 valid data pairs, 660 showed higher urban/suburban levels while 291 cases showed higher rural levels.

One interesting aspect of the 2002 urban-rural data concerns the pattern in seasonal differences between such site pairs. Figure C-2 shows the difference (urban-rural) between the 10 site pairs as a time series.

Figure C-2. Difference in FRM data between 10 urban-rural site pairs for 2002
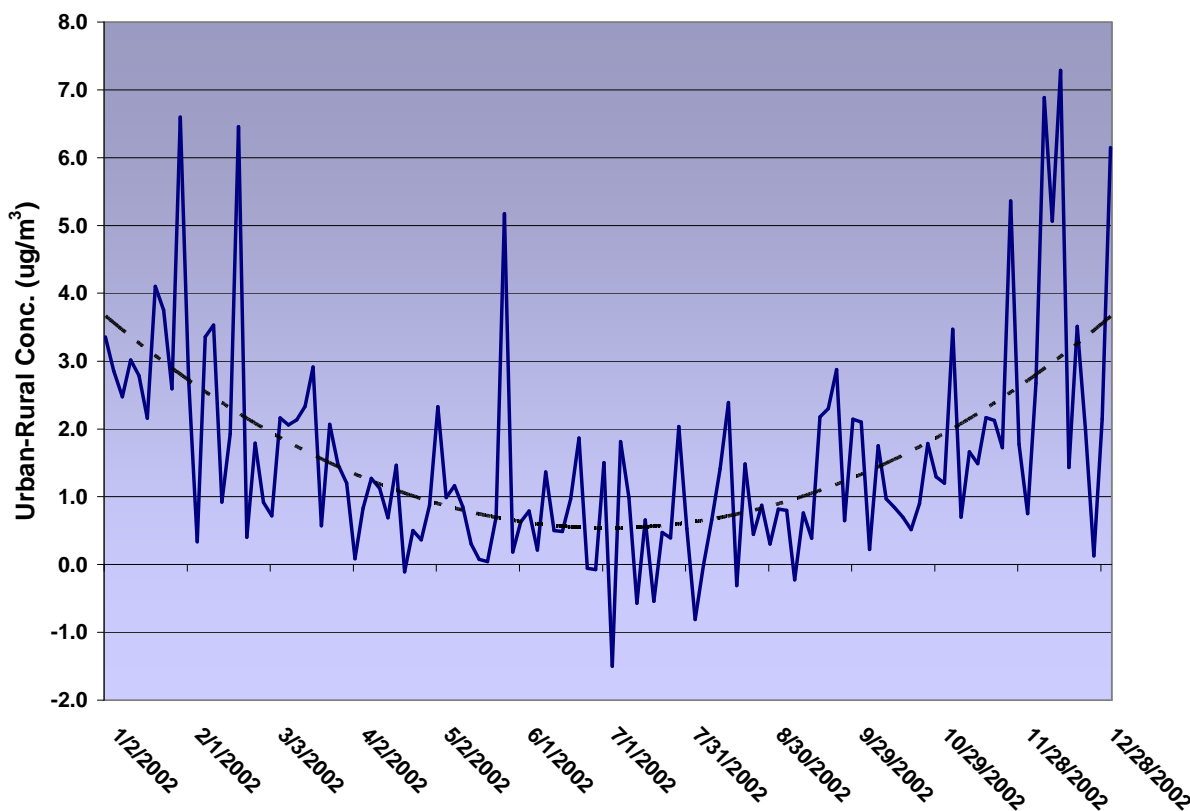
Although some rural-to-urban seasonal differences are to be expected, the variation in the magnitude of this difference is surprising. In the warm/hot months, the mean rural/urban difference amounts to no more than ~0.7 µg/m^3 (based on a best-fit 2nd order polynomial curve), which is a relatively small differential. However, during the cool/cold months that difference climbs to almost 4 µg/m^3, demonstrating a total annual seasonal variation of at least 3 µg/m^3. Because the mean annual concentration of all sites is 12.6 µg/m^3, an annual variation of 3 µg/m^3 becomes significant.

One explanation for the observed seasonal variation concerns the temporal distribution of local and transported emissions. In the summertime, MANE-VU sites repeatedly experience sulfate events due to transport from regions to the south and west. During such events, rural and urban sites throughout MANE-VU record high (i.e., >15 µg/m^3) daily average PM$_{2.5}$ concentrations. During summer stagnation events, atmospheric ventilation is poor and local emissions are added to the transported burden with the result that concentrations throughout the region (rural and urban) are relatively uniform. There are enough of these events to drive the urban-rural difference down to less than 1 µg/m^3 during warm/hot months.

During the wintertime, strong local inversions frequently trap local emissions during the overnight and early morning periods, resulting in elevated urban concentrations. Rural areas experience those same inversions but have relatively fewer local sources so that wintertime concentrations in rural locations tend to be lower than those in nearby urban areas. Medium and long-range fine aerosol transport events do occur during the winter but at a much reduced rate compared to summertime. So, it is the interplay between local and distant sources as well as meteorological conditions that drive the observed seasonal urban-rural difference in FRM concentrations.

C.3. Seasonal relationship between PM$_{2.5}$ and NO$_X$

Because nitrogen oxides (NO$_X$) can be a good indicator of regional as well as local emissions, NO$_X$ data for the MANE-VU region was downloaded from USEPA's AQS. Ultimately, data from six widely separated MANE-VU NO$_X$ sites were selected (one site each in CT, DC, MA, NH, PA and VT). Sites were selected both for high data capture rates and geographic location. The NO$_X$ data were then aggregated into regional averages on a daily basis and compared to PM$_{2.5}$ FRM data from 34 "everyday" sampling sites (which were also averaged on a regional basis).

During 2002, there were virtually no periods when regional mean PM$_{2.5}$ concentrations rose above 20 µg/m^3 and were not accompanied by rising (or already high) NO$_X$ concentrations. However, as seen in Figure C-3, NO$_X$ concentrations vary widely on an annual basis and tend to occur out-of-sync with fine particle concentrations.

Although the min/max extremes of these two pollutants are offset in time, they are highly correlated during some parts of the year. For example, Figure C-4 shows the regional PM$_{2.5}$ and NO$_X$ data for the coldest (Jan., Feb., Nov., and Dec.) and hottest (May, June, July, and Aug.) seasons of 2002. Wintertime NO$_X$ and PM$_{2.5}$ concentrations are rather well correlated (r^2=0.67) while summertime concentrations are not at all linked. This dichotomy can be explained by several coincident effects including: 1) reduced UV radiation during cold months (which reduces photolysis of NO$_2$); 2) the increase in space heating requirements from stationary sources (which preferentially increases morning NO$_X$ emissions); 3) increased NO$_X$ emissions due to "cold-start" mobile source engines; and 4) decreased mixing height depths due to reduced solar input (which allows morning concentrations to build quickly). Note that the spring/fall PM$_{2.5}$ vs. NO$_X$ correlation (not shown) lies about mid-way between the winter/summer values shown in Figure C-4.

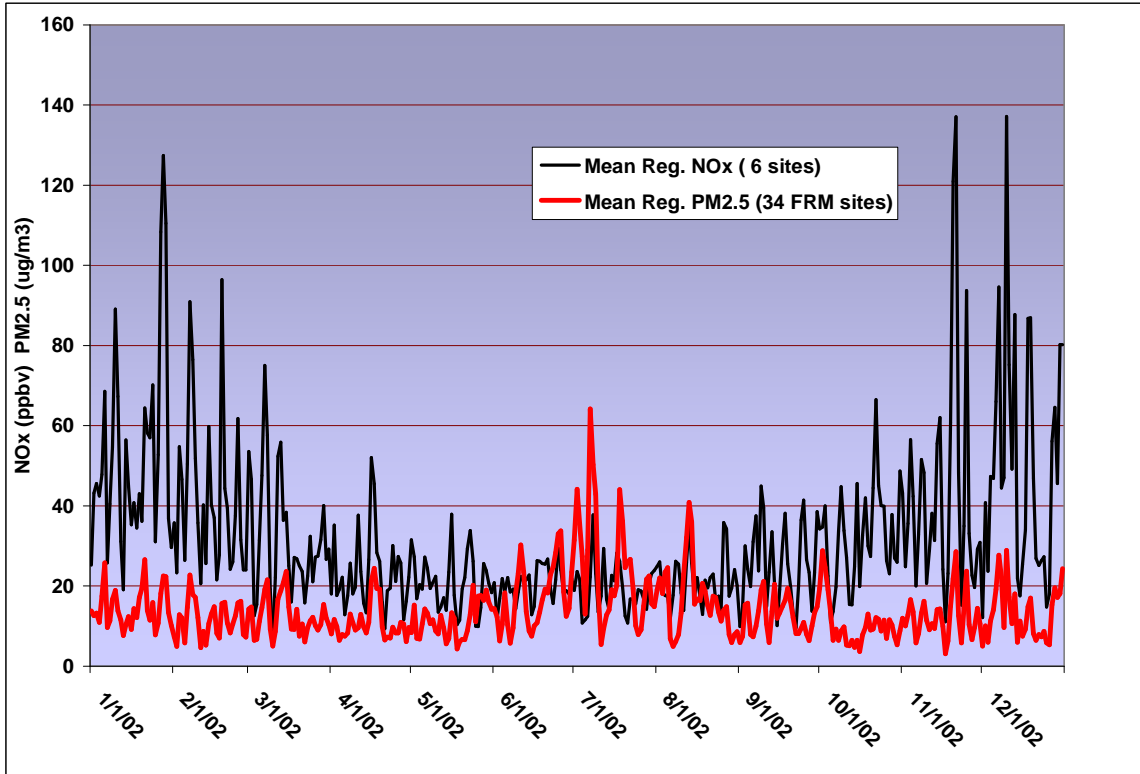**Figure C-3. Regional PM$_{2.5}$ and NO$_X$ in 2002**

Figure C-4. PM$_{2.5}$ vs. NO$_X$ correlation by season