



Policy Assessment for the Review of the Particulate Matter National Ambient Air Quality Standards

Second External Review Draft

June 2010

DISCLAIMER

This draft document has been prepared by staff from the Ambient Standards Group, Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency. Any opinions, findings, conclusions, or recommendations are those of the authors and do not necessarily reflect the views of the EPA. This document is being circulated to obtain review and comment from the Clean Air Scientific Advisory Committee (CASAC) and the general public. Comments on this draft document should be addressed to Beth Hassett-Sipple, U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards, C504-06, Research Triangle Park, North Carolina 27711 (email: hassett-sipple.beth@epa.gov).

***Policy Assessment
for the Review of the Particulate Matter
National Ambient Air Quality Standards
Second External Review Draft***

U.S. Environmental Protection Agency
Office of Air and Radiation
Office of Air Quality Planning and Standards
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EXECUTIVE SUMMARY

This second draft Policy Assessment (PA) has been prepared by staff in the Environmental Protection Agency's (EPA) Office of Air Quality Planning and Standards (OAQPS) in conjunction with the Agency's ongoing review of the national ambient air quality standards (NAAQS) for particulate matter (PM), which include primary (health-based) and secondary (welfare-based) standards. It presents staff conclusions regarding the adequacy of the current suite of PM standards as well as potential alternative standards for consideration in this review, based on the scientific and technical information assessed in other EPA documents, including the *Integrated Science Assessment for Particulate Matter (Final Report)* (ISA, US EPA, 2009a), the *Quantitative Health Risk Assessment for Particulate Matter (Final Report)* (RA, US EPA, 2010a) and the *Particulate Matter Urban-Focused Visibility Assessment (Final Report)* (UFVA, US EPA, 2010b).

This PA is intended to "bridge the gap" between the relevant scientific evidence and technical information and the judgments required of the EPA Administrator in determining whether, and if so how, to revise the PM NAAQS. The current and potential alternative PM standards are considered in terms of the basic elements of the NAAQS: indicator, averaging time, form, and level.

Primary standards for fine particles (Chapter 2):

In assessing the adequacy of the current suite of annual and 24-hour PM_{2.5} standards meant to protect public health against long- and short-term exposures to fine particles, staff concludes that the currently available information clearly calls into question the adequacy of the current standards and that consideration should be given to revising the suite of standards to provide increased public health protection. In considering alternative PM_{2.5} standards, staff concludes that protection from both long- and short-term PM_{2.5} exposures can most effectively and efficiently be provided by relying primarily on the annual standard, with the 24-hour standard providing supplemental protection for days with high peak concentrations. On this basis, staff concludes that consideration should be given to alternative annual PM_{2.5} standard levels in the range of 13 to 11 $\mu\text{g}/\text{m}^3$, in conjunction with retaining the current 24-hour PM_{2.5} standard level of 35 $\mu\text{g}/\text{m}^3$, and that consideration could also be given to an alternative 24-hour PM_{2.5} standard level of 30 $\mu\text{g}/\text{m}^3$ particularly in conjunction with an annual standard level of 11 $\mu\text{g}/\text{m}^3$.

Primary standard for thoracic coarse particles (Chapter 3):

In assessing the adequacy of the current primary 24-hour PM₁₀ standard meant to protect public health against short-term exposures to thoracic coarse particles, staff concludes that consideration should be given to retaining or revising the current standard, depending on the relative weight placed on the evidence supporting associations with PM_{10-2.5} and the uncertainties associated with this evidence. With regard to potential alternative standards, staff concludes that, to the extent consideration is given to revising the current standard to increase public health protection, consideration should be given to retaining the PM₁₀ indicator and the 24-hour averaging time and revising the form and

level, with consideration of levels from $85 \mu\text{g}/\text{m}^3$ down to about $65 \mu\text{g}/\text{m}^3$ in conjunction with a 98th percentile form.

Secondary standards for PM-related visibility impairment (Chapter 4):

In assessing the adequacy of the current suite of secondary annual and 24-hour $\text{PM}_{2.5}$ standards (which are identical to the primary $\text{PM}_{2.5}$ standards) meant to protect against PM-related visibility impairment, staff concludes that the currently available information clearly calls into question the adequacy of the current standards and that consideration should be given to revising the suite of standards to provide increased public welfare protection. In considering alternative standards, staff concludes that consideration should be given to establishing a new indicator based on using speciated $\text{PM}_{2.5}$ mass and relative humidity to calculate $\text{PM}_{2.5}$ light extinction. Staff also concludes that consideration should be given to a 1-hour averaging time, considering only daylight hours with relative humidity no higher than 90 %, and a level, defined in terms of $\text{PM}_{2.5}$ light extinction, in the range of 191 to 64 Mm^{-1} to target protection against visibility impairment related to fine particles.

Secondary standards for non-visibility welfare effects (Chapter 5):

In assessing the adequacy of the current suite of secondary PM standards (which are identical to the primary $\text{PM}_{2.5}$ and PM_{10} standards) meant to protect against PM-related effects other than visibility impairment, staff has considered PM-related effects on climate, ecological effects, and effects on materials. Staff concludes that the currently available information supports retaining control of both fine and coarse particles to address PM-related effects on ecosystems and materials damage and soiling, but that there is insufficient information to assess the adequacy of protection afforded by the current standards. Staff also concludes that there is insufficient information at this time to base a national ambient standard on climate impacts associated with current ambient concentrations of PM or its constituents.

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LIST OF ACRONYMS/ABBREVIATIONS

ACS	American Cancer Society
AOD	Aerosol optical depth
ANS	Autonomic nervous system
AQCD	Air Quality Criteria Document
AQS	EPA's Air Quality System
AR4	Fourth Assessment Report of the Intergovernmental Panel on Climate Change
BC	Black carbon
BP	Blood pressure
C	Carbon
Ca	Calcium
CAA	Clean Air Act
CAPs	Concentrated ambient particles
CASAC	Clean Air Scientific Advisory Committee
CBVD	Cerebrovascular disease
CCN	Cloud Condensation Nuclei
CCSP	US Climate Change Science Program
Cd	Cadmium
CFR	Code of Federal Regulations
CHD	Coronary heart disease
CHF	Congestive heart failure
CHS	Childrens Health Study
CO	Carbon monoxide
CO ₂	Carbon dioxide
COPD	Chronic obstructive pulmonary disease
C-R	Concentration-response relationship
CRP	C-reactive protein
CSN	Chemical Speciation Network
Cu	Copper
CV	Cardiovascular

CVD	Cardiovascular disease
DE	Diesel exhaust
DEP	Diesel exhaust particles
dv	deciview
DV	Design value
DVT	Deep vein thrombosis
EC	Elemental carbon
ECG	Electrocardiogram
EPA	Environmental Protection Agency
FEM	Federal Equivalent Method
FEV ₁	Change in forced expiratory volume in one second
FRM	Federal Reference Method
GEOS	Global Scale Air Circulation Model
GHG	Greenhouse gas
GI	Group interviews
Hg	Mercury
HR	Heart rate
HRV	Heart rate variability
IHD	Ischemic heart disease
IMPROVE	Interagency Monitoring of Protected Visual Environment
IPCC	Intergovernmental Panel on Climate Change
IRP	Integrated Review Plan
ISA	Integrated Science Assessment
IT	Intratracheal
Lag	Time between one event and another
Lag 0	Same day as the death, test, hospital, ED, clinic, physician visit; that occurs on the same day as the exposure to the pollutant(s)
Lag 0-x	All the deaths test, hospital, ED, clinic, physician visit; that occurs on the same day as the exposure to the pollutant(s) and the x days following the day of exposure
MCAPS	Medicare Air Pollution Study
MEA	Millennium Ecosystem Assessment

MI	Myocardial infarction
Mm	Megameter
MSA	Metropolitan Statistical Area
N	Nitrogen
NAAQS	National Ambient Air Quality Standards
NCEA	National Center for Environmental Assessment
NCore	National Core Monitoring Network
Ni	Nickel
NMMAPS	National Morbidity, Mortality, and Air Pollution Study
NO	Nitric oxide
NO ₂	Nitrogen dioxide
NO ₃ ⁻	Nitrate
NO _x	Nitrogen oxides
NRC	National Research Council
O ₃	Ozone
OAQPS	Office of Air Quality Planning and Standards
OAR	Office of Air and Radiation
OC	Organic carbon
OR	Odds ratio
ORD	Office of Research and Development
PA	Policy Assessment
PAH	Polyaromatic Hydrocarbon
PAR	Photosynthetically active radiation
Pb	Lead
PBDES	Polybrominated diphenyl ethers
PM	Particulate matter
PM ₁₀	Particles with an upper 50% cut-point of 10± 0.5 µm aerodynamic diameter and a penetration curve as specified in the Code of Federal Regulations.
PM _{10-2.5}	Particles with an upper 50% cut-point of 10 µm aerodynamic diameter and a lower 50% cut-point of 2.5 µm aerodynamic diameter.

PM _{2.5}	Particles with an upper 50% cut-point of 2.5 µm aerodynamic diameter and a penetration curve as specified in the Code of Federal Regulations.
PM _X	The legal definition for PM _X , as defined in the Code of Federal Regulations, includes both a 50% cut-point and a penetration curve. A 50% cut-point of X µm diameter means that 50% of particles with aerodynamic diameter of X are removed by the inlet and 50% pass through the inlet and are collected on the filter. Depending on the specific penetration curve specified, particles larger than X µm aerodynamic diameter are collected with an efficiency that decreases rapidly for particles larger than X while the collection efficiency for particles smaller than X increases rapidly with decreasing size until 100 % efficiency is reached.
POP	Persistent organic pollutants
PRB	Policy-Relevant Background
RA	Risk Assessment
REA	Risk and Exposure Assessment
RF	Radiative forcing
RH	Relative humidity
RR	Relative risk
SCAB	South Coast Air Basin
SD	Standard deviation
SES	Socioeconomic status
Si	Silicon
S	Sulfur
SO ₂	Sulfur Dioxide
SO ₄ ²⁻	Sulfate
SO _x	Sulfur Oxides
SOCs	Semi-volatile organic compounds
TSP	Total suspended particulate
UFPs	Ultrafine particles
UFVA	Urban-Focused Visibility Assessment
V	Vanadium
VAQ	Visual Air Quality
VOC	Volatile organic compounds

WACAP	Western Airborne Contaminants Assessment Project
WHI	Women's Health Initiative
Zn	Zinc

1. INTRODUCTION

1.1 PURPOSE

The U.S. Environmental Protection Agency (EPA) is presently conducting a review of the national ambient air quality standards (NAAQS) for particulate matter (PM). The plan and schedule for this review were presented in the *Integrated Review Plan for the National Ambient Air Quality Standards for Particulate Matter* (IRP; US EPA, 2008a). The IRP identified key policy-relevant issues to be addressed in this review as a series of questions that frame our consideration of whether the current NAAQS for PM should be retained or revised.

This Policy Assessment (PA), prepared by staff in the EPA's Office of Air Quality Planning and Standards (OAQPS), is intended to help "bridge the gap" between the relevant scientific information and assessments and the judgments required of the EPA Administrator in determining whether, and if so, how, it is appropriate to revise the NAAQS for PM.¹ This second draft PA presents factors relevant to EPA's review of the primary (health-based) and secondary (welfare-based) PM NAAQS. It focuses on both evidence- and risk-based information in evaluating the adequacy of the current PM NAAQS and in identifying potential alternative standards for consideration.

In this second draft PA, we consider the scientific and technical information available in this review as assessed in the *Integrated Science Assessment for Particulate Matter (Final Report)* (ISA, US EPA, 2009a), the *Quantitative Health Risk Assessment for Particulate Matter (Final Report)* (RA, US EPA, 2010a) and the *Particulate Matter Urban-Focused Visibility Assessment (Final Report)* (UFVA, US EPA, 2010b). In so doing, we focus on information that is most pertinent to evaluating the basic elements of NAAQS: indicator², averaging time, form,³ and level. These elements, which together serve to define each standard, must be considered collectively in evaluating the health and welfare protection afforded by the PM standards.

Although this second draft PA should be of use to all parties interested in this PM NAAQS review, it is written with an expectation that the reader has familiarity with the technical discussions contained in the ISA (US EPA, 2009a) and in the quantitative risk and visibility assessment documents (US EPA, 2010a,b).

¹ Preparation of a PA by OAQPS staff reflects Administrator Jackson's decision to modify the NAAQS review process that was presented in the IRP. See <http://www.epa.gov/ttn/naaqs/review.html> for more information on the current NAAQS review process.

² The "indicator" of a standard defines the chemical species or mixture that is to be measured in determining whether an area attains the standard.

³ The "form" of a standard defines the air quality statistic that is to be compared to the level of the standard in determining whether an area attains the standard.

1 **1.2 BACKGROUND**

2 **1.2.1 Legislative Requirements**

3 Two sections of the Clean Air Act (CAA) govern the establishment and revision of the
4 NAAQS. Section 108 (42 U.S.C. section 7408) directs the Administrator to identify and list air
5 pollutants that meet three specified criteria, including air pollutants “emissions of which, in his
6 judgment, cause or contribute to air pollution which may reasonably be anticipated to endanger
7 public health and welfare” and whose “presence . . . in the ambient air results from numerous or
8 diverse mobile or stationary sources” and to issue air quality criteria for those that are listed. Air
9 quality criteria are to “accurately reflect the latest scientific knowledge useful in indicating the
10 kind and extent of all identifiable effects on public health or welfare which may be expected
11 from the presence of [a] pollutant in the ambient air . . .” 42 U.S.C. § 7408(b).

12 Section 109 (42 U.S.C. section 7409) directs the Administrator to propose and
13 promulgate “primary” and “secondary” NAAQS for pollutants for which air quality criteria are
14 issued. Section 109(b)(1) defines a primary standard as one “the attainment and maintenance of
15 which in the judgment of the Administrator, based on [air quality] criteria and allowing an
16 adequate margin of safety, are requisite to protect the public health.”⁴ A secondary standard, as
17 defined in Section 109(b)(2), must “specify a level of air quality the attainment and maintenance
18 of which, in the judgment of the Administrator, based on such [air quality] criteria, is requisite to
19 protect the public welfare from any known or anticipated adverse effects associated with the
20 presence of [the] pollutant in the ambient air.”⁵

21 The requirement that primary standards include an adequate margin of safety was
22 intended to address uncertainties associated with inconclusive scientific and technical
23 information available at the time of standard setting. It was also intended to provide a reasonable
24 degree of protection against hazards that research has not yet identified. *Lead Industries*
25 *Association v. EPA*, 647 F.2d 1130, 1154 (D.C. Cir 1980), cert. denied, 449 U.S. 1042 (1980);
26 *American Petroleum Institute v. Costle*, 665 F.2d 1176, 1186 (D.C. Cir. 1981), cert. denied, 455
27 U.S. 1034 (1982); *American Farm Bureau Federation v. EPA*, 559 F. 3d 512, 533 (D.C. Cir.
28 2009). Both kinds of uncertainties are components of the risk associated with pollution at levels
29 below those at which human health effects can be said to occur with reasonable scientific

⁴ The legislative history of section 109 indicates that a primary standard is to be set at “the maximum permissible ambient air level . . . which will protect the health of any [sensitive] group of the population,” and that for this purpose “reference should be made to a representative sample of persons comprising the sensitive group rather than to a single person in such a group.” S. Rep. No. 91-1196, 91st Cong., 2d Sess. 10 (1970).

⁵ Welfare effects as defined in section 302(h) (42 U.S.C. section 7602(h)) include, but are not limited to, “effects on soils, water, crops, vegetation, man-made materials, animals, wildlife, weather, visibility and climate, damage to and deterioration of property, and hazards to transportation, as well as effects on economic values and on personal comfort and well-being.”

1 certainty. Thus, in selecting primary standards that include an adequate margin of safety, the
2 Administrator is seeking not only to prevent pollution levels that have been demonstrated to be
3 harmful but also to prevent lower pollutant levels that may pose an unacceptable risk of harm,
4 even if the risk is not precisely identified as to nature or degree.

5 In selecting a margin of safety, EPA considers such factors as the nature and severity of
6 the health effects involved, the size of the susceptible population(s) at risk, and the kind and
7 degree of the uncertainties that must be addressed. The selection of any particular approach to
8 providing an adequate margin of safety is a policy choice left specifically to the Administrator's
9 judgment. *Lead Industries Association v. EPA*, supra, 647 F.2d at 1161-62.

10 In setting standards that are "requisite" to protect public health and welfare, as provided
11 in section 109(b), EPA's task is to establish standards that are neither more nor less stringent
12 than necessary for these purposes. In so doing, EPA may not consider the costs of implementing
13 the standards. See generally *Whitman v. American Trucking Associations*, 531 U.S. 457, 471,
14 475-76 (2001).

15 Section 109(d) (1) of the CAA requires that "not later than December 31, 1980, and at 5-
16 year intervals thereafter, the Administrator shall complete a thorough review of the criteria
17 published under section 108 and the national ambient air quality standards . . . and shall make
18 such revisions in such criteria and standards and promulgate such new standards as may be
19 appropriate in accordance with section [108]... and subsection (b) . . ." 42 U.S.C. § 7409(d)(1).
20 Section 109(d)(2) requires that an independent scientific review committee "shall complete a
21 review of the criteria . . . and the national primary and secondary ambient air quality standards . .
22 . and shall recommend to the Administrator any new . . . standards and revisions of existing
23 criteria and standards as may be appropriate . . ." 42 U.S.C. § 7409(d)(2). Since the early
24 1980's, this independent review function has been performed by the Clean Air Scientific
25 Advisory Committee (CASAC).

26 **1.2.2 Previous PM NAAQS Reviews**

27 The EPA initially established NAAQS for PM under section 109 of the CAA in 1971.
28 Since then, the Agency has made a number of changes to these standards to reflect continually
29 expanding scientific information, particularly with respect to the selection of indicator⁶ and level.
30 Table 1-1 provides a summary of the PM NAAQS that have been promulgated to date. These
31 decisions are briefly discussed below.

⁶ Particulate matter is the generic term for a broad class of chemically and physically diverse substances that exist as discrete particles (liquid droplets or solids) over a wide range of sizes, such that the indicator for a PM NAAQS has historically been defined in terms of particle size ranges.

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Table 1-1. Summary of National Ambient Air Quality Standards Promulgated for Particulate Matter 1971-2006⁷

Final Rule	Indicator	Ave. Time	Level	Form
1971 36 FR 8186 April 30, 1971	TSP	24-hour	260 µg/m ³ (primary) 150 µg/m ³ (secondary)	Not to be exceeded more than once per year
		Annual	75 µg/m ³ (primary)	Annual average
1987 52 FR 24634 July 1, 1987	PM ₁₀	24-hour	150 µg/m ³	Not to be exceeded more than once per year on average over a 3-year period
		Annual	50 µg/m ³	Annual arithmetic mean, averaged over 3 years
1997 62 FR 38652 July 18, 1997	PM _{2.5}	24-hour	65 µg/m ³	98 th percentile, averaged over 3 years
		Annual	15 µg/m ³	Annual arithmetic mean, averaged over 3 years ⁸
	PM ₁₀	24-hour	150 µg/m ³	Initially promulgated 99 th percentile, averaged over 3 years; when 1997 standards were vacated, the form of 1987 standards remained in place (not to be exceeded more than once per year on average over a 3-year period)
		Annual	50 µg/m ³	Annual arithmetic mean, averaged over 3 years
2006 71 FR 61144 October 17, 2006	PM _{2.5}	24-hour	35 µg/m ³	98 th percentile, averaged over 3 years
		Annual	15 µg/m ³	Annual arithmetic mean, averaged over 3 years ⁹
	PM ₁₀	24-hour	150 µg/m ³	Not to be exceeded more than once per year on average over a 3-year period

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⁷ When not specified, primary and secondary standards are identical.

⁸ The level of the standard was to be compared to measurements made at sites that represent “community-wide air quality” recording the highest level, or, if specific constraints were met, measurements from multiple community-wide air quality monitoring sites could be averaged (“spatial averaging”).

⁹ The constraints on the spatial averaging criteria were tightened by further limiting the conditions under which some areas may average measurements from multiple community-oriented monitors to determine compliance (see 71 FR 61165-61167).

1 In 1971, EPA established NAAQS for PM based on the original air quality criteria
2 document (DHEW, 1969; 36 FR 8186, April 30, 1971). The reference method specified for
3 determining attainment of the original standards was the high-volume sampler, which collects
4 PM up to a nominal size of 25 to 45 micrometers (μm) (referred to as total suspended particles or
5 TSP). The primary standards (measured by the indicator TSP) were $260 \mu\text{g}/\text{m}^3$, 24-hour average,
6 not to be exceeded more than once per year, and $75 \mu\text{g}/\text{m}^3$, annual geometric mean. The
7 secondary standard was $150 \mu\text{g}/\text{m}^3$, 24-hour average, not to be exceeded more than once per
8 year.

9 In October 1979, EPA announced the first periodic review of the criteria and NAAQS for
10 PM, and significant revisions to the original standards were promulgated in 1987 (52 FR 24634,
11 July 1, 1987). In that decision, EPA changed the indicator for PM from TSP to PM_{10} , the latter
12 including particles with a median aerodynamic diameter less than or equal to $10 \mu\text{m}$, which
13 delineates thoracic particles (i.e., that subset of inhalable particles small enough to penetrate
14 beyond the larynx to the thoracic region of the respiratory tract). The EPA also revised the
15 primary standards by: (1) replacing the 24-hour TSP standard with a 24-hour PM_{10} standard of
16 $150 \mu\text{g}/\text{m}^3$ with no more than one expected exceedance per year; and (2) replacing the annual
17 TSP standard with a PM_{10} standard of $50 \mu\text{g}/\text{m}^3$, annual arithmetic mean. The secondary
18 standard was revised by replacing it with 24-hour and annual standards identical in all respects to
19 the primary standards. The revisions also included a new reference method for the measurement
20 of PM_{10} in the ambient air and rules for determining attainment of the new standards. On
21 judicial review, the revised standards were upheld in all respects. *Natural Resources Defense*
22 *Council v. EPA*, 902 F. 2d 962 (D.C. Cir. 1990), cert. denied, 498 U.S. 1082 (1991).

23 In April 1994, EPA announced its plans for the second periodic review of the criteria and
24 NAAQS for PM, and promulgated significant revisions to the NAAQS in 1997 (62 FR 38652,
25 July 18, 1997). Most significantly, EPA determined that although the PM NAAQS should
26 continue to focus on particles less than or equal to $10 \mu\text{m}$ in diameter, the fine and coarse
27 fractions of PM_{10} should be considered separately. New standards were added, using $\text{PM}_{2.5}$,
28 referring to particles with a nominal median aerodynamic diameter less than or equal to $2.5 \mu\text{m}$,
29 as the indicator for fine particles. The PM_{10} standards were retained for the purpose of
30 regulating the coarse fraction of PM_{10} (referred to as thoracic coarse particles or coarse-fraction
31 particles; generally including particles with a nominal median aerodynamic diameter greater than
32 $2.5 \mu\text{m}$ and less than or equal to $10 \mu\text{m}$, or $\text{PM}_{10-2.5}$). The EPA established two new $\text{PM}_{2.5}$
33 standards: an annual standard of $15 \mu\text{g}/\text{m}^3$, based on the 3-year average of annual arithmetic
34 mean $\text{PM}_{2.5}$ concentrations from single or multiple monitors sited to represent community-wide
35 air quality; and a 24-hour standard of $65 \mu\text{g}/\text{m}^3$, based on the 3-year average of the 98th
36 percentile of 24-hour $\text{PM}_{2.5}$ concentrations at each population-oriented monitor within an area.

1 Also, EPA established a new reference method for the measurement of PM_{2.5} in the ambient air
2 and rules for determining attainment of the new standards. To continue to address thoracic
3 coarse particles, the annual PM₁₀ standard was retained, while the form, but not the level, of the
4 24-hour PM₁₀ standard was revised to be based on the 99th percentile of 24-hour PM₁₀
5 concentrations at each monitor in an area. The EPA revised the secondary standards by making
6 them identical in all respects to the primary standards.

7 Following promulgation of the revised PM NAAQS in 1997, petitions for review were
8 filed by a large number of parties, addressing a broad range of issues. In May 1998, a three-
9 judge panel of the U.S. Court of Appeals for the District of Columbia Circuit issued an initial
10 decision that upheld EPA's decision to establish fine particle standards, holding that "the
11 growing empirical evidence demonstrating a relationship between fine particle pollution and
12 adverse health effects amply justifies establishment of new fine particle standards." *American*
13 *Trucking Associations v. EPA*, 175 F. 3d 1027, 1055-56 (D.C. Cir. 1999) (rehearing granted in
14 part and denied in part, 195 F. 3d 4 (D.C. Cir. 1999), affirmed in part and reversed in part,
15 *Whitman v. American Trucking Associations*, 531 U.S. 457 (2001). The panel also found "ample
16 support" for EPA's decision to regulate coarse particle pollution, but vacated the 1997 PM₁₀
17 standards, concluding in part that PM₁₀ is a "poorly matched indicator for coarse particulate
18 pollution" because it includes fine particles. *Id.* at 1053-55. Pursuant to the court's decision,
19 EPA removed the vacated 1997 PM₁₀ standards from the Code of Federal Regulations (CFR) (69
20 FR 45592, July 30, 2004) and deleted the regulatory provision [at 40 CFR section 50.6(d)] that
21 controlled the transition from the pre-existing 1987 PM₁₀ standards to the 1997 PM₁₀ standards.
22 The pre-existing 1987 PM₁₀ standards remained in place (65 FR 80776, December 22, 2000).
23 The Court also upheld EPA's determination not to establish more stringent secondary standards
24 for fine particles to address effects on visibility (175 F. 3d at 1027).

25 More generally, the panel held (over a strong dissent) that EPA's approach to
26 establishing the level of the standards in 1997, both for the PM and for the ozone (O₃) NAAQS
27 promulgated on the same day, effected "an unconstitutional delegation of legislative authority."
28 *Id.* at 1034-40. Although the panel stated that "the factors EPA uses in determining the degree of
29 public health concern associated with different levels of ozone and PM are reasonable," it
30 remanded the rule to EPA, stating that when EPA considers these factors for potential non-
31 threshold pollutants "what EPA lacks is any determinate criterion for drawing lines" to
32 determine where the standards should be set. Consistent with EPA's long-standing interpretation
33 and D.C. Circuit precedent, the panel also reaffirmed its prior holdings that in setting NAAQS
34 EPA is "not permitted to consider the cost of implementing those standards" *Id.* at 1040-41.

35 On EPA's petition for rehearing, the panel adhered to its position on these points.
36 *American Trucking Associations v. EPA*, 195 F. 3d 4 (D.C. Cir. 1999). The full Court of

1 Appeals denied EPA's request for rehearing en banc, with five judges dissenting. *Id.* at 13. Both
2 sides filed cross appeals on these issues to the United States Supreme Court, which granted
3 certiorari. In February 2001, the Supreme Court issued a unanimous decision upholding EPA's
4 position on both the constitutional and cost issues. *Whitman v. American Trucking Associations*,
5 531 U.S. 457, 464, 475-76. On the constitutional issue, the Court held that the statutory
6 requirement that NAAQS be "requisite" to protect public health with an adequate margin of
7 safety sufficiently cabined EPA's discretion, affirming EPA's approach of setting standards that
8 are neither more nor less stringent than necessary. The Supreme Court remanded the case to the
9 Court of Appeals for resolution of any remaining issues that had not been addressed in that
10 court's earlier rulings. *Id.* at 475-76. In March 2002, the Court of Appeals rejected all
11 remaining challenges to the standards, holding under the traditional standard of review that
12 EPA's PM_{2.5} standards were reasonably supported by the administrative record and were not
13 "arbitrary and capricious." *American Trucking Associations v. EPA*, 283 F. 3d 355, 369-72 (D.C.
14 Cir. 2002).

15 In October 1997, EPA published its plans for the next periodic review of the air quality
16 criteria and NAAQS for PM (62 FR 55201, October 23, 1997), including the 1997 PM_{2.5}
17 standards and the 1987 PM₁₀ standards. After CASAC and public review of several drafts, EPA's
18 National Center for Environmental Assessment (NCEA) finalized the *Air Quality Criteria*
19 *Document for Particulate Matter* (henceforth, AQCD or the "Criteria Document") in October
20 2004 (U.S. EPA, 2004) and OAQPS finalized an assessment document, *Particulate Matter*
21 *Health Risk Assessment for Selected Urban Areas* (Abt, 2005), and a "Staff Paper," *Review of the*
22 *National Ambient Air Quality Standards for Particulate Matter: Policy Assessment of Scientific*
23 *and Technical Information*, in December 2005 (U.S. EPA, 2005). In conjunction with their
24 review of the Staff Paper, CASAC provided advice to the Administrator on revisions to the PM
25 NAAQS (Henderson, 2005a). In particular, most CASAC PM Panel members favored revising
26 the level of the 24-hour PM_{2.5} primary standard in the range of 35 to 30 µg/m³ with a 98th
27 percentile form, in concert with revising the level of the annual PM_{2.5} standard in the range of 14
28 to 13 µg/m³ (Henderson, 2005a, p.7). For thoracic coarse particles, the Panel had reservations in
29 recommending a 24-hour PM_{10-2.5} primary standard, and agreed that there was a need for more
30 research on the health effects of thoracic coarse particles (Henderson, 2005b). With regard to
31 secondary standards, most Panel members strongly supported establishing a new, distinct
32 secondary PM_{2.5} standard to protect urban visibility (Henderson, 2005a, p. 9).

33 On January 17, 2006, EPA proposed to revise the primary and secondary NAAQS for PM
34 (71 FR 2620) and solicited comment on a broad range of options. Proposed revisions included:
35 revising the level of the 24-hour PM_{2.5} primary standard to 35 µg/m³; revising the form, but not
36 the level, of the annual PM_{2.5} primary standard by tightening the constraints on the use of spatial

1 averaging; replacing the 24-hour PM₁₀ primary standard with a 24-hour standard defined in
2 terms of a new indicator, PM_{10-2.5}¹⁰ set at a level of 70 µg/m³ based on the 3-year average of the
3 98th percentile of 24-hour PM_{10-2.5} concentrations; revoking the annual PM₁₀ primary standard;
4 and revising the secondary standards by making them identical in all respects to the proposed
5 suite of primary standards for fine and coarse particles.¹¹ Subsequent to the proposal, CASAC
6 provided additional advice to EPA in a letter to the Administrator requesting reconsideration of
7 CASAC's recommendations for both the primary and secondary PM_{2.5} standards as well as the
8 standards for thoracic coarse particles (Henderson, 2006a).

9 On October 17, 2006, EPA promulgated revisions to the PM NAAQS to provide
10 increased protection of public health and welfare (71 FR 61144). With regard to the primary and
11 secondary standards for fine particles, EPA revised the level of the 24-hour PM_{2.5} standard to 35
12 µg/m³, retained the level of the annual PM_{2.5} standard at 15 µg/m³, and revised the form of the
13 annual PM_{2.5} standard by adding further constraints on the optional use of spatial averaging. The
14 EPA revised the secondary standards for fine particles by making them identical in all respects to
15 the primary standards. With regard to the primary and secondary standards for thoracic coarse
16 particles, EPA retained the level and form of the 24-hour PM₁₀ standard (such that the standard
17 remained at a level of 150 µg/m³ with a one expected exceedance form), and revoked the annual
18 PM₁₀ standard. The EPA also established a new Federal Reference Method (FRM) for the
19 measurement of PM_{10-2.5} in the ambient air (71 FR 61212-13). Although the standards for
20 thoracic coarse particles were not defined in terms of a PM_{10-2.5} indicator, the new FRM for
21 PM_{10-2.5} was established to provide a basis for approving Federal Equivalent Methods (FEMs)
22 and to promote gathering scientific data to support future reviews of the PM NAAQS.

23 Following issuance of the final rule, CASAC articulated its concern that "EPA's final
24 rule on the NAAQS for PM does not reflect several important aspects of the CASAC's advice"
25 (Henderson et al, 2006b). With regard to the PM_{2.5} annual primary standard, CASAC expressed
26 serious concerns regarding the decision to retain the level of the standard at 15 µg/m³. With
27 regard to EPA's final decision to retain the 24-hour PM₁₀ standard for thoracic coarse particles,
28 CASAC acknowledged concerns associated with retaining this standard while recognizing the
29 need to have a standard in place to protect against effects associated with short-term exposures to
30 thoracic coarse particles. With regard to EPA's final decision to revise the secondary PM_{2.5}

¹⁰ This proposed indicator was qualified so as to include any ambient mix of PM_{10-2.5} dominated by particles generated by high-density traffic on paved roads, industrial sources, and construction sources, and to exclude any ambient mix of particles dominated by rural windblown dust and soils and agricultural and mining sources.

¹¹ In recognition of an alternative view expressed by most members of the CASAC PM Panel, the Agency also solicited comments on a subdaily (4- to 8-hour averaging time) secondary PM_{2.5} standard to address visibility impairment, within the range of 20 to 30 µg/m³ and with a form within the range of the 92nd to 98th percentile (71 FR 2685).

1 standards to be identical in all respects to the revised primary PM_{2.5} standards, CASAC
2 expressed concerns that CASAC's advice to establish a distinct secondary standard for fine
3 particles to address visibility impairment was not followed.

4 **1.2.3 Litigation Related to the 2006 PM Standards**

5 Several parties filed petitions for review following promulgation of the revised PM
6 NAAQS in 2006. These petitions addressed the following issues: (1) selecting the level of the
7 primary annual PM_{2.5} standard; (2) retaining PM₁₀ as the indicator of a standard for thoracic
8 coarse particles, retaining the level and form of the 24-hour PM₁₀ standard, and revoking the
9 PM₁₀ annual standard; and (3) setting the secondary PM_{2.5} standards identical to the primary
10 standards. On February 24, 2009, the U.S. Court of Appeals for the District of Columbia Circuit
11 issued its opinion in the case *American Farm Bureau Federation v. EPA*, 559 F. 3d 512 (D.C.
12 Cir. 2009). The court remanded the primary annual PM_{2.5} NAAQS to EPA because EPA failed
13 to adequately explain why the standard provided the requisite protection from both short- and
14 long-term exposures to fine particles, including protection for at-risk populations. *American*
15 *Farm Bureau Federation v. EPA*, 559 F. 3d 512, (D.C. Cir. 2009). With regard to the standards
16 for PM₁₀, the court upheld EPA's decisions to retain the 24-hour PM₁₀ standard to provide
17 protection from thoracic coarse particle exposures and to revoke the annual PM₁₀ standard.
18 *American Farm Bureau Federation* at 533-38. With regard to the secondary PM_{2.5} standards, the
19 court remanded the standards to EPA because the Agency failed to adequately explain why
20 setting the secondary PM standards identical to the primary standards provided the required
21 protection for public welfare, including protection from visibility impairment.

22 The decisions of the court with regard to these three issues are discussed in chapters 2, 3
23 and 4, respectively. The EPA is responding to the court's remands as part of the current review
24 of the PM NAAQS.

25 **1.2.4 Current PM NAAQS Review**

26 The EPA initiated the current review of the air quality criteria for PM in June 2007 with a
27 general call for information (72 FR 35462, June 28, 2007). In July 2007, EPA held two "kick-
28 off" workshops on the primary and secondary PM NAAQS, respectively (72 FR 34003 and
29 34005, June 20, 2007).¹² These workshops provided an opportunity for a public discussion of
30 the key policy-relevant issues around which EPA would structure this PM NAAQS review and

¹² See workshop materials <http://www.regulations.gov/search/Regs/home.html#home> Docket ID numbers EPA-HQ-OAR-2007-0492-008; EPA-HQ-OAR-2007-0492-009; EPA-HQ-OAR-2007-0492-010; and EPA-HQ-OAR-2007-0492-012.

1 the most meaningful new science that would be available to inform our understanding of these
2 issues.

3 Based in part on the workshop discussions, EPA developed a draft IRP outlining the
4 schedule, process, and key policy-relevant questions that would guide the evaluation of the air
5 quality criteria for PM and the review of the primary and secondary PM NAAQS (US EPA,
6 2007). On November 30, 2007, EPA held a consultation with CASAC on the draft IRP (72 FR
7 63177, November 8, 2007), which included the opportunity for public comment. The final IRP
8 (US EPA, 2008a) incorporated comments from CASAC (Henderson, 2008) and the public on the
9 draft plan as well as input from senior Agency managers.¹³

10 As part of the process of preparing the PM ISA, NCEA hosted a peer review workshop in
11 June 2008 on preliminary drafts of key ISA chapters (73 FR 30391, May 27, 2008). The first
12 external review draft ISA (US EPA, 2008b) was reviewed by CASAC and the public at a
13 meeting held in April 2009 (74 FR 2688, February 19, 2009). Based on CASAC and public
14 comments, NCEA prepared a second draft ISA (US EPA, 2009b), which was reviewed by
15 CASAC and the public at a meeting held on October 5-6, 2009 (74 FR 46586, September 10,
16 2009). Based on CASAC and public comments, NCEA prepared the final ISA (US EPA,
17 2009a; 74 FR 66353, December 15, 2009).

18 In preparing the Risk and Exposure Assessment (REA) documents that build on the
19 scientific evidence presented in the ISA, OAQPS released two planning documents: *Particulate*
20 *Matter National Ambient Air Quality Standards: Scope and Methods Plan for Health Risk and*
21 *Exposure Assessment* and *Particulate Matter National Ambient Air Quality Standards: Scope*
22 *and Methods Plan for Urban Visibility Impact Assessment* (henceforth, Scope and Methods
23 Plans, US EPA, 2009c,d). These planning documents outlined the scope and approaches that
24 staff planned to use in conducting quantitative assessments as well as key issues that would be
25 addressed as part of the assessments. In designing and conducting the initial health risk and
26 visibility impact assessments, we considered CASAC comments (Samet 2009a,b) on the Scope
27 and Methods Plans made during an April 2009 consultation (74 FR 7688, February 19, 2009) as
28 well as public comments. Two draft assessment documents, *Risk Assessment to Support the*
29 *Review of the PM_{2.5} Primary National Ambient Air Quality Standards: External Review Draft -*
30 *September 2009* (US EPA 2009e) and *Particulate Matter Urban-Focused Visibility Assessment -*

¹³ The process followed in this review varies from the NAAQS review process described in section 1.1 of the IRP (US EPA, 2008a). On May 21, 2009, EPA Administrator Jackson called for key changes to the NAAQS review process including reinstating a policy assessment document that contains staff analyses of the scientific bases for alternative policy options for consideration by senior Agency management prior to rulemaking. In conjunction with this change, EPA will no longer issue a policy assessment in the form of an advance notice of proposed rulemaking (ANPR) as discussed in the IRP. For more information on the overall process followed in this review including a description of the major elements of the process for reviewing NAAQS see Jackson (2009).

1 *External Review Draft* - September 2009 (US EPA, 2009f) were reviewed by CASAC and the
2 public at a meeting held on October 5-6, 2009. Based on CASAC (Samet 2009c,d) and public
3 comments, OAQPS staff revised these draft documents and released second draft assessment
4 documents (US EPA, 2010d,e) in January and February 2010 (75 FR 4067, January 26, 2010) for
5 CASAC and public review at a meeting held on March 10-11, 2010. Based on CASAC (Samet,
6 2010a, b) and public comments on the second draft assessment documents, we revised these
7 documents and released final assessment documents in June 2010 (US EPA, 2010a,b).

8 A preliminary draft PA (US EPA, 2009g) was released in September 2009 for
9 informational purposes and to facilitate discussion with CASAC at the October 5-6, 2009
10 meeting on the overall structure, areas of focus, and level of detail to be included in the PA.
11 CASAC's comments on the preliminary draft PA encouraged the development of a document
12 focused on the key policy-relevant issues that draws from and is not repetitive of information in
13 the ISA and REAs. These comments were considered in developing a first draft PA (US EPA,
14 2010c) that built upon the information presented and assessed in the final ISA and two second
15 draft REAs. The EPA presented an overview of the first draft PA at a CASAC meeting on
16 March 10, 2010. CASAC and public review of the first draft PA was discussed during public
17 teleconferences on April 8-9, 2010 (75 FR 8062, February 23, 2010) and May 7, 2010 (75
18 FR19971, April 16, 2010).

19 CASAC (Samet, 2010c) and public comments on the first draft PA were considered in
20 developing this second draft PA which will be reviewed by CASAC at an upcoming meeting
21 scheduled for July 26-27, 2010 (75 FR 32763, June 9, 2010). We will consider CASAC and
22 public comments on this second draft PA in preparing a final PA which will include final staff
23 conclusions related to the adequacy of the current PM standards and alternative standards for
24 consideration that are supported by the currently available scientific evidence and quantitative
25 assessments. We plan to release the final PA in September 2010.

26 **1.3 GENERAL APPROACH AND ORGANIZATION OF THIS DOCUMENT**

27 This second draft PA includes staff's evaluation of the policy implications of the
28 scientific assessment of the evidence presented and assessed in the ISA and the results of
29 quantitative assessments based on that evidence presented and assessed in the REAs. Taken
30 together, this information informs staff conclusions and the identification of policy options for
31 consideration in addressing public health and welfare effects associated with exposure to ambient
32 PM.

33 Since the last review, much new information is now available on PM air quality and
34 human health effects directly in terms of PM_{2.5} and, to a much more limited degree, PM_{10-2.5} and
35 ultrafine particles (UFPs). Since the purpose of this review is to evaluate the adequacy of the

1 current standards, which separately address fine and thoracic coarse particles, staff is focusing
2 this policy assessment and associated quantitative analyses primarily on the evidence related
3 directly to PM_{2.5} and PM_{10-2.5}. In so doing, we are considering PM₁₀-related evidence primarily
4 to help inform our understanding of key issues and to help interpret and provide context for
5 understanding the public health and welfare impacts of ambient fine and coarse particles. We are
6 also considering the currently available evidence related to UFPs as well as PM_{2.5} components to
7 aid in considering whether there is support to consider standards with a different size fraction
8 and/or distinct standards focused on regulating specific PM_{2.5} components or categories of fine
9 particle sources.

10 Following this introductory chapter, this document is organized into two main parts:
11 review of the primary PM NAAQS (chapters 2 and 3) and review of the secondary PM NAAQS
12 (chapters 4 and 5). Chapters 2 and 3 present staff observations and conclusions related to review
13 of the primary standards for fine and thoracic coarse particles, respectively. Each chapter
14 includes background information on the rationale for previous reviews and the policy assessment
15 approaches followed in the current review, focusing on evidence-based considerations and, as
16 appropriate, quantitative risk-based considerations. Staff conclusions are presented with regard
17 to the adequacy of the current primary standards and potential alternative primary standards for
18 consideration, in terms of indicators, averaging times, forms, and levels. Chapter 4 focuses on
19 PM-related visibility impairment, and presents staff observations and conclusions with regard to
20 the adequacy of the current standards and potential distinct secondary standards for
21 consideration, in terms of alternative indicators, averaging times, forms, and levels. Chapter 5
22 focuses on other PM-related welfare effects, including effects on climate, ecological effects, and
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2 REVIEW OF THE PRIMARY STANDARDS FOR FINE PARTICLES

This chapter presents staff conclusions with regard to the adequacy of the current suite of primary PM_{2.5} standards and the alternative primary standards for fine particles that are appropriate for consideration in this review. Our assessment of these issues is framed by a series of key policy-relevant questions, which expand upon those presented at the outset of this review in the IRP. Answers to these questions will inform decisions by the Administrator on whether, and if so how, to revise the current suite of primary fine particle standards.

Staff notes that final decisions regarding the primary standards must draw upon scientific information and analyses about health effects and risks, as well as judgments made about how to deal with the uncertainties that are inherent in the scientific evidence and analyses. Ultimately, the final decisions are largely public health policy judgments. Our approach to informing these judgments recognizes that the available health effects evidence generally reflects a continuum consisting of ambient levels at which scientists generally agree that health effects occur through lower levels at which the likelihood and magnitude of the response become increasingly uncertain.

Our approach for reviewing the primary standards for fine particles is presented in section 2.1. Staff conclusions regarding the adequacy of the current suite of primary PM_{2.5} standards are presented in section 2.2, focusing on both evidence-based and quantitative risk-based considerations. Section 2.3 presents our conclusions with respect to alternative fine particle standards, focusing on each of the basic elements of the standards: pollutant indicator (section 2.3.1), averaging time (section 2.3.2), form (section 2.3.3), and level (section 2.3.4). Section 2.4 summarizes staff conclusions on the primary fine particle standards. Key uncertainties and areas for future research and data collection efforts are included in section 2.5.

2.1 APPROACH

Staff's approach for reviewing the current primary PM_{2.5} standards, which involves translating scientific and technical information into the basis for addressing key policy-relevant questions, takes into consideration the approaches used in previous PM NAAQS reviews and the court's remand of the primary annual PM_{2.5} standard set in 2006. The past and current approaches described below are all based most fundamentally on using information from epidemiological studies to inform the selection of PM standards that, in the Administrator's judgment, protect public health with an adequate margin of safety. Such information can be in the form of air quality distributions over which health effect associations have been observed, or in the form of concentration-response (C-R) functions that support quantitative risk assessment. However, evidence- and risk-based approaches using information from epidemiological studies

1 to inform decisions on PM standards are complicated by the recognition that no population
2 threshold, below which it can be concluded with confidence that PM-related effects do not occur,
3 can be discerned from the available evidence. As a result, any approach to reaching decisions on
4 what standards are appropriate necessarily requires judgments about how to translate the
5 information available from the epidemiological studies into a basis for appropriate standards,
6 which includes consideration of how to weigh the uncertainties in reported associations across
7 the distributions of PM concentrations in the studies or in quantitative estimates of risk. Such
8 approaches are consistent with setting standards that are neither more nor less stringent than
9 necessary, recognizing that a zero-risk standard is not required by the CAA.

10 **2.1.1 Approaches Used in Previous Reviews**

11 **2.1.1.1 Review Completed in 1997**

12 In setting the 1997 primary PM_{2.5} annual and 24-hour standards, the Agency relied
13 primarily on an evidence-based approach that focused on epidemiological evidence, especially
14 from short-term exposure studies of fine particles judged to be the strongest evidence at that
15 time. The EPA did not place much weight on quantitative risk estimates from the very limited
16 risk assessment conducted, but did conclude that the risk assessment results confirmed the
17 general conclusions drawn from the epidemiological evidence that a serious public health
18 problem was associated with ambient PM levels allowed under the then current PM₁₀ standards
19 (62 FR 38665/1, July 18, 1997).

20 The EPA considered the epidemiological evidence and data on air quality relationships
21 to set an annual PM_{2.5} standard that was intended to be the “generally controlling” standard; i.e.,
22 the primary means of lowering both long- and short-term ambient concentrations of PM_{2.5}.¹ In
23 conjunction with the annual standard, EPA also established a 24-hour PM_{2.5} standard to provide
24 supplemental protection against days with high peak concentrations, localized “hotspots,” and
25 risks arising from seasonal emissions that might not be well controlled by a national annual
26 standard (62 FR 38669/3). Recognizing that there are various ways to combine two standards to
27 achieve an appropriate degree of public health protection, such as an approach that only
28 considered short- and long-term exposure evidence, analyses, and standards independently, EPA
29 concluded that the selected approach based on a generally controlling annual standard was the

¹ In so doing, EPA noted that because an annual standard would focus control programs on annual average PM_{2.5} concentrations, it would not only control long-term exposure levels, but would also generally control the overall distribution of 24-hour exposure levels, resulting in fewer and lower 24-hour peak concentrations. Alternatively, a 24-hour standard that focused controls on peak concentrations could also result in lower annual average concentrations. Thus, EPA recognized that either standard could provide some degree of protection from both short- and long-term exposures, with the other standard serving to address situations where the daily peaks and annual averages are not consistently correlated (62 FR 38669).

1 most effective and efficient approach. This conclusion was based in part on one of the key
2 observations from the quantitative risk assessment that most of the aggregated annual risk
3 associated with short-term exposures through a year results from the large number of days during
4 which the 24-hour average concentrations are in the low- to mid-range, well below the peak 24-
5 hour concentrations. As a result, lowering a wide range of ambient 24-hour PM_{2.5} concentrations
6 by means of a generally controlling annual standard, as opposed to focusing on control of peak
7 24-hour concentrations, was determined to be the most effective and efficient way to reduce total
8 population risk (62 FR 38670 to 38671).

9 In setting the level of the annual standard in 1997, EPA first determined a level for the
10 annual standard based on the short-term exposure studies, and then considered whether the key
11 long-term exposure studies suggested the need for a lower level. While recognizing that health
12 effects may occur over the full range of concentrations observed in the studies, EPA concluded
13 that the strongest evidence for short-term PM_{2.5} exposure-related effects occurs at concentrations
14 near the long-term (e.g., annual) average in the short-term exposure studies. The EPA selected a
15 level for the annual standard at or below the long-term mean concentrations in studies providing
16 evidence of associations with short-term exposures, placing greatest weight on those short-term
17 exposure studies that reported clearly statistically significant associations with mortality and
18 morbidity effects (62 FR 38676/1). Further consideration of the average PM_{2.5} concentrations
19 across the cities in the key long-term exposure studies of mortality and respiratory effects in
20 children did not provide a basis for establishing a lower annual standard level. Because the
21 annual standard level selected was below the range of annual concentrations most strongly
22 associated with both short- and long-term exposure effects at that time, and because even small
23 changes in annual means in this concentration range could make a significant difference in
24 overall risk reduction and total population exposures, EPA concluded that this standard would
25 provide an adequate margin of safety against effects observed in these epidemiological studies
26 (62 FR 68676/3).

27 The selection of the level of the annual standard was done in conjunction with having
28 first selected the form of the annual standard to be based on the concentration measured at a
29 single monitor sited to represent community-wide air quality, or a value resulting from an
30 average of measurements from multiple community-wide air quality monitoring sites that met
31 specific criteria and constraints (“spatial averaging”). This decision emphasized consistency
32 with the types of air quality measurements that were used in the relevant epidemiological studies.
33 In reaching this decision, EPA recognized the importance of ensuring that spatial averaging
34 would not result in inequities in the level of protection provided by the PM_{2.5} standards in some
35 areas. Because the annual standard, defined in terms of single or averaged community-wide air
36 quality monitoring sites, could not be expected to offer an adequate margin of safety against the

1 effects of all potential short-term exposures in areas with strong local or seasonal sources that
2 could not be directly evaluated in the epidemiological studies, EPA set the level of the 24-hour
3 standard to supplement the control afforded by the annual standard based on air quality
4 relationships between annual and 24-hour concentrations. This approach was intended to
5 provide an adequate margin of safety against infrequent or isolated peak concentrations that
6 could occur in areas that attain the annual standard (62 FR 38677).

7 **2.1.1.2 Review Completed in 2006**

8 In 2006, EPA used a different evidence-based approach to assess the appropriateness of
9 the levels of the 24-hour and annual PM_{2.5} standards. Based on an expanded body of
10 epidemiological evidence that was stronger and more robust, including both short- and long-term
11 exposure studies, the Administrator decided that using evidence of effects associated with
12 periods of exposure that were most closely matched to the averaging time of each standard was
13 the most appropriate public health policy approach for evaluating the scientific evidence to
14 inform selecting the level of each standard. Thus, the Administrator relied upon evidence from
15 the short-term exposure studies as the principal basis for selecting the level of the 24-hour PM_{2.5}
16 standard that would protect against effects associated with short-term exposures. The
17 Administrator relied upon evidence from long-term exposure studies as the principal basis for
18 selecting the level of the annual PM_{2.5} standard that would protect against effects associated with
19 long-term exposures.

20 With respect to quantitative risk-based considerations, the Administrator determined that
21 the estimates of risks likely to remain upon attainment of the 1997 suite of PM_{2.5} standards were
22 indicative of risks that could be reasonably judged important from a public health perspective,
23 and, thus, supported revision of the standards. However, the Administrator judged that the
24 quantitative risk assessment had important limitations and did not provide an appropriate basis
25 for selecting the levels of the revised standards (71 FR 61174/1-2). The Administrator more
26 heavily weighed the implications of the uncertainties associated with the quantitative risk
27 assessment than the Clean Air Scientific Advisory Committee (CASAC) did in their comments
28 on the proposed rulemaking, where CASAC stated, “[w]hile the risk assessment is subject to
29 uncertainties, most of the PM Panel found EPA’s risk assessment to be of sufficient quality to
30 inform its recommendations...The risk analyses indicated that the uncertainties would increase
31 rapidly below an annual level of 13 µg/m³ – and that was the basis for the PM Panel’s
32 recommendation of 13 µg/m³ as the lower bound for the annual PM_{2.5} standard level”
33 (Henderson, 2006, p.3).

34 With regard to the primary annual PM_{2.5} standard, the Administrator placed the greatest
35 weight on the long-term means of the concentrations associated with mortality effects in two key

1 long-term exposure studies in the record, the American Cancer Society (ACS) and Harvard Six
2 Cities studies (71 FR at 61172 to 61177). Important validation and reanalyses of the original
3 studies provided “evidence of generally robust associations and provide[d] a basis for greater
4 confidence in the reported associations than in the last review,” and the extended ACS study
5 provided “new evidence of mortality related to lung cancer and further substantiate[d] the
6 statistically significant associations with cardiorespiratory-related mortality observed in the
7 original studies” (71 FR 61172/1-2). The Administrator also recognized the availability of long-
8 term exposure studies that provided evidence of respiratory morbidity, including changes in lung
9 function measurements and decreased growth in lung function as reported in the 24-Cities study
10 and the Southern California Children’s Health Study, respectively (Dockery et al. 1996,
11 Gauderman et al. 2002). In retaining the level of the annual standard at $15 \mu\text{g}/\text{m}^3$, the
12 Administrator selected a level that was “appreciably below” the long-term average
13 concentrations reported in the key long-term mortality studies and “basically at the same level”
14 as the long-term average concentrations in the long-term respiratory morbidity studies. In the
15 judgment of the Administrator, the two long-term respiratory morbidity studies did “not warrant
16 setting a lower level for the annual standard than the level warranted based on the key mortality
17 studies” (71 FR 61176/3).

18 In considering the form of the primary annual $\text{PM}_{2.5}$ standard, the Administrator
19 strengthened the standard by tightening the criteria for use of spatial averaging. Based on a
20 much larger set of $\text{PM}_{2.5}$ air quality data than was available in the 1997 review, analyses were
21 conducted concerning the potential for disproportionate impacts on potentially vulnerable
22 subpopulations. These analyses suggested that “the highest concentrations in an area tend to be
23 measured at monitors located in areas where the surrounding population [was] more likely to
24 have lower education and income levels, and higher percentages of minority populations” (71 FR
25 61166/2, see also US EPA, 2005, section 5.3.6.1; Schmidt et al., 2005, Attachment A/Analysis
26 7).²

27 In revising the level of the 24-hour $\text{PM}_{2.5}$ standard from $65 \mu\text{g}/\text{m}^3$ to $35 \mu\text{g}/\text{m}^3$, the
28 Administrator placed greatest weight on the much expanded body of epidemiological evidence
29 from U.S. and Canadian short-term $\text{PM}_{2.5}$ exposure studies with more reliable air quality data
30 that was reanalyzed to address statistical modeling issues. The Administrator recognized that

² As summarized in footnote 29 at 71 FR 61166/2, the 2004 AQCD noted that some epidemiologic studies, most notably the ACS study of associations between long-term $\text{PM}_{2.5}$ exposure and mortality, reported larger effect estimates in the cohort subgroup with lower education levels (US EPA, 2004, p 8-103). The 2004 AQCD also noted that lower education level may be a marker for lower socioeconomic status (SES) that may be related to increased vulnerability to the effects of fine particle exposures, for example, as a result of greater exposure from proximity to sources such as roadways and industry, as well as other factors such as poorer health status and limited access to health care (US EPA, 2004, section 9.2.4.5).

1 these studies provided no evidence of clear effect thresholds or lowest-observed effect levels.
2 Nonetheless, in focusing on the 98th percentile air quality values in these studies, the
3 Administrator sought to establish a standard level that would require improvements in air quality
4 generally in areas in which the distribution of daily short-term exposure to PM_{2.5} could
5 reasonably be expected to be associated with serious health effects. The Administrator
6 concluded that although future air quality improvement strategies in any particular area were not
7 yet defined, most such strategies were likely to move a broad distribution of PM_{2.5} air quality
8 values in an area lower, resulting in reductions in risk associated with exposures to PM_{2.5} levels
9 across a wide range of concentrations and not just at the 98th percentile concentrations (71 FR
10 61168/3).

11 **2.1.2 Remand of Primary Annual PM_{2.5} Standard**

12 As noted above in section 1.2.4, several parties filed petitions for review following
13 promulgation of the revised PM NAAQS in 2006. These petitions challenged several aspects of
14 the final rule including the selection of the level of the primary PM_{2.5} annual standard. More
15 specifically, petitioners representing public health and environmental groups (American Lung
16 Association, Environmental Defense, and the National Parks Conservation Association) and
17 several states and state agencies argued that the decision to retain the level of the annual PM_{2.5}
18 standard at 15 µg/m³ was “arbitrary, capricious, an abuse of discretion, or otherwise not in
19 accordance with law.” 42 U.S.C. 7607(d)(9). The primary 24-hour PM_{2.5} standard was not
20 challenged by any of the litigants and, thus, not considered in the court’s review and final
21 decision.

22 On judicial review, the D.C. Circuit remanded the primary annual PM_{2.5} NAAQS to EPA
23 because the Agency failed to adequately explain why the annual standard provided the requisite
24 protection from both short- and long-term exposures to fine particles including protection for
25 susceptible populations. *American Farm Bureau Federation v. EPA*, 559 F. 3d 512 (D.C. Cir.
26 2009). With respect to human health protection from short-term PM_{2.5} exposures, the court
27 considered the different approaches used by EPA in the 1997 and 2006 PM NAAQS decisions,
28 as summarized above. The court found that EPA failed to adequately explain why a 24-hour
29 PM_{2.5} standard by itself would provide the protection needed from short-term exposures and
30 remanded the annual PM_{2.5} standard to EPA “for further consideration of whether it is set at a
31 level requisite to protect the public health while providing an adequate margin of safety from the
32 risk of short-term exposures to PM_{2.5}.” *American Farm Bureau Federation*, 559 F. 3d at 520-24.

33 With respect to protection from long-term exposure to fine particles, the court found that
34 EPA failed to adequately explain how the primary annual PM_{2.5} standard provided an adequate
35 margin of safety for children and other susceptible populations. The court found that EPA did

1 not provide a reasonable explanation of why certain morbidity studies, including a study of
2 children in Southern California showing lung damage associated with long-term PM_{2.5} exposure
3 (Gauderman et.al, 2000) and a multi-city study (24-Cities Study) evaluating decreased lung
4 function in children associated with long-term PM_{2.5} exposures (Raizenne et al., 1996), did not
5 call for a more stringent annual PM_{2.5} standard. Id. at 522-23. Specifically, the court found that:

6
7 EPA was unreasonably confident that, even though it relied solely upon long-term
8 mortality studies, the revised standard would provide an adequate margin of safety with
9 respect to morbidity among children. Notably absent from the final rule, moreover, is
10 any indication of how the standard will adequately reduce risk to the elderly or to those
11 with certain heart or lung diseases despite (a) the EPA's determination in its proposed
12 rule that those subpopulations are at greater risk from exposure to fine particles and (b)
13 the evidence in the record supporting that determination. Id. at 525.
14

15 In remanding the primary annual standard for reconsideration, the court did not vacate the
16 annual standard. Id. at 530.

17 **2.1.3 Current Approach**

18 This review is founded on an assessment of a much expanded body of epidemiological
19 evidence, more extensive air quality data and analyses, and a more comprehensive risk
20 assessment relative to the information available in past reviews, as presented in the ISA and RA.
21 As a result, staff's approach to reaching conclusions about the adequacy of the current suite of
22 PM_{2.5} standards and potential alternative standards that are appropriate for consideration is
23 broader and more integrative than in past reviews. Our approach also reflects consideration of
24 the issues raised by the court in its remand of the primary annual PM_{2.5} standard, since decisions
25 made in this review, and the rationales for those decisions, will comprise the Agency's response
26 to the remand.

27 Our approach in this second draft PA takes into account both evidence-based and risk-
28 based considerations, and the uncertainties related to both types of information, as well as advice
29 from CASAC and public comments on the first draft PA. In so doing, we are seeking to provide
30 as broad an array of policy options as is supportable by the available information, recognizing
31 that the selection of a specific approach to reaching final decisions on the primary PM_{2.5}
32 standards will reflect the judgments of the Administrator as to what weight to place on the
33 various approaches and types of information presented in the final PA.

34 We believe it is most appropriate to consider the protection against PM_{2.5}-related
35 mortality and morbidity effects, associated with both long- and short-term exposures, afforded by
36 the annual and 24-hour standards taken together, as was done in the 1997 review, rather than to
37 consider each standard separately, as was done in the 2006 review. This approach reflects the

1 recognition that changes in PM_{2.5} air quality designed to meet an annual standard would likely
2 result not only in lower annual average concentrations but also in fewer and lower peak 24-hour
3 concentrations. It also reflects the recognition that setting the annual standard to be the
4 “generally controlling” standard, and setting the 24-hour standard to provide supplemental
5 protection, continues to be seen as the most effective and efficient way to reduce total population
6 risk associated with both long- and short-term exposures and provide requisite protection in areas
7 across the country. While we recognize that changes designed to meet a 24-hour standard would
8 result not only in fewer and lower peak 24-hour concentrations but also in lower annual average
9 concentrations, we note that simulated changes in annual average concentrations associated with
10 just meeting alternative 24-hour standards are highly sensitive to how air quality changes are
11 simulated and are more uncertain and variable across areas. The extent to which these two
12 standards are interrelated in any given area depends in large part on the relative levels of the
13 standards, the peak-to-mean ratios that characterize air quality patterns in an area, and whether
14 changes in air quality designed to meet a given suite of standards are likely to be of a more
15 regional or more localized nature.

16 Our consideration of the protection afforded by the current and alternative suites of
17 standards focuses on PM_{2.5}-related health effects associated with long-term exposures, for which
18 quantitative estimates of risks to public health are appreciably larger, and also considers effects
19 and estimated risks associated with short-term exposures. In both cases, we place greatest
20 weight on associations that have been judged in the ISA to be causal and likely causal, while also
21 considering associations judged to be suggestive of a causal relationship or that focus on specific
22 susceptible populations. We focus on studies conducted in the U.S. and Canada, as studies in
23 other countries reflect air quality and exposure patterns that are not necessarily typical of the
24 U.S.,³ and place relatively greater weight on statistically significant associations that yield
25 relatively more precise effect estimates and that are judged to be robust to confounding by other
26 air pollutants. In the case of short-term exposure studies, we place greatest weight on large
27 multi-city studies, while also considering associations in single-city studies.

28 In translating information from epidemiological studies into the basis for reaching staff
29 conclusions on the adequacy of the current suite of standards (section 2.2), we have considered a
30 number of factors. As an initial matter, we have considered the extent to which the currently
31 available evidence and related uncertainties strengthens or calls into question conclusions from
32 the last review regarding associations between fine particle exposures and health effects. We
33 have also considered evidence on susceptible populations and potential impacts on such
34 populations. Further, we have explored the extent to which PM_{2.5}-related health effects have

³ Nonetheless, we recognize the importance of all studies, including international studies, in the ISA’s assessment of the weight of the evidence that informs causality determinations.

1 been observed in areas where air quality distributions extend to lower levels than previously
2 reported or in areas that would likely meet the current suite of standards.

3 In translating information from epidemiological studies into the basis for reaching staff
4 conclusions on alternative standard levels for consideration (section 2.3.4), we have explored
5 various approaches that reflect the absence of discernible thresholds in the C-R functions from
6 long- and short-term exposure studies. In so doing, we recognize that there is no single criterion
7 that comprises the “correct” approach, but rather there are various approaches that are reasonable
8 to consider. Identifying the implications of various approaches in reaching conclusions on the
9 range of alternative standard levels that is appropriate to consider can help inform decision
10 making, which will necessarily also take into account the limitations and uncertainties in the
11 evidence and public health policy judgments as to the degree of health protection that is to be
12 achieved.

13 In reaching staff conclusions on the range of *annual standard levels* that is appropriate to
14 consider in setting a generally controlling annual standard intended to provide protection from
15 effects associated with long- and short-term exposures, we have explored two types of
16 approaches. One such approach looks directly at confidence bounds on the C-R relationships
17 from both long- and short-term exposure studies. The other approach is based on the use of
18 statistical metrics that characterize air quality distributions from multi-city studies. We
19 recognize that these two types of approaches are intrinsically related, since C-R functions are
20 most certain across a concentration range with the greatest data density, in which the bulk of the
21 data exist.

22 In considering the first type of approach, we have explored the extent to which analyses
23 characterizing confidence bounds on C-R relationships have been published for studies of health
24 effects associated with long- or short-term exposures to PM_{2.5}. Such analyses could potentially
25 be used to characterize a concentration below which uncertainty in a C-R relationship
26 substantially increases or is judged to be indicative of an unacceptable degree of uncertainty
27 about the existence of a continuing C-R relationship.

28 In considering statistical air quality metrics, we first recognize that there are two air
29 quality distributions that are relevant to consider. One is the distribution based on concentrations
30 averaged across ambient monitors within each area included in a given study, which we refer to
31 as a composite monitor distribution. This is the air quality data typically used in analyses of C-R
32 relationships presented in epidemiological studies of both long- and short-term exposures, in
33 both multi- and single-city studies. The second relevant air quality distribution is based on
34 concentrations measured at the monitor within each area that records the highest concentration.
35 This maximum monitor distribution is relevant because this is the distribution that is generally
36 used to determine whether a given standard is met in an area, and to determine the extent to

1 which ambient PM_{2.5} concentrations need to be reduced in order to bring an area into attainment
2 with the standard. For composite monitor distributions, we have compiled distributional
3 statistics drawn from multi-city studies, as supplemented by information obtained from study
4 authors. For maximum monitor distributions, we were able to obtain information for a much
5 more limited set of studies from EPA's public air quality data base, the Air Quality System
6 (AQS).⁴

7 More specifically, to the extent available, we have compiled data for long- and short-term
8 exposure multi-city studies on the long-term mean PM_{2.5} concentration over the time period of
9 each study, as well as the standard deviation⁵ and lower percentiles of the distribution (e.g., 25th
10 and 10th percentiles). We have explored the implications of considering these various air quality
11 metrics as a basis for reaching staff conclusions as to the range of concentrations that is
12 appropriate to consider for alternative annual standard levels. This approach also includes a
13 broader consideration of uncertainties related to the C-R relationships from long- and short-term
14 exposure multi-city studies, most notably uncertainties related to our currently limited
15 understanding of the heterogeneity of relative risk estimates in areas across the country, with
16 may be attributed in part to the potential for different components within the mix of ambient fine
17 particles to differentially contribute to health effects observed in the studies and to exposure-
18 related factors.

19 In reaching staff conclusions on alternative *24-hour standard levels* that are appropriate
20 to consider in setting a 24-hour standard intended to supplement the protection afforded by
21 potential alternative annual standards, we have focused on comparing 24-hour and annual air
22 quality statistics for areas across the U.S. Consistent with the aim of the approach used in 1997,
23 such a standard would be intended to provide supplemental protection against the effects of
24 short-term PM_{2.5} exposures, especially in areas with high peak-to-mean ratios, possibly
25 associated with strong local or seasonal sources, or for potential PM_{2.5}-related effects that may be
26 associated with shorter-than-daily exposure periods. By considering information on peak-to-
27 mean air quality ratios, we reach conclusions as to the level of a 24-hour standard would be
28 appropriate to consider for this purpose.

29 Based on the evidence-based considerations outlined above, we then develop integrated
30 conclusions with regard to alternative suites of standards, including both annual and 24-hour
31 standards that we believe are appropriate for consideration in this review based on the currently
32 available evidence and air quality information. In so doing, we discuss the roles that each
33 standard might be expected to play in the protection afforded by alternative suites of standards.

⁴ Focusing on multi-city studies, AQS data was available from 1999 and onwards.

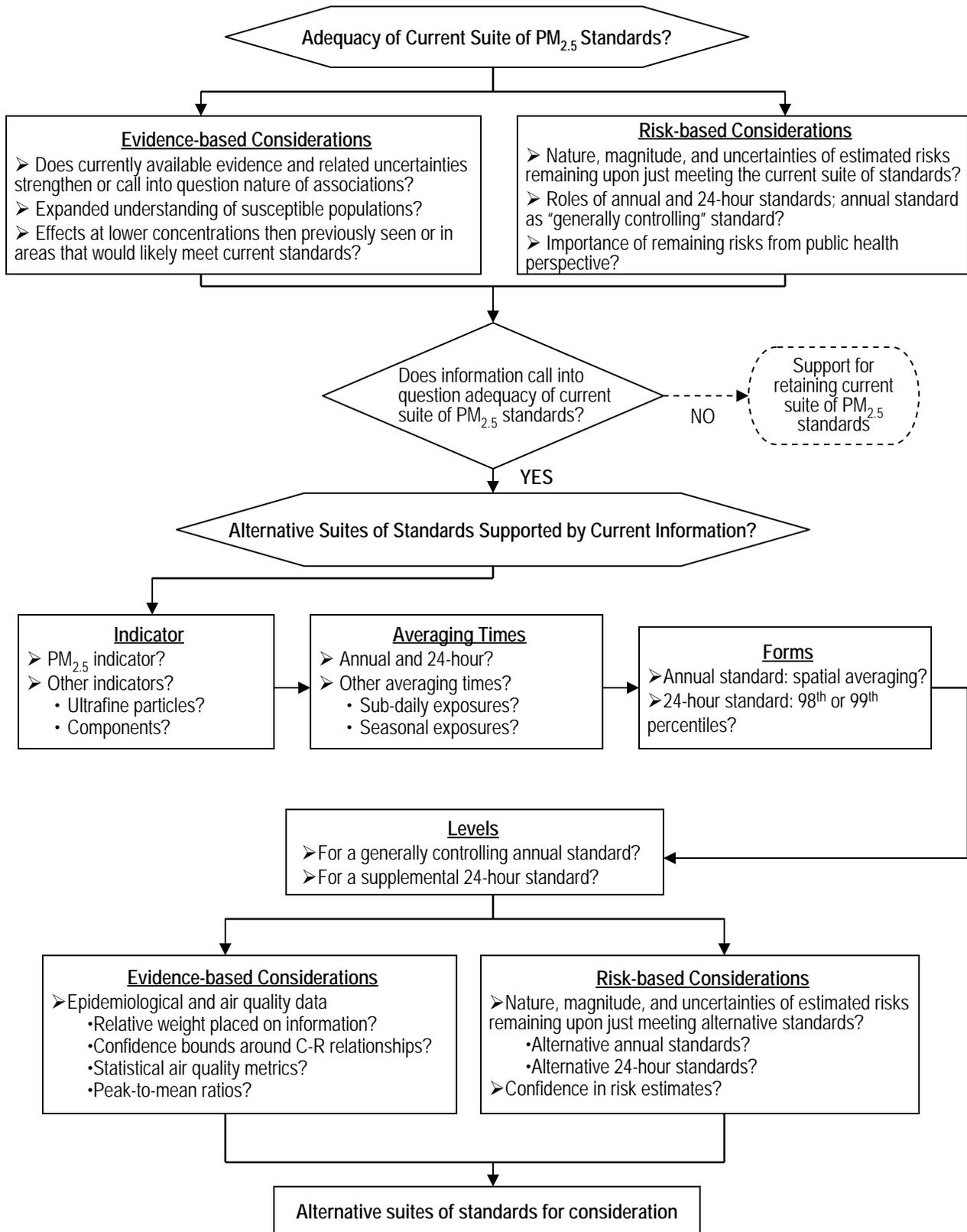
⁵ A range of one standard deviation around the mean represents approximately 68% of normally distributed data, and below the mean falls between the 25th and 10th percentiles.

1 Beyond these evidence-based considerations, we also consider the quantitative risk
2 estimates and the key observations presented in the RA. This assessment included an evaluation
3 of 15 urban case study areas and estimated risk associated with a number of health endpoints
4 associated with long-term and short-term PM_{2.5} exposures (US EPA, 2010a). As part of our risk-
5 based considerations, we have considered estimates of the magnitude of PM_{2.5}-related risks
6 associated with recent air quality levels and air quality simulated to just meet the current and
7 alternative suites of standards using alternative simulation approaches. We have also
8 characterized the risk reductions, relative to the risks remaining upon just meeting the current
9 standards, associated with just meeting alternative suites of standards. In so doing, we recognize
10 the uncertainties inherent in such risk estimates, and have taken such uncertainties into account
11 by considering the sensitivity of the “core” risk estimates to alternative assumptions and methods
12 likely to have substantial impact on the estimates. In addition, we have conducted additional
13 analyses to characterize the representativeness of the urban study areas within a broader national
14 context. We have considered this risk-based information to help inform our conclusions on the
15 adequacy of the current suite of standards, potential alternative suites of standards that are
16 appropriate for consideration in this review, and on the roles that the annual and 24-hour
17 standards may play in affording protection against effects related to both long- and short-term
18 PM_{2.5} exposures.

19 Staff conclusions reflect our understanding of both evidence-based and risk-based
20 considerations to inform two overarching questions related to: (1) the adequacy of the current
21 suite of PM_{2.5} standards and (2) potential alternative standards for consideration in this review to
22 provide appropriate protection from the effects associated with both long- and short-term
23 exposures to fine particles. In addressing these broad questions, we have organized the
24 discussions below around a series of more specific questions reflecting different aspects of each
25 overarching question. When evaluating the health protection afforded by the current or any
26 alternative suites of standards considered, we have taken into account the four basic elements of
27 the NAAQS (e.g., indicator, averaging time, form, and level). Figure 2-1 provides an overview
28 of the policy-relevant questions that frame our review, as discussed more fully below.

29 We believe that the approach outlined above, when presented in the final PA, will
30 provide a comprehensive basis to help inform the judgments required of the Administrator in
31 reaching decisions about the current and potential alternative primary PM_{2.5} standards and in
32 responding to the remand of the 2006 annual PM_{2.5} standard.

Figure 2-1. Overview of Approach for Review of Primary PM_{2.5} Standards



1 **2.2 ADEQUACY OF CURRENT STANDARDS**

2 In considering the adequacy of the current suite of PM_{2.5} standards, staff addresses the
3 following overarching question:

4 **Does the currently available scientific evidence and risk-based information, as reflected in**
5 **the ISA and RA, support or call into question the adequacy of the protection afforded by**
6 **the current suite of fine particle standards?**

7 To inform the answer to this broad question, we address a series of more specific
8 questions to aid in considering the currently available scientific evidence and the results of recent
9 quantitative risk analyses in a policy-relevant context, as discussed below. In considering the
10 scientific and technical information, we reflect upon both the information available in the last
11 review and information that is newly available since the last review as assessed and presented in
12 the ISA and the RA (US EPA, 2009a; US EPA, 2010a).

13 **2.2.1 Evidence-based Considerations**

14 Our review of the adequacy of the current suite of PM_{2.5} primary standards begins by
15 considering the strength of the evidence, susceptible populations, and the air quality distributions
16 over which health effects associations have been reported.

- 17 • **To what extent does the currently available scientific evidence and related uncertainties**
18 **strengthen or call into question evidence of associations between ambient fine particle**
19 **exposures and health effects?**

20 In considering the strength of the associations between long- and short-term exposures to
21 PM_{2.5} and health effects, we note that in the last review EPA concluded that there was “strong
22 epidemiological evidence” for PM_{2.5} linking long-term exposures with cardiovascular and lung
23 cancer mortality and respiratory morbidity and short-term exposures with cardiovascular and
24 respiratory mortality and morbidity (US EPA, 2004, p. 9-46; US EPA, 2005, p. 5-4). Overall,
25 the epidemiological evidence supported “likely causal associations” between PM_{2.5} and both
26 mortality and morbidity from cardiovascular and respiratory diseases, based on “an assessment
27 of strength, robustness, and consistency in results” (US EPA, 2004, p. 9-48).

28 In looking across the extensive new scientific evidence available in this review, our
29 overall understanding of health effects associated with fine particle exposures has been greatly
30 expanded. The currently available evidence is stronger in comparison to evidence available in
31 the last review because of its breadth and the substantiation of previously observed health effects
32 (US EPA, 2009a, section 2.3.1). A number of large multi-city epidemiological studies have been
33 conducted throughout the U.S. including extended analyses of studies that were important to
34 inform decisionmaking in the last review. These studies have reported consistent increases in

1 morbidity and/or mortality related to ambient PM_{2.5} concentrations, with the strongest evidence
2 reported for cardiovascular-related effects. In addition, the findings of new toxicological and
3 controlled human exposure studies provide stronger support for a number of potential biologic
4 mechanisms or pathways for PM-related cardiovascular and respiratory effects (US EPA, 2009a,
5 chapter 5; Figures 5-4 and 5-5). In summary, the ISA concludes, “[t]he new evidence ... greatly
6 expands upon the evidence available in the 2004 PM AQCD particularly in providing greater
7 understanding of the underlying mechanisms for PM_{2.5} induced cardiovascular and respiratory
8 effects for both short- and long-term exposures” (US EPA, 2009a, p. 2-17).

9 With regard to causal inferences described in the ISA, we note that since the last review
10 EPA has developed a more formal framework for reaching causal determinations that draws
11 upon the assessment and integration of evidence from across epidemiological, controlled human
12 exposure, and toxicological studies, and the related uncertainties, that ultimately influence our
13 understanding of the evidence (US EPA, 2009a, section 1.5). This framework employs a five-
14 level hierarchy that classifies the overall weight of evidence and causality using the following
15 categorizations: causal relationship, likely to be a causal relationship, suggestive of a causal
16 relationship, inadequate to infer a causal relationship, and not likely to be a causal relationship
17 (US EPA, 2009a, Table 1-3).

18 Using this causal framework, the ISA concludes that the collective evidence is largely
19 consistent with past studies and substantially strengthens what was known in the last review to
20 reach the conclusion that a *causal* relationship exists between both long- and short-term
21 exposures to PM_{2.5} and mortality and cardiovascular effects including cardiovascular-related
22 mortality. The ISA also concludes that the collective evidence continues to support *likely causal*
23 associations between long- and short-term PM_{2.5} exposures and respiratory effects, including
24 respiratory-related mortality. Further, the ISA concludes that available evidence is *suggestive* of
25 a causal relationship between long-term PM_{2.5} exposures and other health effects, including
26 developmental and reproductive effects (e.g., low birth weight) and carcinogenic, mutagenic, and
27 genotoxic effects (e.g., lung cancer mortality). Table 2-1 summarizes these causal
28 determinations (US EPA, 2009a, sections 2.3.1 and 2.6).

1

Table 2-1. Summary of Causal Determinations for PM_{2.5}

Exposure Duration	Outcome	Causality Determination
Long-term	Mortality	Causal
	Cardiovascular Effects	Causal
	Respiratory Effects	Likely to be Causal
	Reproductive and Developmental Effects	Suggestive
	Cancer, Mutagenicity, Genotoxicity Effects	Suggestive
Short-term	Mortality	Causal
	Cardiovascular Effects	Causal
	Respiratory Effects	Likely to be Causal
	Central Nervous System Effects	Inadequate

2 Source: adapted from US EPA, 2009, Table 2-6.

3 Health Effects Associated with Long-term PM_{2.5} Exposure

4 With regard to *mortality*, the ISA concludes that newly available evidence significantly
5 strengthens the link between long-term exposure to PM_{2.5} and mortality, while providing
6 indications that the magnitude of the PM_{2.5}-mortality association may be larger than previously
7 estimated (US EPA, 2009a, sections 7.2.10, 7.2.11, 7.6.1; Figures 7-6 and 7-7). A number of
8 large U.S. cohort studies have been published since the last review, including extended analyses
9 of the American Cancer Society (ACS) and Harvard Six Cities studies (US EPA, 2009a, pp 7-84
10 to 7-85; Figure 7-6; Krewski et al., 2009; Pope et al., 2004; Jerrett et al., 2005; Laden et al.,
11 2006). In addition, new long-term PM_{2.5} exposure studies evaluating mortality impacts in
12 additional cohorts are now available (US EPA, 2009a, section 7.6). For example, the Women's
13 Health Initiative (WHI) reported effects of PM_{2.5} on cardiovascular-related mortality in post-
14 menopausal women with no previous history of cardiac disease (Miller et al., 2007), while
15 multiple studies observed PM_{2.5}-associated mortality among older adults using Medicare data
16 (Eftim et al., 2008; Zeger et al., 2007, 2008). Collectively, these new studies, along with
17 evidence available in the last review, provide us with consistent and stronger evidence of
18 associations between long-term exposure to PM_{2.5} and mortality (U.S. EPA, 2009a, sections 2.3.1
19 and 7.6).

20 The strength of the causal association between long-term PM_{2.5} exposure and mortality
21 also builds upon new studies providing evidence of improvement in community health following

1 reductions in ambient fine particles. Pope et al. (2009) have documented the population health
2 benefits of reducing ambient air pollution by correlating past reductions in ambient PM_{2.5}
3 concentrations with increased life expectancy. These investigators report that reductions in
4 ambient fine particles during the 1980s and 1990s account for as much as 15 percent of the
5 overall improvement in life expectancy in 51 U.S. metropolitan areas, with the fine particle
6 reductions reported to be associated with an estimated increase in mean life expectancy of
7 approximately 5 to 9 months (US EPA, 2009a, p. 7-95; Pope et al., 2009). An extended analysis
8 of the Harvard Six Cities study found that as cities cleaned up their air, locations with the largest
9 reductions in PM_{2.5} saw the largest improvements in reduced mortality rates, while those with the
10 smallest decreases in PM_{2.5} concentrations saw the smallest improvements (Laden et al., 2006).
11 Another extended follow-up to the Harvard Six Cities study investigated the delay between
12 changes in ambient PM_{2.5} concentrations and changes in mortality (Schwartz et al., 2008) and
13 reported that the effects of changes in PM_{2.5} were seen within the 2 years prior to death (US
14 EPA, 2009a, p. 7-92; Figure 7-9).

15 With regard to *cardiovascular effects*, several new studies have examined the association
16 between cardiovascular effects and long-term PM_{2.5} exposures in multi-city studies conducted in
17 the U.S. and Europe. The ISA concludes that the strongest evidence comes from recent studies
18 investigating *cardiovascular-related mortality*. This includes evidence from a number of large,
19 multi-city U.S. long-term cohort studies including extended follow-up analyses of the ACS and
20 Harvard Six Cities studies, as well as the WHI (US EPA, 2009a, sections 7.2.10 and 7.6.1;
21 Krewski et al., 2009; Pope et al., 2004; Laden et al., 2006; Miller et al., 2007). Pope et al. (2004)
22 reported a positive association between mortality and long-term PM_{2.5} exposure for a number of
23 specific cardiovascular diseases, including ischemic heart disease (IHD), dysrhythmia, heart
24 failure, and cardiac arrest (US EPA, 2009a, p. 7-84; Figure 7-7). Krewski et al. (2009) provides
25 further evidence for IHD-related mortality associated with long-term PM_{2.5} exposure (US EPA,
26 2009a, p. 7-84, Figure 7-7).

27 With regard to *cardiovascular-related morbidity* associated with long-term PM_{2.5}
28 exposures, studies were not available in the last review. Recent studies, however, have provided
29 new evidence linking long-term exposure to PM_{2.5} with cardiovascular outcomes that has
30 “expanded upon the continuum of effects ranging from the more subtle subclinical measures to
31 cardiopulmonary mortality” (US EPA, 2009a, p. 2-17). In the current review, studies are now
32 available that evaluated a number of endpoints ranging from subtle indicators of cardiovascular
33 health to serious clinical events associated with coronary heart disease (CHD) and
34 cerebrovascular disease (CVD), including myocardial infarction (MI), coronary artery
35 revascularization (e.g., bypass graft, angioplasty, stent, atherectomy), congestive heart failure
36 (CHF), and stroke. The most significant new evidence comes from the WHI study which

1 provides evidence of nonfatal cardiovascular events including both coronary and cerebrovascular
2 events (Miller et al., 2007; US EPA, 2009a, sections 7.2.9 and 7.6.1). Toxicological studies
3 provide supportive evidence that the cardiovascular morbidity effects observed in long-term
4 exposure epidemiological studies are biologically plausible and coherent with studies of
5 cardiovascular-related mortality as well as with studies of cardiovascular-related effects
6 associated with short-term exposures to PM_{2.5}, as described below (US EPA, 2009a, p 7-19).

7 With regard to *respiratory effects*, the ISA concludes that extended analyses of studies
8 available in the last review as well as new epidemiological studies conducted in the U.S. and
9 abroad provide stronger evidence of *respiratory-related morbidity* associated with long-term
10 PM_{2.5} exposure. The strongest evidence for respiratory-related effects available in this review is
11 from studies that evaluated decrements in lung function growth, increased respiratory symptoms,
12 and asthma development (U.S. EPA, 2009a, sections 2.3.1.2, 7.3.1.1, and 7.3.2.1).⁶ Specifically,
13 extended analyses of the Southern California Children’s Health Study (CHS) provided evidence
14 that clinically important deficits in lung function⁷ associated with long-term exposure to PM_{2.5}
15 persisted into early adulthood (U.S., EPA, 2009a, p. 7-27; Gauderman et al., 2004). Additional
16 analyses of the CHS cohort reported an association between long-term PM_{2.5} exposure and
17 bronchitic symptoms (US EPA, 2009a, p. 7-24; McConnell et al., 2003) and a strong modifying
18 effect on the association between lung function and asthma incidence (US EPA, 2009, 7-25;
19 Islam et al., 2007). The outcomes observed in these more recent reports from the Southern
20 California CHS, including evaluation of a broader range of endpoints and longer follow-up
21 periods, were larger in magnitude and more precise. Supporting these results are new
22 longitudinal cohort studies conducted by other researchers in varying locations using different
23 methods (U.S. EPA, 2009a, section 7.3.9.1). New evidence from a U.S. cohort of cystic fibrosis
24 (CF) patients provided evidence of association between long-term PM_{2.5} exposures and
25 exacerbations of respiratory symptoms resulting in hospital admissions or use of home
26 intravenous antibiotics (US EPA, 2009a, p. 7-25; Goss et al., 2004).

27 Toxicological studies provide coherence and biological plausibility for the respiratory
28 effects observed in epidemiological studies (US EPA, 2009a, p. 7-42). For example, pre- and
29 postnatal exposure to ambient levels of urban particles has been found to affect lung
30 development in an animal model (US EPA, 2009a, section 7.3.2.2; p. 7-43). This finding is

⁶ Supporting evidence comes from studies “that observed associations between long-term exposure to PM₁₀ and an increase in respiratory symptoms and reductions in lung function grown in areas where PM₁₀ is dominated by PM_{2.5}” (US EPA, 2009a, p. 2-12).

⁷ Clinical significance was defined as a FEV₁ below 80% of the predicted value, a criterion commonly used in clinical settings to identify persons at increased risk for adverse respiratory conditions (US EPA, 2009a, p. 7-29-7-30). The primary NAAQS for SO₂ also includes this interpretation for FEV₁ (75 FR 35525, June 22, 2010).

1 important because impaired lung development is one mechanism by which PM exposure may
2 decrease lung function growth in children (US EPA, 2009a, p. 2-12; section 7.3).

3 With regard to *respiratory-related mortality* associated with long-term PM_{2.5} exposure,
4 the ISA concludes that the evidence is “limited and inconclusive” (US EPA, 2009a, p. 7-41).
5 The extended follow-up of the Harvard Six Cities study reported a positive but non-statistically
6 significant association between long-term PM_{2.5} exposure and respiratory-related mortality
7 (Laden et al., 2006), whereas Pope et al. (2004) found no association (US EPA, 2009a, p. 7-84).
8 There is emerging but limited evidence for an association between long-term PM_{2.5} exposure and
9 respiratory mortality in post-neonatal infants where long-term exposure was considered as
10 approximately one month to one year (US EPA, 2009a, pp. 7-54 to 7-55). Emerging evidence of
11 short- and long-term exposure to PM_{2.5} and respiratory morbidity and infant mortality provide
12 some support for the weak respiratory mortality effects observed.

13 Beyond effects considered to have causal or likely causal associations with long-term
14 PM_{2.5} exposure discussed above, the following health outcomes are classified as having evidence
15 suggestive of a causal association with long-term PM_{2.5} exposure: (1) reproductive and
16 developmental effects and (2) cancer, mutagenicity, and genotoxicity effects (US EPA, 2009a,
17 Table 2-6). With regard to *reproductive and developmental effects*, the ISA notes, “[e]vidence is
18 accumulating for PM_{2.5} effects on low birth weight and infant mortality, especially due to
19 respiratory causes during the post-neonatal period” (US EPA, 2009a, section 2.3.1.2). New
20 evidence available in this review reports a significant association between exposure to PM_{2.5}
21 during pregnancy and lower birth weight, pre-term birth, and intrauterine growth restriction, and
22 a significant association between post-natal exposure to PM_{2.5} and an increased risk of infant
23 mortality (US EPA, 2009a, section 7.4). The ISA further notes that “[i]nfants and fetal
24 development processes may be particularly vulnerable to PM exposure, and although the
25 physical mechanisms are not fully understood, several hypotheses have been proposed involving
26 direct effects on fetal health, altered placenta function, or indirect effects on the mother’s health”
27 (US EPA, 2009a, section 7.4.1). While toxicological studies provide some evidence that
28 supports an association between long-term PM_{2.5} exposure and adverse reproductive and
29 developmental outcomes, there is “little mechanistic information or biological plausibility for an
30 association between long-term PM exposure and adverse birth outcomes (e.g., low birth weight
31 or infant mortality)” (US EPA, 2009a, p. 2-13).

32 With regard to *cancer, mutagenic and genotoxic effects*, “[m]ultiple epidemiologic
33 studies have shown a consistent positive association between PM_{2.5} and lung cancer mortality,
34 but studies have generally not reported associations between PM_{2.5} and lung cancer incidence”
35 (US EPA, 2009a, p. 2-13 and sections 2.3.1.2 and 7.5; Table 7-7; Figures 7-6 and 7-7). The
36 extended follow-up to the ACS study reported an association between long-term PM_{2.5} exposure

1 and lung cancer mortality (US EPA, 2009a, p. 7-71; Krewski et al., 2009) as did the extended
2 follow-up to the Harvard Six Cities study when considering the entire 25-year follow-up period.
3 There is some evidence, primarily from *in vitro* studies, providing biological plausibility for the
4 PM-lung cancer relationships observed in epidemiological studies (US EPA, 2009a, p. 7-80),
5 although toxicological studies of carcinogenicity, mutagenicity, and genotoxicity generally
6 reported mixed results (US EPA, 2009a, section 7.5).

7 Health Effects Associated with Short-term PM_{2.5} Exposure

8 In considering effects associated with short-term PM_{2.5} exposure, the body of currently
9 available scientific evidence has been expanded greatly by the publication of a number of new
10 multi-city time-series studies that have used uniform methodologies to investigate the effects of
11 short-term fine particle exposures on public health. This body of evidence provides a more
12 expansive data base and considers multiple locations representing varying regions and seasons
13 that provide evidence on the influence of climate and air pollution mixes on PM_{2.5}-associated
14 health effects. These studies provide more precise estimates of the magnitude of effects
15 associated with short-term PM_{2.5} exposure than most smaller-scale single-city studies that were
16 more commonly available in the last review (U.S. EPA 2009a, chapter 6).

17 With regard to *mortality*, extended and expanded analyses of a multi-city study available
18 in the last review as well as new U.S. multi-city and single-city short-term PM_{2.5} exposure
19 studies have found generally consistent positive associations between short-term PM_{2.5} exposures
20 and cardiovascular- and respiratory-related mortality as well as all-cause (non-accidental)
21 mortality (US EPA, 2009a, sections 2.3.1.1, 6.2.11 and 6.5.2.2; Figures 6-26, 6-27, and 6-28). A
22 Canadian multi-city study available in the last review was expanded from 8 to 12 cities (Burnett
23 and Goldberg, 2003; Burnett et al., 2004), with consistent findings of a positive and statistically
24 significant association between short-term PM_{2.5} exposure and mortality (US EPA, 2009a, p 6-
25 182, Figure 2-1), although the influence of NO₂ and limited PM_{2.5} data for several years during
26 the study period somewhat diminish these findings. In an analysis of National Morbidity,
27 Mortality, and Air Pollution Study (NMMAPS) data, Dominici et al. (2007a) reported
28 associations between fine particle exposures and all-cause and cardio-respiratory mortality (US
29 EPA, 2009a, p. 6-175, Figure 6-26). In a study of 112 U.S. cities, Zanobetti and Schwartz
30 (2009) reported positive associations (in 99% of the cities) and frequently statistically significant
31 associations (in 55% of the cities) between short-term PM_{2.5} exposure and total (non-accidental)
32 mortality (US EPA, 2009a, pp 6-176 to 6-179; Figures 6-23 and 6-24). Collectively, these
33 studies provide generally consistent and much stronger evidence for PM_{2.5}-associated mortality
34 than the evidence available in the last review.

1 With regard to *cardiovascular effects*, new multi-city as well as single-city short-term
2 PM_{2.5} exposure studies conducted since the last review supports a largely positive and frequently
3 statistically significant relationship between short-term exposure to PM_{2.5} and cardiovascular-
4 related disease and mortality, substantiating prior findings. For example, among a multi-city
5 cohort of older adults participating in the Medicare Air Pollution Study (MCAPS) investigators
6 reported evidence of a positive association between short-term PM_{2.5} exposures and hospital
7 admissions related to cardiovascular outcomes (US EPA, 2009a, pp. 6-57 to 58; Dominici et al,
8 2006a; Bell et al, 2008). The strongest evidence for cardiovascular morbidity effects has been
9 observed predominately for ischemic heart disease (IHD) and congestive heart failure (CHF)
10 hospital admissions and emergency department visits, and cardiovascular-related mortality (US
11 EPA, 2009a, Figure 2-1, p. 6-79, sections 6.2.10.3, 6.2.10.5, and 6.2.11; Bell et al., 2008;
12 Dominici et al., 2006a; Tolbert et al., 2007; Zanobetti and Schwartz, 2009). Furthermore, these
13 findings are supported by a recent study of a multi-city cohort of women participating in the
14 WHI that reported a positive but non-statistically significant association between short-term
15 exposure to PM_{2.5} and ECG measures of myocardial ischemia (Zhang et al., 2009).

16 In focusing on *respiratory effects*, the strongest evidence from short-term PM_{2.5} exposure
17 studies has been observed for respiratory-related emergency department visits and hospital
18 admissions for chronic obstructive pulmonary disease (COPD) and respiratory infections (U.S.
19 EPA, 2009a, sections 2.3.1.1 and 6.3.8.3; Figures 2-1 and 6-13; Dominici et al., 2006a;).
20 Evidence for PM_{2.5}-related respiratory effects has also been observed in panel studies, which
21 indicate associations with respiratory symptoms, pulmonary function, and pulmonary
22 inflammation among asthmatic children. Although not consistently observed, some controlled
23 human exposure studies have reported small decrements in various measures of pulmonary
24 function following controlled exposures to PM_{2.5} (US EPA, 2009a, p. 2-10). Furthermore, the
25 comparatively larger body of toxicological evidence since the last review is coherent with the
26 evidence from epidemiological and controlled human exposure studies that examined short-term
27 exposures to PM_{2.5} and respiratory effects (US EPA 2009a, section 6.3.10.1).

28 Uncertainties in the Evidence

29 With respect to understanding the nature and magnitude of PM_{2.5}-related risks, as
30 discussed above, we recognize that epidemiological studies evaluating health effects associated
31 with long- and short-term PM_{2.5} exposures have reported heterogeneity in responses both within
32 and between cities and geographic regions within the U.S. This heterogeneity may be attributed,
33 in part, to differences in the fine particle composition. However, the currently available evidence
34 and limited availability of city-specific PM_{2.5} speciation data does not allow conclusions to be
35 drawn regarding the relative toxicity of PM_{2.5} components, combinations of PM_{2.5} components,

1 or sources of fine particles in different locations. Overall, the ISA concludes “that many
2 constituents of PM_{2.5} can be linked with multiple health effects, and the evidence is not yet
3 sufficient to allow differentiation of those constituents or sources that are more closely related to
4 specific health outcomes” (US EPA, 2009a, p. 2-17.)

5 Measurement error is also an important source of uncertainty. Variability in the
6 associations observed across PM_{2.5} epidemiological studies may be due in part to exposure error
7 related to the use of county-level air quality data and our limited understanding of factors that
8 may influence exposures (e.g., topography, the built environment, climate, source characteristics,
9 ventilation usage, personal activity patterns). As noted in the ISA, biases and uncertainties in
10 exposure estimates can result in biases and uncertainties in associated health effect estimates (US
11 EPA, 2009a, p. 2-17).

12 We note challenges with interpreting differences in health effects observed in the eastern
13 versus western parts of the U.S. As noted in section 2.3.2 of the ISA, western U.S. counties tend
14 to be larger and more topographically diverse than eastern U.S. counties and regional differences
15 in climate and infrastructure can effect time spent outdoors or indoors, housing characteristics
16 including air conditioning usage, and personal activity patterns. In light of these differences, the
17 ISA notes “...the available evidence and the limited amount of city-specific speciated PM_{2.5} data
18 does not allow conclusions to be drawn that specifically differentiate effects of PM in different
19 locations “ (US EPA 2009a, p. 2-17). Therefore, we recognize that important uncertainties
20 remain in this review related to understanding the temporal and spatial variability in PM_{2.5}
21 concentrations, including PM_{2.5} components, and associated health impacts across different
22 geographic areas.

23 Summary

24 In considering the extent to which newly available scientific evidence strengthens or calls
25 into question evidence of associations identified in the last review between ambient fine particle
26 exposures and health effects, we recognize that much progress has been made in assessing some
27 key uncertainties related to our understanding of health effects associated with short- and long-
28 term exposure to PM_{2.5}. As briefly discussed above as well as in the more complete discussion of
29 the evidence as assessed in the ISA, we note that the newly available information combined with
30 information available in the last review provides substantially stronger confidence in a causal
31 association between short- and long-term exposures to PM_{2.5} and mortality and cardiovascular
32 effects. In addition, the newly available evidence reinforces and expands the evidence
33 supporting the likely causal nature of the associations between short- and long-term exposure to
34 PM_{2.5} and respiratory effects. Causal inferences, as discussed in the ISA, are based not only on
35 the more expansive epidemiological evidence available in this review but also reflects

1 consideration of important progress that has been made to advance our understanding of a
2 number of potential biologic modes of action or pathways for PM-related cardiovascular and
3 respiratory effects (US EPA 2009a, chapter 5). With respect to suggestive evidence for a
4 broader range of effects, the body of scientific evidence is somewhat expanded but is still limited
5 with respect to associations between long-term PM_{2.5} exposures and developmental and
6 reproductive effects as well as cancer, mutagenic, and genotoxic effects. Thus, we conclude that
7 there is stronger and more consistent and coherent support for associations between short- and
8 long-term PM_{2.5} exposure and a broader range of health outcomes than was available in the last
9 review, providing the basis for fine particle standards at least as protective as the current PM_{2.5}
10 standards.

11 Having reached this initial conclusion, we then consider how the new evidence informs
12 our understanding of susceptible populations by addressing the following question:

- 13 • **To what extent does the currently available scientific evidence expand our**
14 **understanding of susceptible populations?**

15 Specific groups within the general population, referred to as susceptible populations, are
16 at increased risk for experiencing adverse health effects related to PM exposures.⁸ These groups
17 could exhibit a greater risk of PM-related health effects than the general population for a number
18 of reasons including, being affected by lower concentrations of PM, experiencing a larger health
19 impact at a given PM concentration, and/or being exposed to higher PM concentrations than the
20 general population. Given the heterogeneity of individual responses to PM exposures, the
21 severity of the health effects experienced by a susceptible population may be much greater than
22 that experienced by the population at large.

23 As summarized below, the currently available epidemiological and controlled human
24 exposure evidence expands our understanding of previously identified susceptible populations
25 (e.g., children, older adults, and individuals with pre-existing heart and lung disease) and
26 supports the identification of additional susceptible populations (e.g., persons with lower
27 socioeconomic status (SES), genetic differences) (US EPA, 2009a, section 2.4.1, Table 8-2). In
28 addition, toxicological studies provide underlying support for the biological mechanisms that
29 potentially lead to increased susceptibility to PM-related health effects. These studies also.

30 Lifestages: Children and Older Adults

31 Two different lifestages have been associated with increased susceptibility to PM-related
32 health effects: childhood (i.e., less than 18 years of age) and older adulthood (i.e., 65 years of
33 age and older). Childhood represents a lifestage where susceptibility to PM exposures may be

⁸ Although studies have primarily used exposures to PM₁₀ or PM_{2.5}, the available evidence suggests that the identified factors also increase risk from PM_{10-2.5} (US EPA, 2009a, section 8.1.8).

1 related to the following observations: children spend more time outdoors; children have greater
2 activity levels than adults; children have exposures resulting in higher doses per body weight and
3 lung surface area; and the developing lung is prone to damage, including irreversible effects,
4 from environmental pollutants as it continues to develop through adolescence (US EPA, 2009a,
5 section 8.1.1.2). Older adults represent a lifestage where susceptibility to PM-associated health
6 effects may be related to the higher prevalence of pre-existing cardiovascular and respiratory
7 diseases found in this age group compared to younger age groups as well as the gradual decline
8 in physiological processes that occur as part of the aging process (US EPA, 2009a, section
9 8.1.1.1).

10 With regard to mortality, recent epidemiological studies have found that older adults are
11 at greater risk of all-cause (non-accidental) mortality associated with short-term exposure to both
12 PM_{2.5} and PM₁₀, providing consistent and stronger evidence of effects in this susceptible
13 population compared to the last review. Epidemiological studies that examined the association
14 between mortality and long-term exposure to PM_{2.5} that stratified the results by age (i.e., less
15 than 65 years of age compared to aged 65 and older; different age groups within the aged 65 and
16 older population) reported results that are generally consistent with the findings of these short-
17 term exposure studies while also providing some evidence that risk declines with increasing age
18 starting at age 60, such that there is no evidence of an association among persons 85 years and
19 older (US EPA, 2009a, Figure 7-7, section 8.1.1.1, Zeger et al., 2008).

20 With regard to morbidity effects, currently available studies provide evidence that older
21 adults have heightened responses, especially for cardiovascular-related effects, and children have
22 heightened responses for respiratory-related effects associated with long- and short-term PM_{2.5}
23 exposures. With regard to older adults, epidemiological studies provide evidence of increases in
24 PM_{2.5}-related risk of MI, coronary revascularization,⁹ and their combination with CHD-related
25 death for participants free of CVD at baseline (Miller et al., 2007) as well as cardiovascular-
26 related hospitalization (Dominici et al., 2006; Bell et al., 2008). Further, dosimetry studies have
27 shown a depression of fine and coarse PM clearance in all regions of the respiratory tract with
28 increasing age beyond young adulthood, suggesting that older adults are at greater risk of PM-
29 related respiratory health effects (US EPA, 2009a, section 8.1.1).

30 With regard to respiratory-related effects in children associated with long-term PM
31 exposures, our understanding of effects on lung development has been strengthened based on
32 newly available evidence that is consistent and coherent across different study designs, locations,
33 and research groups. For example, the strongest evidence comes from the extended follow-up
34 for the Southern California CHS which includes several new studies that report positive

⁹ Coronary revascularization includes percutaneous coronary interventions, such as angioplasty.

1 associations between long-term exposure to PM_{2.5} and respiratory morbidity, particularly for
2 such endpoints as lung function growth, respiratory symptoms (i.e., bronchitic symptoms), and
3 respiratory disease incidence (US EPA, 2009a, section 7.3; Gauderman et al., 2004). These
4 analyses provide evidence that PM_{2.5}-related effects persist into early adulthood and are more
5 robust and larger in magnitude than previously reported.

6 With regard to respiratory effects in children associated with short-term exposures to PM,
7 currently available studies provide stronger evidence of respiratory-related hospitalizations with
8 larger effect estimates observed among children. In addition, reductions in lung function (FEV₁)
9 and an increase in respiratory symptoms and medication use associated with PM exposures has
10 been reported among asthmatic children (US EPA, 2009a, sections 6.3.1, 6.3.2.1, 8.4.9).

11 In addition, accumulating evidence suggests that the developing fetus may also represent
12 an additional lifestage that is susceptible to PM exposures. The ISA further notes that “[i]nfants
13 and fetal development processes may be particularly vulnerable to PM exposure, and although
14 the physical mechanisms are not fully understood, several hypotheses have been proposed
15 involving direct effects on fetal health, altered placenta function, or indirect effects on the
16 mother’s health” (US EPA, 2009a, section 7.4.1). Evidence is accumulating for PM_{2.5}-related
17 effects on low birthweight and infant mortality, especially due to respiratory causes during the
18 post-neonatal period (US EPA, 2009a, sections 2.3.1.2 and 7.4).

19 Pre-existing Diseases/Health Conditions

20 A number of health conditions have been found to put individuals at greater risk for
21 adverse effects following exposure to PM. The currently available evidence confirms and
22 strengthens evidence in the last review that individuals with underlying cardiovascular and
23 respiratory diseases are more susceptible to PM exposures.

24 The majority of the epidemiological studies that examined associations between short-
25 term PM exposures and cardiovascular outcomes focused on cardiovascular-related hospital
26 admissions and emergency department visits (US EPA, section 8.1.6.1). There is some new
27 evidence that individuals with pre-existing IHD are at greater risk of PM-associated hospital
28 admissions and emergency department visits related to cardiovascular effects. Additional studies
29 have focused on hypertension and on the effects of PM on cardiac function in individuals with
30 dysrhythmia with mixed results. One epidemiological study (US EPA, 2009a, section 7.2.9;
31 Zanobetti and Schwartz, 2007) investigated associations between long-term exposure to PM₁₀
32 and the progression of disease or reduced survival in a 21-city study of people discharged
33 following an acute MI, finding significant associations for mortality, CHF and new
34 hospitalization for MI.

1 With regard to individuals with pre-existing respiratory illnesses (e.g, asthma, COPD),
2 the ISA presents and assesses a number of studies that evaluate a broad range of health outcomes
3 (e.g., mortality, asthma symptoms) in response to PM exposures (US EPA, 2009a, section
4 8.1.6.2). Evidence in asthmatics is stronger and more consistent, while studies of persons with
5 COPD have reported mixed results. Epidemiological studies have examined the effect of short-
6 term exposure to PM in asthmatics finding an increase in medication use, asthma attacks, and
7 respiratory symptoms (i.e., asthma symptoms, cough, shortness of breath, and chest tightness).
8 Controlled human exposure studies report that healthy and asthmatic subjects exposed to coarse,
9 fine and ultrafine CAPs, exhibit similar respiratory responses, although these studies excluded
10 moderate and severe asthmatics that would be expected to show increased susceptibility to PM
11 exposure. Toxicological studies using diesel exhaust particles (DEPs) provide mechanistic
12 support that PM exposure results in allergic sensitization, and individuals with allergic airway
13 conditions are at greater risk of adverse effects upon exposure to PM_{2.5}. Further, there is
14 emerging but limited evidence which suggests that non-allergic respiratory morbidities may also
15 increase the susceptibility of an individual to PM-related respiratory effects (US EPA, 2009a, p.
16 8-12).

17 There is also emerging evidence that suggests the influence of additional pre-existing
18 diseases or health conditions, including diabetes and obesity on the manifestation of PM-related
19 health effects. The ISA notes that additional research exploring the effect of PM exposures on
20 obese individuals and identifying the biological pathway(s) that could increase the susceptibility
21 of diabetic and obese individuals to PM could improve our understanding of these potentially
22 susceptible populations (US EPA 2009a, pp. 2-23-2-24).

23 Socioeconomic Status

24 Stronger evidence is available in this review indicating that people from lower
25 socioeconomic strata are a susceptible population relative to PM exposures (US EPA, 2009a,
26 section 8.1.7). Persons with lower SES¹⁰ have been generally found to have a higher prevalence
27 of pre-existing diseases; limited access to medical treatment; and limited access to fresh foods
28 leading to a reduced intake of antioxidants, polyunsaturated fatty acids and vitamins, which can
29 increase this population's risk to PM-related effects. Evidence available in the last review from
30 the ACS and Harvard Six Cities cohort studies indicated increased mortality risk with long-term
31 exposure to PM_{2.5} in the cohort subgroups with lower education levels (US EPA 2004, section
32 9.2.4.5). In this review, additional support is available to identify persons with lower SES as a
33 susceptible population. For example, Krewski et al. (2009) found moderate evidence for

¹⁰ SES is a composite measure that usually consists of economic status, measured by income; social status measured by education; and work status measured by occupation (US EPA, 2009a, p. 8-14).

1 increased lung cancer mortality in individuals with a high school education or less in response to
2 long-term exposure to PM_{2.5}. However, IHD-related mortality associated with long-term PM_{2.5}
3 exposures was most strongly associated with individuals with higher education levels (US EPA,
4 2009a, p. 8-15).

5 Genetic Factors

6 Investigation of potential genetic susceptibility has provided evidence that individuals
7 with null alleles or polymorphisms in genes that mediate the antioxidant response to oxidative
8 stress (e.g., GSTM1), regulate enzyme activity (i.e., MTHFR and cSHMT), or regulate levels of
9 procoagulants (i.e., fibrinogen) are more susceptible to PM-related effects. However, some
10 evidence suggests that polymorphisms in genes (e.g., HFE) may provide protection for PM-
11 related effects. Emerging evidence also suggests that PM exposure can impart epigenetic effects
12 (i.e., DNA methylation) (US EPA, 2009a, p. 8-16). More research is needed to better understand
13 the relationship between genetic effects and potential susceptibility to PM-related effects.

14 Summary

15 In summary, we conclude that there are several susceptible populations that are likely to
16 be at increased risk of PM-related effects, including the lifestages of childhood and older
17 adulthood, those with preexisting heart and lung diseases, and those of lower SES. We also
18 recognize that there is emerging, though still limited evidence for additional potentially
19 susceptible populations, such as those with diabetes, people who are obese, or those with genetic
20 factors. We note that the available evidence does not generally allow distinctions to be drawn
21 between the PM indicators in terms of whether populations are more susceptible to a particular
22 size fraction (i.e., PM_{2.5} and PM_{10-2.5}).

- 23 • **To what extent does the currently available scientific evidence report associations that**
24 **extend to air quality concentrations that are lower than had previously been observed**
25 **or that are observed in areas that would likely meet the current suite of PM_{2.5}**
26 **standards?**

27 In focusing our attention on whether the available evidence supports consideration of
28 standards that are more protective than the current suite of PM_{2.5} standards, we first recognize
29 that the ISA concludes there is no evidence to support the existence of a discernible threshold
30 below which effects would not occur (US EPA, 2009a, section 2.4.3). Next, we consider
31 whether the evidence provides information for health effects associated with air quality
32 concentrations that are lower than had previously been observed, or if epidemiological studies
33 have reported effects in areas that would likely meet the current suite of PM_{2.5} standards.

1 Associations with Long-term PM_{2.5} Exposure

2 Extended follow-up analyses of the ACS and Harvard Six Cities studies provide
3 consistent and stronger evidence of a causal association with mortality at lower air quality
4 distributions than had previously been observed. The original and reanalysis of the ACS study
5 reported positive and statistically significant effects associated with a long-term mean PM_{2.5}
6 concentration of 18.2 µg/m³ across 50 metropolitan areas for 1979-1983 (Pope et al., 1995;
7 Krewski et al., 2000).¹¹ In extended analyses, positive and statistically significant effects of
8 approximately similar magnitude were associated with declining PM_{2.5} concentrations, from an
9 aggregate long-term mean in 58 metropolitan areas of 21.2 µg/m³ in the original monitoring
10 period (1979-1983) to 14.0 µg/m³ for 116 metropolitan areas in the most recent years evaluated
11 (1999-2000), with an overall average across the two study periods in 51 metropolitan areas of
12 17.7 µg/m³ (Pope et al., 2002; Krewski et al., 2009). With regard to the Harvard Six Cities
13 Study, the original and reanalysis reported positive and statistically significant effects associated
14 with a long-term mean PM_{2.5} concentration of 18.0 µg/m³ for 1980-1985 (Dockery et al., 1993;
15 Krewski et al., 2000). In an extended follow-up of this study, the aggregate long-term mean
16 concentration across all years evaluated was 16.4 µg/m³ for 1980-1988¹² (Laden et al., 2006).
17 In an additional analysis of the extended follow-up of the Harvard Six Cities cohort study,
18 investigators reported the C-R relationship was linear and “clearly continuing below the level” of
19 the current annual standard (US EPA, 2009a, p. 7-92; Schwartz et al., 2008).

20 We then consider new cohort studies that provide evidence of mortality associated with
21 air quality distributions that are generally lower than those reported in the ACS and Harvard Six
22 Cities studies, with effect estimates that were similar or greater in magnitude. The Women’s
23 Health Initiative (WHI) reported positive and most often statistically significant associations
24 between long-term PM_{2.5} exposure and cardiovascular-related mortality, with much larger
25 relative risk estimates than in the ACS and Harvard Six Cities studies, as well as morbidity
26 effects at an aggregate long-term mean PM_{2.5} concentration of 13.5 µg/m³ for 2000 (Miller et al.,
27 2007). Using the Medicare cohort, Eftim et al. (2008) reported somewhat higher effect estimates
28 than in the ACS and Harvard Six Cities studies with aggregate long-term mean concentrations of
29 13.6 µg/m³ and 14.1 µg/m³, respectively, for 2000-2002. The MCAPS reported associations
30 between long-term PM_{2.5} exposure and mortality for the eastern region of the U.S. at an
31 aggregated long-term PM_{2.5} median concentration of 14.0 µg/m³, although no association was
32 reported for the western region with an aggregated long-term PM_{2.5} median concentration of 13.1

¹¹ The study periods referred to in this document reflect the years of air quality data that were included in the analyses, whereas the study periods identified in the ISA reflect the years of health status data that were included.

¹² Aggregate mean concentration provided by study author (Laden, 2009).

1 $\mu\text{g}/\text{m}^3$ (US EPA, 2009a, p. 7-88; Zeger et al., 2008)¹³. Premature mortality in children reported
2 in a national infant mortality study as well as mortality in a cystic fibrosis cohort including both
3 children and adults reported positive but not statistically significant effects associated with long-
4 term aggregate mean concentrations of $14.8 \mu\text{g}/\text{m}^3$ and $13.7 \mu\text{g}/\text{m}^3$, respectively (Woodruff et al.,
5 2008; Goss et al., 2004).

6 With respect to respiratory morbidity effects associated with long-term $\text{PM}_{2.5}$ exposure,
7 the across-city mean of 2-week average $\text{PM}_{2.5}$ concentrations reported in the initial Southern
8 California CHS was approximately $15.1 \mu\text{g}/\text{m}^3$ (Peters et al., 1999). These results were found to
9 be consistent with results of cross-sectional analyses of the 24-City Study (Dockery et al., 1996;
10 Raizenne et al., 1996), which reported a long-term cross-city mean $\text{PM}_{2.5}$ concentration of 14.5
11 $\mu\text{g}/\text{m}^3$. In this review, extended analysis of the Southern California CHS provided stronger
12 evidence of $\text{PM}_{2.5}$ -related respiratory effects, at lower air quality concentrations than had
13 previously been reported, with a four-year aggregate mean concentration of $13.8 \mu\text{g}/\text{m}^3$ across
14 the 12 study communities (McConnell et al., 2003; Gauderman et al., 2004, US EPA, 2009a,
15 Figure 7-4).

16 Broadening our consideration to effects for which evidence is suggestive of a causal
17 association, we note birth outcome studies that reported positive and statistically significant
18 effects related to aggregate long-term mean $\text{PM}_{2.5}$ concentrations of approximately $12 \mu\text{g}/\text{m}^3$ (US
19 EPA, 2009a, Table 7-5; Bell et al., 2007; Liu et al., 2007; Parker et al., 2005). In contrast, Parker
20 and Woodruff (2008) reported no overall association with birthweight with an aggregate long-
21 term mean concentration of $13.5 \mu\text{g}/\text{m}^3$ (US EPA, 2009a, section 7.4.1.1).

22 Collectively, the currently available evidence provides support for associations between
23 long-term $\text{PM}_{2.5}$ exposure and mortality and morbidity effects that extend to air quality
24 concentrations that are lower than had previously been observed, with aggregated long-term
25 mean $\text{PM}_{2.5}$ concentrations extending to well below the level of the current standard. These
26 studies evaluated a broader range of health outcomes in the general population and in susceptible
27 populations than were considered in the last review, and include extended follow-up for
28 prospective epidemiological studies that were important in the last review as well as additional
29 evidence in important new cohorts.

¹³ Zeger et al. (2008) also reported positive and statistically significant effects for the central region, with an aggregate long-term mean $\text{PM}_{2.5}$ concentration of $10.7 \mu\text{g}/\text{m}^3$. However, in contrast to the eastern and western risk estimates, the central risk estimate increased with adjustment for COPD (used as a proxy for smoking status). Due to the potential for confounding bias influencing the risk estimate for the central region, we have not focused on the results reported in the central region to inform the adequacy of the current suite of standards.

1 Associations with Short-term PM_{2.5} Exposure

2 In considering long-term average ambient concentrations from short-term PM_{2.5} exposure
3 studies, an expanded follow-up analyses of a Canadian multi-city study provides evidence of a
4 causal association between short-term exposures to PM_{2.5} and mortality at lower air quality
5 distributions than had previously been observed. Specifically, the expansion of the multi-city
6 Canadian study from 8 (Burnett and Goldberg, 2003; aggregate long-term mean PM_{2.5}
7 concentration of 13.3 µg/m³) to 12 Cities (Burnett et al 2004) yielded results consistent with
8 prior findings of a positive and statistically significant association, but at a lower aggregate long-
9 term mean PM_{2.5} concentration of 12.8 µg/m³. In a multi-city time-series analysis of 112 U.S.
10 cities published since the last review, Zanobetti and Schwartz (2009) reported a positive and
11 statistically significant association with all-cause, cardiovascular-related (e.g., MI, stroke), and
12 respiratory-related mortality and short-term PM_{2.5} exposure, in which the aggregate long-term
13 mean PM_{2.5} concentration was 13.2 µg/m³ (ranging from 6.6 µg/m³ to 24.7 µg/m³) (US EPA,
14 2009a, Figure 6-24). Furthermore, city-specific effect estimates indicate the association between
15 short-term exposure to PM_{2.5} and total mortality and cardiovascular- and respiratory-related
16 mortality is consistently positive for an overwhelming majority (99%) of the 112 cities across a
17 wide range of air quality concentrations (US EPA, 2009a, Figure 6-24, p. 6-178 to 179). The
18 authors report that for all-cause mortality, city-specific effect estimates were statistically
19 significant for 55% of the 112 cities, with long-term city-mean PM_{2.5} concentrations ranging
20 from 7.8 µg/m³ to 18.7 µg/m³ and 24-hour PM_{2.5} city-mean 98th percentile concentrations
21 ranging from 18.4 to 64.9 µg/m³.

22 With regard to cardiovascular and respiratory morbidity effects, in the first analysis of the
23 MCAPS cohort conducted by Dominici et al. (2006a) across 204 US counties, investigators
24 reported a statistically significant association with hospitalizations for cardiovascular and
25 respiratory diseases and short-term PM_{2.5} exposure, in which the aggregate long-term mean
26 PM_{2.5} concentration was 13.4 µg/m³ (ranging from 4 µg/m³ to 23 µg/m³). Furthermore, a sub-
27 analysis restricted to days with 24-hour average concentrations of PM_{2.5} at or below 35 µg/m³
28 indicated that, in spite of a reduced statistical power from a smaller number of study days,
29 statistically significant associations were still observed between short-term exposure to PM_{2.5}
30 and hospital admissions for cardiovascular and respiratory diseases (Dominici, 2006b¹⁴). These
31 results, along with the observation that approximately 50% of the 204 county-specific mean 98th
32 percentile PM_{2.5} concentrations aggregated across all years were below the 24-hour standard of

¹⁴ This sub-analysis was not included in the original publication (Dominici et al., 2006a). Authors provided sub-analysis results for the Administrator's consideration as a letter to the docket following publication of the proposed rule in January 2006.

1 35 $\mu\text{g}/\text{m}^3$, suggests that the overall health effects observed across the U.S. are not primarily
2 driven by the higher end of the $\text{PM}_{2.5}$ air quality distribution (Bell, 2009, personal
3 communication from Dr. Michelle Bell regarding air quality data for Bell et al., 2008 and
4 Dominici et al., 2006). In an extended analysis of the MCAPS study, Bell et al (2008) reported a
5 positive and statistically significant increase in cardiovascular hospitalizations associated with
6 short-term $\text{PM}_{2.5}$ exposure, in which the aggregate long-term mean $\text{PM}_{2.5}$ concentration was 12.9
7 $\mu\text{g}/\text{m}^3$ (ranging from 4 $\mu\text{g}/\text{m}^3$ to 20 $\mu\text{g}/\text{m}^3$).

8 In considering single-city short-term $\text{PM}_{2.5}$ exposure studies that would likely have met
9 the current suite of standards, the following studies reported positive and statistically significant
10 associations in areas: Mar et al. (2004) reported an association for short-term $\text{PM}_{2.5}$ exposures
11 in relation to respiratory symptoms among children in Phoenix; and two studies noted in the last
12 review, Peters et al. (2001) and Delfino et al. (2007), reported an association with short-term
13 exposure to $\text{PM}_{2.5}$ and MI- and respiratory-related hospital admissions in Boston and Montreal,
14 Canada, respectively. Single-city studies that reported positive but statistically non-significant
15 associations for cardiovascular and respiratory endpoints include a number of studies reporting a
16 wide range of air quality distributions conducted in Saint John (Steib et al., 2000), Phoenix
17 (Wilson et al., 2007), Denver (Rabinovitch et al., 2006), Edmonton (Villeneuve et al., 2006), and
18 Nueces County, TX (Lisabeth et al., 2008). Other single-city short-term $\text{PM}_{2.5}$ exposure
19 analyses reported null findings for cardiovascular and respiratory morbidity effects in association
20 with short-term exposure to $\text{PM}_{2.5}$ in areas that would likely have met the current suite of
21 standards, including Phoenix (i.e., respiratory symptoms in adults; Mar et al., 2004); Spokane
22 (Slaughter et al. 2005), Denver (Rabinovitch et al., 2004) and Edmonton (Villeneuve et al.,
23 2006). In light of the mixed findings reported in single-city studies, particularly for studies
24 conducted in areas such as Phoenix, Denver, and Edmonton that report both positive and null
25 findings, we place comparatively greater weight on the results from multi-city studies in
26 considering the adequacy of the current suite of standards.

27 Collectively, the findings from multi-city short-term $\text{PM}_{2.5}$ exposure studies together with
28 support from single city studies at concentrations that would likely have met the current suite of
29 standards provide evidence of $\text{PM}_{2.5}$ associated health effects. These findings are further
30 bolstered by evidence of statistically significant $\text{PM}_{2.5}$ associated health effects occurring in
31 analyses restricted to days in which 24-hour average $\text{PM}_{2.5}$ concentrations were below 35 $\mu\text{g}/\text{m}^3$
32 (Dominici, 2006b).

33 Summary

34 In evaluating the currently available scientific evidence, we conclude that the evidence
35 from long and short-term $\text{PM}_{2.5}$ exposure studies calls into question whether the current suite of

1 PM_{2.5} primary standards protects public health with an adequate margin of safety from effects
2 associated with long- and short-term exposures to PM_{2.5}. We also conclude that this evidence
3 provides strong support for considering fine particle standards that would impart increased
4 protection beyond that afforded by the current annual and 24-hour PM_{2.5} standards. More
5 protective standards would reflect the substantially stronger and broader body of evidence for
6 mortality and cardiovascular and respiratory morbidity effects now available in this review both
7 at lower concentrations of air quality than had previously been observed and at levels allowed by
8 the current suite of PM_{2.5} standards.

9 **2.2.2 Risk-based Considerations**

10 Looking beyond evidence-based considerations, staff also has considered the extent to
11 which health risks estimated to occur upon just meeting the current suite of PM_{2.5} standards may
12 be judged to be important from a public health perspective. For this review, we have estimated
13 risk for a set of health effect endpoints based on a number of selection criteria (US EPA, 2010a,
14 section 3.3.1). Specifically, we have estimated risks for (a) all-cause, IHD, cardiopulmonary and
15 lung cancer mortality related to long-term PM_{2.5} exposure, (b) non-accidental, cardiovascular
16 (CV) and respiratory mortality related to short-term PM_{2.5} exposure, and (c) cardiovascular-
17 related and respiratory-related hospital admissions and asthma-related emergency department
18 visits associated with short-term PM_{2.5} exposure. In the discussion below, we focus on
19 cardiovascular-related endpoints, since the causal association for these endpoints based on
20 available literature is assessed in the ISA to be the strongest of the endpoints considered. The
21 modeled risks for the broader set of health effect endpoints are included in the RA (US EPA,
22 2010a).

23 As discussed below, three factors figure prominently in the interpretation of the risk
24 estimates associated with simulating just meeting the current suite of standards, including: (1) the
25 importance of changes in annual mean PM_{2.5} concentrations in a study area in estimating changes
26 in risks for both long- and short-term exposures, (2) the peakiness of ambient PM_{2.5}
27 concentrations in a study area,¹⁵ and (3) the spatial pattern of ambient PM_{2.5} reductions that result
28 from using different approaches to simulate just meeting the current standard levels (i.e., rollback
29 approaches). The latter two factors are interrelated and influence the degree of risk reduction
30 estimated under the current suite of standards.

31 The magnitude of both long- and short-term exposure-related risk estimated to remain
32 upon just meeting the current suite of standards is strongly associated with the simulated change

¹⁵ The term “peakiness” refers to air quality distributions across urban areas that have high peak-to-mean ratios (i.e. the ratio of the 24-hour design value to the annual mean design value) relative to distributions in other urban study areas in the U.S.

1 in annual mean PM_{2.5} concentrations. The role of annual mean PM_{2.5} concentrations in driving
2 long-term exposure-related risk estimates is intuitive given that risks are modeled using the
3 annual mean air quality metric.¹⁶ The fact that short-term exposure-related risk estimates are
4 also driven by changes in long-term mean PM_{2.5} concentrations is less intuitive, since changes in
5 mean 24-hour PM_{2.5} concentrations are used to estimate changes in risk for this time period.¹⁷
6 Analyses show that short-term exposure-related risks are not primarily driven by the small
7 number of days with PM_{2.5} concentrations in the upper tail of the air quality distribution, but
8 rather by the large number of days with PM_{2.5} concentrations at and around the mean of the
9 distribution (US EPA, 2010a, section 3.1.2.2). Consequently, changes in annual mean PM_{2.5}
10 concentrations are related, to a large extent, to changes in short-term exposure-related risk.
11 Therefore, we focus on changes in annual mean PM_{2.5} concentrations to inform our
12 understanding of patterns of both long-term and short-term exposure-related risk estimates across
13 the set of urban study areas evaluated in the quantitative RA.

14 The peakiness of ambient PM_{2.5} concentrations within a study area informs the type of
15 rollback approach used to simulate just meeting the current suite of standards to determine the
16 magnitude of the reduction in annual mean PM_{2.5} concentrations for that study area and
17 consequently the degree of risk reduction.¹⁸ For example, study areas with relative peaky PM_{2.5}
18 distributions are likely to have greater estimated risk reductions for the current suite of standards
19 (depending on their mix of 24-hour and annual design values), and such locations can be
20 especially sensitive to the type of rollback approach used, with the proportional rollback
21 approach resulting in notably greater estimated risk reduction compared with the locally-focused
22 rollback approach. In contrast, study areas with less peaky PM_{2.5} concentrations typically
23 experience greater simulated risk reductions for the current annual-standard level compared with
24 24-hour standard levels (again depending on the mix of 24-hour and annual design values). In
25 addition, the type of rollback approach used will tend to have less of an impact on the magnitude
26 of risk reductions for study areas with less peaky PM_{2.5} concentrations. Rigorous consideration

¹⁶ As noted in section 3.2.1 of the RA (U.S.EPA, 2010a), estimates of long-term exposure-related mortality are actually based on an annual mean PM_{2.5} concentration across monitors in a study area (i.e., the composite monitor mean). Therefore, in considering changes in long-term exposure-related mortality, it is most appropriate to compare composite monitor estimates generated for a study area under each suite of standards. The annual mean at the highest reporting monitor (i.e., maximum monitor) for a study area (i.e., the annual design value) determines the percent reduction in PM_{2.5} concentrations required to meet a particular standard. Both types of air quality estimates are provided in Table 3-4 of the RA and both are referenced in this discussion of core risk estimates, as appropriate.

¹⁷ Estimates of short-term exposure-related mortality and morbidity are based on composite monitor 24-hour PM_{2.5} concentrations. However, similar to the case with long-term exposure-related mortality, it is the 98th percentile 24-hour concentration estimated at the maximum monitor (the 24-hour design value) that will determine the degree of reduction required to meet a given 24-hour standard.

¹⁸ The peakiness of ambient PM_{2.5} levels also has a direct bearing on whether the 24-hour or annual standard will be controlling for a particular study area, with more peaky distributions generally being associated with locations where the 24-hour standard is controlling.

1 of these two factors, allowed us to better understand the nature and pattern of estimated risk
2 reductions and risk remaining upon simulation of just meeting the current suite of standards
3 across the urban study areas (see U.S.EPA, 2010a, section 5.2.1).

4 We have considered a series of questions to inform our understanding of the adequacy of
5 the current suite of fine particle standards based on insights obtained from the quantitative RA.
6 We begin by considering the overall confidence associated with the RA and the degree to which
7 the set of urban study areas included in the RA is representative of urban areas across the U.S..
8 We then consider the nature and magnitude of risk estimated to remain based on simulating just
9 meeting the current suite of standards.

10 • **What is the level of confidence associated with risk estimates generated for simulating**
11 **just meeting the current suite of PM_{2.5} standards?**

12 A number of design elements were included in the quantitative RA to increase the overall
13 confidence in the risk estimates generated for the 15 urban study areas. These elements
14 included: (a) use of a deliberative process in specifying components of the risk model that
15 reflects consideration of the latest research on PM_{2.5} exposure and risk (US EPA, 2010a, section
16 5.1.1), (b) integration of key sources of variability into the design as well as the interpretation of
17 risk estimates (U.S.EPA, 2010a, section 5.1.2), (c) assessment of the degree to which the urban
18 study areas are representative of areas in the U.S. experiencing higher PM_{2.5}-related risk
19 (U.S.EPA, 2010a, section 5.1.3), and (d) identification and assessment of important sources of
20 uncertainty and the impact of these uncertainties on the core risk estimates (U.S.EPA, 2010a,
21 section 5.1.4).¹⁹ Two additional analyses examined potential bias and overall confidence in the
22 risk estimates. The first analysis explored potential bias in the core risk estimates by considering
23 a set of alternative reasonable risk estimates generated as part of a sensitivity analysis. The
24 second analysis compared the annual mean PM_{2.5} concentrations associated with simulating just
25 meeting the current suite of standards with the air quality distribution used in deriving the C-R
26 functions applied in modeling mortality risk.²⁰ Greater confidence is associated with risk
27 estimates based on annual mean PM_{2.5} concentrations that are within the region of the air quality
28 distribution used in deriving the C-R functions where the bulk of the data reside (i.e., within one
29 standard deviation (SD) around the mean). Each of the design elements listed above together
30 with the two additional analyses is discussed below.

31 Staff used a deliberative process to specify each of the key analytical elements
32 comprising the core risk model, including: selection of urban study areas; selection of health

¹⁹ The “core” risk estimates produced in this assessment refer to those generated using the combination of modeling elements and input datasets in which we had the highest confidence relative to other modeling choices considered (note, that alternative modeling elements have been included as part off the sensitivity analyses completed).

²⁰ This analysis also considered simulations of alternative standards as discussed in section 2.4.2.

1 endpoints, including specification of the C-R functions to use in modeling those endpoints; and
2 choice of rollback approach used to simulate just meeting the current suite of standards. This
3 deliberative process involved rigorous review of the currently available literature addressing both
4 PM_{2.5} exposure and risk combined with the application of a formal set of criteria to guide
5 development of each of the key analytical elements in the RA (US EPA, 2010a, section 5.1.1)²¹
6 The application of this deliberative process increases overall confidence in the risk estimates by
7 insuring that the estimates are based on the best available science and data characterizing PM_{2.5}
8 exposure and risk, and that they reflect consideration of input from external experts on PM
9 exposure and risk through CASAC and public reviews.

10 We considered key sources of variability that can impact the nature and magnitude of
11 risks associated with the current standard levels across the urban study areas. These sources of
12 uncertainty include those that contribute to differences in risk across urban study areas, but do
13 not directly affect the degree of risk reduction associated with the simulation of the current
14 standard levels (e.g., differences in baseline incidence rates, demographics and population
15 behavior). We also focused on factors that not only introduce variability into risk estimates
16 across study areas, but also play an important role in determining the magnitude of risk
17 reductions upon simulation of current standard levels (e.g., peakiness in ambient PM_{2.5}
18 concentrations within individual urban study areas and the nature of the rollback approach used
19 to simulate just meeting the current standards – see earlier discussion).

20 Single and multi-factor sensitivity analyses were combined with a qualitative analysis to
21 assess the impact of potential sources of uncertainty on the core risk estimates. The qualitative
22 uncertainty analysis supplemented the quantitative sensitivity analyses by allowing coverage for
23 sources of uncertainty which could not be readily included in the sensitivity analysis (US EPA,
24 2010a, section 3.5.3). The quantitative sensitivity analyses informed our understanding of
25 sources of uncertainty which may have a moderate to large impact on the core risk estimates.
26 With respect to the long-term exposure-related mortality risk estimates, the most important
27 sources of uncertainty identified in the quantitative sensitivity analyses included: selection of C-
28 R function;²² modeling risk down to policy-relevant background (PRB) versus lowest measured
29 level (LML); and the choice of rollback approach used. With regard to the qualitative analysis of
30 uncertainty the following sources were identified as potentially having a large impact on core
31 risk estimates for the long-term exposure-related mortality: characterization of inter-urban

²¹ In addition, as discussed in section 1.2.4, the RA design reflects consideration of CASAC and public comments on the Scope and Methods Plan and two draft assessment documents.

²² In the case of long-term exposure-related mortality, we considered both alternative C-R functions from the epidemiological study providing the C-R function used in the core analysis (i.e., alternative functions obtained from the Krewski et al. (2009) study involving the ACS dataset) as well as alternative C-R functions identified in other studies (i.e., C-R functions obtained from Krewski et al. (2000) based on the Harvard Six Cities dataset).

1 population exposures; impact of historical air quality; and potential variation in effect estimates
2 reflecting differences in PM_{2.5} composition. Together, the qualitative analysis of uncertainty and
3 quantitative sensitivity analyses provided us with a comprehensive understanding of which
4 sources of uncertainty could have a significant impact on the core risk estimates. This
5 information proved useful in interpreting core risk estimates and increased our overall
6 confidence in the analysis.²³

7 In addition to identifying sources of uncertainty with a moderate to large impact on the
8 core risk estimates, the single and multi-element sensitivity analyses also produced a set of
9 reasonable alternative risk estimates that allowed us to place the results of the core analysis in
10 context with regard to uncertainty and potential bias.²⁴ Most of the alternative model
11 specifications supported by available literature produced risk estimates that are higher (by up to a
12 factor of 2 to 3) than the core risk estimates. This was not unexpected. The C-R functions used
13 in the core analysis for estimating mortality risks associated with long-term PM_{2.5} exposures
14 were selected from the extended analysis of the ACS study (Krewski et al., 2009). The C-R
15 functions used in the sensitivity analysis were from the reanalysis and validation of the Harvard
16 Six Cities study (Krewski et al., 2000). In generalizing the results of the extended analyses of
17 the ACS and Harvard Six Cities studies across the broader national population, we recognize
18 differences in the underlying populations enrolled in these long-term cohort studies, specifically
19 related to socioeconomic status (SES), a factor in considering impacts on susceptible
20 populations. As noted in the last review, the ACS study population has a higher SES status (e.g.,
21 educational status) relative to the Harvard Six Cities study population (12% versus 28% of the
22 cohort had less than a high school education, respectively) (US EPA, 2004a, p. 8-118). The
23 Harvard Six Cities cohort may provide a more representative sample of the broader national
24 population than the ACS cohort.

25 As discussed above, lower SES groups have been identified as a susceptible population.
26 Therefore, use of effect estimates reported in the ACS study which does not provide
27 representative coverage for lower-SES groups, may result in risk estimates that are biased low.
28 In contrast, risk estimates developed in the sensitivity analysis based on the Harvard Six Cities

²³ Given increased emphasis placed in this analysis on long-term exposure-related mortality, the uncertainty analyses completed for this health endpoint category are more comprehensive than those conducted for short-term exposure-related mortality and morbidity, which to some extent reflects limitations in study data available for addressing uncertainty in the later category (U.S.EPA 2010a, section 3.5.4.2).

²⁴ The alternative set of reasonable risk estimates are based on alternative model specifications to those used in the core risk model. These alternative reasonable risk estimates were only generated for long-term exposure-related mortality and not for any of the short-term exposure-related mortality or morbidity endpoints. Consequently, consideration of overall confidence (and potential bias) in the core risk estimates based on consideration for these alternative risk estimates is limited to estimates of mortality associated with long-term PM_{2.5} exposures.

1 study data set, provide better coverage for lower SES populations and therefore result in higher
2 risk estimates (US EPA, 2010a, section 5.1.5).

3 While being mindful that the use of C-R functions from Krewski et al. (2009) introduces
4 potential for low bias in the core risk estimates, we also recognize many strengths of this study
5 and reasons for continued use in generating the core risk estimates, including: the large number
6 of metropolitan statistical areas (MSAs), inclusion of two time periods for the air quality data
7 which allowed us to consider different exposure windows; and analysis of a wide range of C-R
8 function models. Therefore, we concluded that C-R functions obtained from this study had the
9 greatest overall support and should be used in the core risk model. Consideration of the
10 alternative set of reasonable risk estimates provided several observations relevant to the
11 interpretation of the core risk estimates including: (a) the core estimates are unlikely to
12 underestimate risk and (b) the degree of potential bias in the core risk estimates could range up to
13 at least a factor of 2-3 higher.²⁵

14 In considering the overall confidence in the core risk estimates, we have compared the
15 PM_{2.5} concentrations simulated under the current suite of standard levels across the urban study
16 areas to the distribution of PM_{2.5} concentrations used in deriving the C-R functions used for
17 long-term exposure-related mortality (as presented in Krewski et al., 2009). Specifically, this
18 assessment compared the composite monitor annual mean PM_{2.5} concentrations used in modeling
19 long-term exposure-related mortality risk in the core analysis to the distribution of annual mean
20 PM_{2.5} concentrations from the 1999-2000 ACS exposure period.²⁶ Generally, when composite
21 monitor annual mean concentrations were within one SD of the mean of the ACS dataset (i.e., in
22 the range of 14 \pm 3 $\mu\text{g}/\text{m}^3$), we had relatively high confidence in the risk estimates, since they
23 were based on PM_{2.5} concentrations that roughly matched those used in deriving the C-R
24 functions. However, as composite monitor annual mean PM_{2.5} concentrations extend below this

²⁵ We note that these findings regarding potential bias in the core risk estimates were based on modeling PM_{2.5}-attributable IHD and all-cause mortality associated with long-term PM_{2.5} exposure for the current suite of standards. However, we would expect these observations regarding overall confidence in the core risk estimates to hold for other long-term exposure-related mortality endpoints modeled in the RA for both the alternative annual and 24-hour standard levels considered. Furthermore, given increased emphasis placed in this analysis on long-term exposure-related mortality, as noted earlier, the uncertainty analyses completed for this health endpoint category are more comprehensive than those conducted for short-term exposure-related mortality and morbidity effects, which to some extent reflects limitations in study data available for addressing uncertainty in the later category. Therefore, an alternative set of reasonable risk estimates was not generated to supplement core risk estimates generated for short-term PM_{2.5} exposure.

²⁶ As discussed in sections 3.3.3 and 4.0 of the RA (U.S.EPA, 2010a), each category of long-term exposure-related mortality was estimated using separate C-R functions derived from the 1979-1983 and 1999-2000 ACS monitoring periods. For purposes of comparing composite monitor annual mean PM_{2.5} concentrations to the ACS data sets used in deriving the C-R functions, we focused on the later monitoring period (1999-2000), since ambient PM_{2.5} concentrations from this period more closely matched those associated with the study areas in our consideration of recent air quality conditions. The 1999-2000 ACS monitoring period had a mean PM_{2.5} concentration of 14 $\mu\text{g}/\text{m}^3$, a SD of 3.0 $\mu\text{g}/\text{m}^3$ and an LML of 5.8 $\mu\text{g}/\text{m}^3$ (see Table 1 in Krewski et al., 2009).

1 range, our confidence in the risk estimates decreased, with our confidence being significantly
2 reduced when composite monitor annual mean concentrations approach the LML of the ACS
3 dataset (i.e., 5.8 $\mu\text{g}/\text{m}^3$).

4 • **How representative is the set of urban study areas for the broader set of urban areas in
5 the U.S. expected to experience elevated risk from ambient PM_{2.5} exposure?**

6 The goal in selecting urban study areas was to provide coverage for the range of larger
7 urban areas in the U.S. expected to experience relatively elevated risk due to ambient PM_{2.5}
8 exposure and other factors associated with PM_{2.5} risk (e.g., elevated baseline incidence rates for
9 relevant health endpoints, relatively larger susceptible populations). As part of considering our
10 overall confidence in the quantitative RA, we assessed the representativeness of the 15 urban
11 study areas in the broader national context.. Three separate analyses were used to explore
12 representativeness:

- 13 • A comparison of PM_{2.5}-risk-related attributes of the 15 urban study areas against
14 national distributions of these same attributes suggested that the urban study areas
15 likely reflect the distribution of risk for the nation, with the potential for better
16 characterization at the high end of that distribution (US EPA, 2010a, section 4.4.1).²⁷
- 17 • An analysis of where the 15 urban study areas fall along the distribution of U.S.
18 counties included in a national-scale mortality analysis suggested that we have
19 captured counties likely to experience elevated PM_{2.5}-related risk (US EPA, 2010a,
20 section 4.4.2).
- 21 • An evaluation of the mix of design values across the 15 urban study areas as
22 contrasted with design values for the broader set of urban study areas in the U.S. This
23 analysis suggested that (a) the 15 urban study areas reasonably capture the key
24 groupings of urban areas in the U.S. likely to experience elevated risk due to PM_{2.5}
25 exposures and (b) we have included study areas likely to experience relatively greater
26 degrees of PM_{2.5}-related risk (US EPA, 2010a, section 4.5.1).

27 Based on these analyses, we conclude that these study areas are generally representative
28 of urban areas in the U.S. likely to experience relatively elevated levels of risk related to ambient
29 PM_{2.5} exposure.

30 • **What is the nature and magnitude of the *long-term* and *short-term* exposure-related
31 risks remaining upon just meeting the current suite of PM_{2.5} standards?**

32 In considering PM_{2.5}-related risks likely to remain upon just meeting the current PM_{2.5}
33 annual and 24-hour standards in the 15 urban study areas included in the quantitative RA, we

²⁷ This representativeness analysis also showed that the urban study areas do not capture areas with the highest baseline mortality risks or the oldest populations (both of which can result in higher PM_{2.5}-related mortality estimates). However, some of the areas with the highest values for these attributes have relatively low PM_{2.5} concentrations (e.g., urban areas in Florida) and consequently failure to include these areas in the set of urban study areas is unlikely to exclude high PM_{2.5}-risk locations.

1 focus on the 13 areas that would not meet the current standards based on recent air quality (2005-
2 2007). These 13 areas have annual and/or 24-hour design values that are above the levels of the
3 current standards (see Tables 2-2 and 2-3).²⁸ Based on the core risk estimates for these areas,
4 using the proportional rollback approach, we make the following key observations regarding the
5 magnitude of risk remaining upon simulation of just meeting the current suite of standards:

- 6 • *Long-term exposure-related mortality risk remaining:* IHD-related mortality
7 attributable to long-term PM_{2.5} exposure was estimated to range from less than 100 to
8 approximately 2,000 cases per year. The variability in these estimates reflect, to a
9 great extent, differences in the size of study area populations. These estimates
10 represent from 4 to 17% of all IHD-related mortality in a given year for the urban
11 study areas, representing a measure of risk that takes into account differences in
12 population size and baseline mortality rates (see Table 2-2).
- 13 • *Short-term exposure-related mortality risk remaining:* CV-related mortality
14 associated with short-term PM_{2.5} exposure was estimated to range from less than 10 to
15 500 cases per year. These estimates represent approximately 1 to 2% of total CV-
16 related mortality in a given year for the urban study areas (see Table 2-3).
- 17 • *Short-term exposure-related morbidity risk remaining:* CV-related hospitalizations
18 were estimated to range from approximately 10 to 800 cases per year across the study
19 areas, which is approximately equivalent to less than 1% of total CV-related
20 hospitalizations (see Table 2-3).

21 Although long- and short-term exposure-related mortality have similar patterns (in terms
22 of the subset of urban study areas experiencing risk reductions for the current suite of standard
23 levels), the magnitude of risk remaining is substantially lower for short-term exposure-related
24 mortality. These findings were expected, since, as noted earlier, changes in annual mean PM_{2.5}
25 concentrations were expected to drive both long- and short-term exposure-related risk, resulting
26 in similar overall patterns in risk reduction for both exposure periods (in terms of the subset of
27 study areas experiencing risk reductions). We note, however, that the variability in the effect
28 estimates used to model short-term exposure-related health endpoints across urban study areas
29 introduced additional variation into the pattern of risk reduction across study areas.

- 30 • *Substantial variability exists in the magnitude of risk remaining across study areas:*
31 Estimated risks remaining upon just meeting the current suite of standards vary
32 substantially across study areas, even when considering risks normalized for
33 differences in population size and baseline incidence rates. This variability is a
34 consequence of the substantial differences in the annual mean PM_{2.5} concentrations
35 across study areas that result from simulating just meeting the current standards. This
36 is important because, as discussed above, annual mean concentrations are highly
37 correlated with both long- and short-term exposure-related risk. This variability in

²⁸ Of the 15 study areas, only Dallas and Phoenix have both annual and 24-hour design values below the levels of the current standards based on 2005-2007 air quality.

1 annual mean PM_{2.5} concentrations occurred especially in those study areas in which
2 the 24-hour standard is the “controlling” standard. In such areas, the variability in
3 estimated risks across study areas was largest when regional patterns of reductions in
4 PM_{2.5} concentrations were simulated, using the proportional rollback approach, as
5 was done in the core analyses. Less variability was observed when more localized
6 patterns of PM_{2.5} reductions were simulated using the locally-focused rollback
7 approach, as was done in a sensitivity analysis. When simulations were done using
8 the locally-focused rollback approach, estimated risks remaining upon just meeting
9 the current suite of standards were appreciably larger than those estimated in the core
10 analysis (US EPA, 2010a, section 4.3.1.1).

11 • *Simulation of just meeting the current suite of standards results in annual mean PM_{2.5}*
12 *concentrations well below the current standard for some study areas:* In simulating
13 just meeting the current suite of standards, the resulting composite monitor annual
14 mean PM_{2.5} concentrations ranged from about 15 µg/m³ (for those study areas in
15 which the annual standard was controlling) down to as low as about 8 µg/m³ (for
16 those study areas in which the 24-hour standard was controlling or the annual mean
17 concentration was well below 15 µg/m³ based on recent air quality). As discussed
18 above, as the composite monitor annual mean PM_{2.5} concentrations used in generating
19 risk estimates extend below 11.0 µg/m³ (one SD below the mean for the 1999-2000
20 ACS monitoring period, Krewski et al., 2009) we have increasingly less confidence in
21 the risk estimates, with confidence decreasing significantly as composite monitor
22 concentrations approach the LML for the ACS dataset (5.8 µg/m³). Typically, for the
23 15 urban study areas assessed, the locations where the 24-hour standard was the
24 controlling standard were simulated to have the lowest composite monitor annual
25 mean PM_{2.5} concentrations. We observe that all four of the urban study areas with
26 simulated composite monitor annual mean PM_{2.5} concentrations below 11 µg/m³ have
27 the 24-hour standard controlling (U.S.EPA, 2010a, Table 3-4). While such locations
28 often are estimated to have the greatest risk reductions, there is also reduced
29 confidence associated with these risk estimates.

30 • **To what extent are the risks remaining upon simulation of the current suite of**
31 **standards important from a public health perspective?**

32 Estimates of long-term exposure-related IHD mortality risk under simulation of the
33 current suite of standard levels range from less than 100 deaths per year for the urban study area
34 with the lowest risk to approximately 2,000 deaths per year for the urban study areas with the
35 greatest risk. Estimates of risk for the urban areas included in the RA suggest that IHD mortality
36 related to long-term PM_{2.5} exposure would likely be in the thousands of deaths per year on a
37 national scale. Based on these risk estimates for IHD mortality alone, we conclude that risks
38 estimated to remain upon simulation of the current suite of standards are important from a public
39 health standpoint. This reflects consideration of both the severity of the effect and the magnitude
40 of the effect. We have also estimated risks for long-term exposure related mortality risk related
41 to cardiopulmonary effects and lung cancer, which increase the total annual incidence of

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Table 2-2. Estimated Incidence and Percent of Total Annual Incidence Associated with Long-term PM_{2.5} Exposure Based on Simulation of the Current Suite of Standards (for IHD mortality based on 2007 PM_{2.5} Concentrations)^{1,2}

Risk Assessment Location	Incidence of Ischemic Heart Disease Mortality Associated with Long-term Exposure to PM _{2.5} ³		Percent of Incidence of Ischemic Heart Disease Mortality Associated with Long-term Exposure to PM _{2.5} ³	
	Exposure Period: 1979-1983	Exposure Period: 1999-2000	Exposure Period: 1979-1983	Exposure Period: 1999-2000
Atlanta, GA	220 (180 - 258)	277 (227 - 324)	13.2% (10.9% - 15.5%)	16.7% (13.7% - 19.5%)
Baltimore, MD	297 (243 - 349)	374 (307 - 440)	11.7% (9.6% - 13.7%)	14.7% (12.1% - 17.3%)
Birmingham, AL	131 (107 - 154)	165 (135 - 194)	10.9% (8.9% - 12.9%)	13.8% (11.3% - 16.2%)
Dallas, TX	195 (159 - 230)	247 (202 - 291)	9% (7.3% - 10.6%)	11.4% (9.3% - 13.4%)
Detroit, MI	377 (308 - 445)	478 (390 - 563)	9.1% (7.4% - 10.7%)	11.5% (9.4% - 13.5%)
Fresno, CA	77 (63 - 92)	98 (80 - 116)	6.7% (5.5% - 8%)	8.5% (7% - 10.1%)
Houston, TX	344 (281 - 405)	434 (355 - 511)	10.7% (8.8% - 12.6%)	13.6% (11.1% - 16%)
Los Angeles, CA	860 (701 - 1018)	1094 (890 - 1296)	6.1% (4.9% - 7.2%)	7.7% (6.3% - 9.1%)
New York, NY	1755 (1435 - 2070)	2222 (1814 - 2620)	9.3% (7.6% - 11%)	11.8% (9.6% - 13.9%)
Philadelphia, PA	261 (214 - 308)	330 (270 - 389)	10.5% (8.6% - 12.3%)	13.2% (10.8% - 15.6%)
Phoenix, AZ	317 (258 - 374)	402 (327 - 476)	6.7% (5.5% - 7.9%)	8.5% (6.9% - 10.1%)
Pittsburgh, PA	256 (209 - 302)	324 (264 - 382)	9.3% (7.6% - 11%)	11.8% (9.6% - 13.9%)
Salt Lake City, UT	15 (12 - 18)	19 (16 - 23)	2.9% (2.4% - 3.4%)	3.7% (3% - 4.4%)
St. Louis, MO	446 (365 - 525)	563 (461 - 662)	11.2% (9.2% - 13.2%)	14.2% (11.6% - 16.7%)
Tacoma, WA	38 (31 - 46)	49 (40 - 58)	3.7% (3% - 4.4%)	4.7% (3.8% - 5.6%)

1 The current primary PM_{2.5} standards include an annual standard set at 15 µg/m³ and a 24-hour standard set at 35 µg/m³.

2 Numbers rounded to the nearest whole number. Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the PM coefficient.

3 Estimates based on Krewski et al. (2009), using ambient PM_{2.5} concentrations from 1979 - 1983 and from 1999-2000, respectively. Incidence is for 30+ year olds within each urban study area.

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Table 2-3. Estimated Incidence and Percent of Total Annual Incidence Associated with Short-Term PM_{2.5} Exposure Based on Simulation of the Current Suite of Standards (CV mortality and hospital admissions based on 2007 PM_{2.5} concentrations)^{1,2}

Risk Assessment Location	Incidence of Cardiovascular Mortality Associated with Short-term Exposure to PM _{2.5} ³	Incidence of Cardiovascular Hospitalizations Associated with Short-term Exposure to PM _{2.5} ⁴	Percent of Incidence of Cardiovascular Mortality Associated with Short-term Exposure to PM _{2.5} ³	Percent of Incidence of Cardiovascular Hospital Admissions Associated with Short-term Exposure to PM _{2.5} ⁴
Atlanta, GA	32 (-33 - 95)	41 (-27 - 109)	0.8% (-0.8% - 2.4%)	0.4% (-0.2% - 1%)
Baltimore, MD	62 (-4 - 126)	216 (159 - 273)	1.6% (-0.1% - 3.2%)	1.3% (1% - 1.7%)
Birmingham, AL	-1 (-42 - 40)	16 (-11 - 43)	0% (-1.5% - 1.5%)	0.3% (-0.2% - 0.9%)
Dallas, TX	29 (-19 - 76)	28 (-18 - 73)	0.8% (-0.5% - 2.2%)	0.3% (-0.2% - 0.7%)
Detroit, MI	60 (-8 - 127)	233 (171 - 295)	1% (-0.1% - 2.2%)	1.1% (0.8% - 1.4%)
Fresno, CA	12 (-9 - 33)	23 (0 - 46)	0.7% (-0.5% - 2%)	0.5% (0% - 0.9%)
Houston, TX	46 (-31 - 122)	56 (-37 - 149)	0.9% (-0.6% - 2.4%)	0.3% (-0.2% - 0.8%)
Los Angeles, CA	-30 (-132 - 72)	258 (3 - 511)	-0.2% (-0.7% - 0.4%)	0.5% (0% - 0.9%)
New York, NY	473 (276 - 668)	752 (552 - 951)	2.1% (1.2% - 3%)	1.2% (0.8% - 1.5%)
Philadelphia, PA	84 (22 - 145)	203 (149 - 257)	2.1% (0.5% - 3.6%)	1.3% (0.9% - 1.6%)
Phoenix, AZ	84 (-4 - 170)	108 (1 - 215)	1.3% (-0.1% - 2.7%)	0.5% (0% - 1%)
Pittsburgh, PA	43 (-9 - 93)	140 (103 - 177)	1.1% (-0.2% - 2.3%)	1.1% (0.8% - 1.4%)
Salt Lake City, UT	9 (-2 - 20)	9 (0 - 18)	0.8% (-0.2% - 1.7%)	0.4% (0% - 0.7%)
St. Louis, MO	106 (24 - 187)	178 (131 - 225)	1.9% (0.4% - 3.3%)	1.3% (0.9% - 1.6%)
Tacoma, WA	11 (-6 - 27)	19 (-46 - 82)	0.7% (-0.4% - 1.8%)	0.5% (-1.3% - 2.3%)

1 The current primary PM_{2.5} standards include an annual standard set at 15 µg/m³ and a 24-hour standard set at 35 µg/m³.

2 Percents rounded to the nearest tenth. Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the PM coefficient.

3 Based on location-specific single pollutant concentration-response function estimates from Zanobetti and Schwartz (2009) that have been "shrunken" towards the appropriate regional means. "Shrunken" coefficient estimates and their standard errors were sent to EPA by A. Zanobetti via email.

4 Incidence estimates were calculated using the appropriate regional concentration-response function estimates reported in Table 2 of Bell et al. (2008). Location-specific C-R function estimates were not available from this study.

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1 mortality attributable to long-term PM_{2.5} exposure (see U.S. EPA, 2010a, section 4.2.1).

2 In addition to long-term exposure-related mortality, we estimated cardiovascular and
3 respiratory-related mortality risk associated with short-term PM_{2.5} exposure. We note that these
4 mortality estimates are up to an order of magnitude smaller than estimates related to long-term
5 exposure-related mortality.²⁹ As part of the RA, we also estimated respiratory and
6 cardiovascular-related hospital admissions as well as asthma-related emergency department
7 visits associated with short-term exposure to PM_{2.5}, with estimates of cardiovascular and
8 respiratory hospital admissions together ranging up to approximately 1,000 admissions per year
9 across the urban study areas, with the estimated incidence of asthma-related emergency
10 department visits being several fold higher. Further, as discussed in section 2.2.1, we recognize
11 that the currently available scientific information includes evidence for a broader range of health
12 endpoints and susceptible populations beyond those included in the quantitative RA, including
13 lung function growth and respiratory symptoms in children and reproductive and developmental
14 effects. Taken together, the set of quantitative risk estimates related to long- and short-term
15 PM_{2.5} exposure, together with consideration of the health endpoints which could not be
16 quantified, further strengthen the conclusion that risks estimated to remain following simulated
17 attainment of the current suite of standards are important from a public health perspective, both
18 in terms of severity and magnitude.

19 **2.2.3 Staff Conclusions on Adequacy of Current Standards**

20 Collectively, taking into consideration the responses to specific questions focusing on
21 different ways to address the adequacy of the current suite of PM_{2.5} standards, we revisit the
22 overarching policy question: does the currently available scientific evidence and risk-based
23 information support or call into question the adequacy of the protection afforded by the current
24 suite of fine particle standards?

25 With respect to evidence-based considerations, the currently available evidence provides
26 stronger evidence beyond what was available in the last review, that associations between short-
27 and long-term PM_{2.5} exposures and a broad range of adverse health effects exist. The currently
28 available information strengthens the associations between PM_{2.5} and mortality and
29 cardiovascular and respiratory morbidity effects observed in the last review and expands our
30 understanding of a broader range of health outcomes as well as our understanding of effects in
31 susceptible populations. The currently available evidence provides support for associations that
32 extend to lower concentrations than what had been observed in the last review, including at

²⁹ Estimates of short-term exposure-related and long-term exposure-related mortality should not be added because there is the potential for overlap (i.e., the long-term exposure-related mortality estimate picking up some of the short-term exposure-related signal on a daily basis, aggregated over the year).

1 ambient concentrations below the levels of the current standards providing the basis for
2 consideration of alternative standards that would provide increased protection beyond that
3 afforded by the current PM_{2.5} standards.

4 In relation to risk-based considerations for informing our understanding of the adequacy
5 of the current fine particle standards, we focus on the estimates of PM_{2.5}-related mortality and
6 morbidity effects likely to remain upon simulations of just meeting the current standards in a
7 number of example urban areas. In considering the core risk estimates together with our
8 understanding of the uncertainties in these estimates, based upon extensive sensitivity analyses,
9 we conclude that the risks estimated to be associated with just meeting the current standards can
10 reasonably be judged to be important from a public health perspective. We further conclude that
11 these estimated risks provide support for consideration of alternative standards that would
12 provide increased protection beyond that afforded by the current PM_{2.5} standards.

13 We recognize that important uncertainties and research questions remain when
14 considering both evidence- and risk-based approaches. Nonetheless, we note that much progress
15 has been made in reducing some key uncertainties since the last review, including important
16 progress in advancing our understanding of potential mechanisms by which ambient PM_{2.5} is
17 causally linked with mortality, cardiovascular, and respiratory effects observed in
18 epidemiological studies. Additional information continues to emerge for a broader range of
19 health effects including reproductive and development effects and more information is available
20 to understand susceptible populations including children, older adults, individuals with pre-
21 existing cardiovascular and respiratory disease, persons at lower SES, and persons with genetic
22 susceptibility.

23 As was true in the last review, we recognize that as the body of available evidence has
24 expanded, it has added greatly both to our knowledge of health effects associated with fine
25 particle exposures, as well as to the complexity inherent in interpreting the evidence in a policy-
26 relevant context as a basis for setting appropriate standards. In evaluating both evidence-based
27 and risk-based considerations, along with associated limitations and uncertainties, we reach the
28 conclusion that the available information clearly calls into question the adequacy of the current
29 suite of PM_{2.5} standards and provides strong support for giving consideration to revising the
30 current standards to provide increased public health protection.

31 **2.3 CONSIDERATION OF ALTERNATIVE STANDARDS**

32 Having reached the conclusion that the currently available scientific evidence calls into
33 question the adequacy of the current suite of PM_{2.5} standards, staff considers a second
34 overarching question:

1 **What alternative suites of fine particle standards are supported by the currently available**
2 **scientific evidence and risk-based information, as reflected in the ISA and RA?**

3 To inform the answer to this overarching question, we have posed a series of more
4 specific questions to inform decisions regarding the basic elements of the NAAQS: indicator
5 (section 2.3.1), averaging time (section 2.3.2), form (section 2.3.3), and level (section 2.3.4).
6 These elements are considered collectively in evaluating the health protection afforded by
7 alternative suites of standards under consideration. In considering the currently available
8 scientific and technical information, we consider both the information available in the last review
9 and information that is newly available since the last review as assessed and presented in the ISA
10 and RA prepared for this review (US EPA, 2009a; US EPA, 2010a).

11 **2.3.1 Indicator**

12 In initially setting standards for fine particles in 1997, EPA concluded it was more
13 appropriate to control fine particles as a group, rather than singling out any particular component
14 or class of fine particles for which only very limited evidence was available. In establishing a
15 size-based indicator to distinguish fine particles from particles in the coarse mode, EPA noted
16 that the available epidemiological studies of fine particles were based largely on PM_{2.5} and also
17 considered monitoring technology that was generally available. The selection of a 2.5 µm size
18 cut reflected the regulatory importance of defining an indicator that would more completely
19 capture fine particles under all conditions likely to be encountered across the U.S., especially
20 when fine particle concentrations are likely to be high, while recognizing that some small coarse
21 particles would also be captured by current methods to monitor PM_{2.5} (62 FR 38667 to 38668,
22 July, 18, 1997). In the last review, based on the same considerations, EPA again recognized that
23 the available information supported retaining the PM_{2.5} indicator and remained too limited to
24 support a distinct standard for any specific PM_{2.5} component or fine particle source (71 FR 61162
25 to 61164, October 17, 2006).

26 • **Does the currently available information provide support for the continued use of a**
27 **PM_{2.5} mass-based indicator for fine particles?**

28 In this review, epidemiological studies linking cardiovascular and respiratory effects as
29 well as mortality with long- and short-term fine particle exposures continue to be largely indexed
30 by PM_{2.5}. Based on the same considerations that informed the last two reviews, we again
31 conclude that it is appropriate to retain a PM_{2.5} indicator to provide protection associated with
32 long- and short-term exposure to fine particles.

1 We also look to the expanded body of evidence available in this review to consider
2 whether there is sufficient evidence to support a separate standard for ultrafine particles³⁰ (UFPs)
3 and whether there is sufficient evidence to establish distinct standards focused on regulating
4 specific PM_{2.5} components or sources of fine particles, as addressed below.

5 • **To what extent does the currently available information provide support for**
6 **considering a separate indicator for UFPs?**

7 A number of studies available in this review have focused on UFPs, as a subset of PM_{2.5}.
8 As noted in the ISA, the enormous number and larger surface area of UFPs are important
9 considerations for focusing on this subfraction of fine particles in assessing potential public
10 health impacts (US EPA, 2009a, p. 6-83).³¹ Per unit mass, UFPs may have more opportunity to
11 interact with cell surfaces due to their greater surface area and their greater particle number
12 compared with larger particles (US EPA, 2009a, p. 5-3). Greater surface area also increases the
13 potential for soluble components (e.g., transition metals, organics) to adsorb to UFPs and
14 potentially cross cell membranes and epithelial barriers (US EPA, 2009a, p. 6-83). In
15 addition, evidence available in this review suggests that the ability of particles to enhance
16 allergic sensitization is associated more strongly with particle number and surface area than
17 particle mass (US EPA, 2009a, p. 6-127).³²

18 New evidence, primarily from controlled human exposure and toxicological studies,
19 expands our understanding of UFP-related cardiovascular and respiratory effects. However, this
20 evidence is still very limited and largely focused on exposure to diesel exhaust (DE), for which
21 the ISA concludes “it is unclear if the effects observed are due to UFP, larger particles (i.e.,
22 PM_{2.5}), or the gaseous components of DE” (US EPA, 2009a, p. 2-22). In addition, the ISA notes
23 uncertainties associated with the controlled human exposure studies as concentrated ambient
24 particle (CAP) systems have been shown to modify the composition of UFPs (US EPA, 2009a, p.
25 2-22, see also section 1.5.3). Relatively few epidemiological studies have examined potential
26 cardiovascular and respiratory effects associated with short-term exposures to UFPs. These
27 studies have reported inconsistent and mixed results (US EPA, 2009a, section 2.3.5).

28 Collectively, in considering the body of scientific evidence available in this review, the
29 ISA concludes that the currently available evidence is *suggestive of a causal association* between
30 short-term exposures to UFPs and cardiovascular and respiratory effects. Furthermore, the ISA

³⁰ Ultrafine particles, generally including particles with a nominal aerodynamic diameter less than 0.1 μm, are emitted directly to the atmosphere or are formed by nucleation of gaseous constituents in the atmosphere (US EPA, 2009a, p. 3-3).

³¹ Particle number is most highly concentrated in the UFP fraction with volume (or mass) most concentrated in the larger size fractions (US EPA, 2009a, p. 3-2).

³² More information on possible modes of action for effects associated with UFPs exposures is discussed in sections 5.1 and 5.4 of the ISA.

1 concludes that evidence is *inadequate to infer a causal association* between short-term exposure
2 to UFPs and mortality as well as long-term exposure to UFPs and all outcomes evaluated (US
3 EPA, 2009a, sections 2.3.5, 6.2.12.3, 6.3.10.3, 6.5.3.3, 7.2.11.3, 7.3.9, 7.4.3.3, 7.5.4.3, and
4 7.6.5.3; Table 2-6).

5 With respect to our understanding of ambient UFP concentrations, at present, there is no
6 national network of UFP samplers; thus, only episodic and/or site-specific data sets exist (US
7 EPA, 2009a, p. 2-2).³³ Therefore, a national characterization of concentrations, temporal and
8 spatial patterns, and trends is not possible, and the availability of ambient UFP measurements to
9 support health studies are extremely limited. In general, measurements of UFPs are highly
10 dependent on monitor location and, therefore, more subject to exposure error than accumulation
11 mode particles (US EPA, 2009a, p. 2-22). The UFP number concentrations fall off sharply
12 downwind from sources, as UFPs may grow into the accumulation mode by coagulation or
13 condensation (US EPA, 2009a, p. 3-89). Limited studies of UFP ambient measurements suggest
14 these particles exhibit a high degree of spatial and temporal heterogeneity driven primarily by
15 differences in nearby source characteristics (US EPA, 2009a, p. 3-84). Internal combustion
16 engines and, therefore, roadways are a notable source of UFPs, so concentrations of UFPs near
17 roadways are generally expected to be elevated (US EPA, 2009a, p. 2-3). Concentrations of
18 UFPs have been reported to drop off much more quickly with distance from roadways than
19 larger particle sizes (US EPA, 2009a, p. 3-84).

20 In considering both the currently available health effects evidence and the air quality data
21 for UFPs, we conclude that this information is still too limited to support a distinct PM standard
22 for UFPs.

- 23 • **To what extent does the currently available information provide support for**
24 **considering a separate indicator for a specific PM_{2.5} component or source category of**
25 **fine particles? Conversely, to what extent does the currently available information**
26 **provide support for eliminating any component or source category from the mix of fine**
27 **particles included in the PM_{2.5} indicator?**

28 In addressing the issue of particle composition, the ISA concludes that, “[f]rom a
29 mechanistic perspective, it is highly plausible that the chemical composition of PM would be a
30 better predictor of health effects than particle size” (US EPA, 2009a, p. 6-202). Heterogeneity of
31 ambient concentrations of PM_{2.5} constituents (e.g., elemental carbon (EC), organic carbon (OC),
32 sulfates, and nitrates) observed in different geographical regions as well as regional
33 heterogeneity in PM_{2.5}-related health effects reported in a number of epidemiological studies are
34 consistent with this hypothesis (US EPA, 2009a, section 6.6).

³³ The ISA contains a review of the current scientific information related to measurements of UFPs (US EPA, 2009a, sections 3.5.1 and 3.5.2).

1 With respect to the availability of ambient measurement data for fine particle components
2 in this review, there are now more extensive ambient PM_{2.5} speciation measurement data
3 available through the Chemical Speciation Network (CSN).³⁴ Data from the CSN monitoring
4 network provide further evidence of spatial and seasonal variation in both PM_{2.5} mass and
5 composition among cities/regions (US EPA, 2009a, pp. 3-50 to 3-60; Figures 3-12 to 3-18;
6 Figure 3-47). Some of this variation may be related to regional differences in meteorology,
7 sources, and topography (US EPA, 2009a, p. 2-3).

8 The currently available epidemiological, toxicological, and controlled human exposure
9 studies have evaluated the health effects associated with ambient PM_{2.5} constituents and
10 categories of fine particle sources, using a variety of quantitative methods applied to a broad set
11 of PM_{2.5} constituents, rather than selecting a few constituents a priori (US EPA, 2009a, p. 2-26).
12 Epidemiological studies have used measured ambient PM_{2.5} speciation data, including
13 monitoring data from the CSN, while all of the controlled human exposure and most of the
14 toxicological studies have used CAPs and analyzed the constituents therein (US EPA, 2009a, p.
15 6-203).³⁵ The CSN provides PM_{2.5} speciation measurements generally on a one-in-three or one-
16 in-six day schedule and, thus, do not capture data every day at most sites. To expand our
17 understanding of the role of specific PM_{2.5} components and sources with respect to the observed
18 health effects, researchers have expressed a strong interest in having access to PM_{2.5} speciation
19 measurements collected more frequently.³⁶

20 With respect to epidemiological studies evaluating short-term exposures to fine particle
21 constituents, several new multi-city studies are now available. These studies continue to show an
22 association between mortality and cardiovascular and/or respiratory morbidity effects and short-

³⁴The CSN consists of 54 Speciation Trends Network (STN) sites as well as about 150 SLAMS supplemental sites across the country measuring over 40 chemical species. A limited number of CSN monitors began collecting ambient data in 2000 with the majority of sites collecting data starting in 2001. These sites collect aerosol samples over 24 hours on filters that are analyzed for PM_{2.5} mass, trace elements, major ions (e.g., sulfates, nitrates, ammonium), and EC/OC.

³⁵ Most studies considered between 7 to 20 ambient PM_{2.5} constituents, with EC, OC, sulfate, nitrate, and metals most commonly measured. Many of the studies grouped the constituents with various factorization or source apportionment techniques to examine the relationship between the grouped constituents and various health effects. However, not all studies labeled the constituent groupings according to their presumed source and a small number of controlled human exposure and toxicological studies did not use any constituent grouping. These differences across studies substantially limit any integrative interpretation of these studies (US EPA, 2009a, p. 6-203).

³⁶ As outlined in section 6.6.2.11 of the ISA, some investigators have circumvented the issue of less than daily speciation data by using the PM_{2.5} chemical species data in a second stage regression to explain the heterogeneity in PM₁₀ or PM_{2.5} mortality risk estimates across cities and assuming that the relative contributions of PM_{2.5} have remained the same over time (US EPA, 2009a, p. 6-206). In April 2008, EPA co-sponsored a workshop to discuss modifications to the current ambient air quality monitoring networks that would advance our understanding of the impacts of PM exposures on public health/welfare in the most meaningful way, including improving our understanding of fine particle components. A summary of the workshop recommendations, including recommendations for daily PM_{2.5} speciation measurements in large urban areas, is available at www.epa.gov/ORD/npd/pdfs/FINAL-April-2008-AQ-Health-Research-Workshop-Summary-Dec-2008.pdf.

1 term exposures to various PM_{2.5} components including nickel (Ni), vanadium (V), EC, OC, and
2 sulfates (US EPA, 2009a, sections 6.5.2.5 and 6.6). Lippmann et al. (2006) and Dominici et al.
3 (2007) evaluated the heterogeneity in the PM₁₀–mortality association as evaluated in the
4 NMMAPS data by analyzing the PM_{2.5} speciation data. Nickel and V were identified as
5 significant predictors of variation in PM₁₀-related mortality across cities, with Ni levels in New
6 York City being reported as particularly high (US EPA, 2009a, section 6.5.2.5; Figure 6-31).³⁷
7 Bell et al. (2009) and Peng et al. (2009) conducted similar analyses focusing on the variation in
8 PM_{2.5}-related cardiovascular and respiratory hospital admissions in older adults. Peng et al.
9 (2009) focused on the components that make up the majority of PM_{2.5} mass and using multi-
10 pollutant models reported only EC and OC were significantly associated with risk of
11 hospitalization for cardiovascular disease. Bell et al. (2009) used data from twenty PM_{2.5}
12 components and found that EC, Ni, and V were most positively and significantly associated with
13 the risk of PM_{2.5}-related hospitalizations suggesting that the observed associations between PM_{2.5}
14 and hospitalizations may be primarily due to particles from oil combustion and traffic (US EPA,
15 2009a, section 6.2.10.1). In a study of 25 U.S. cities, Franklin et al. (2008) focused on a time-
16 series regression of mortality related to PM_{2.5} mass by season and also examined effect
17 modification due to various PM_{2.5} species. They concluded that Al, As, Ni, Si and sulfates were
18 significant effect modifiers of PM_{2.5} mortality risk estimates, and “simultaneously including Al,
19 Ni, and sulfates together or Al, Ni, and As together further increased explanatory power. Of the
20 species examined, Al and Ni explained the most residual heterogeneity” (US EPA, 2009a, p. 6-
21 194; Table 6-17).³⁸ Furthermore, Ostro et al (2006) examined associations between PM_{2.5}
22 components and mortality in six California counties and found an association between mortality,
23 especially cardiovascular-related mortality and several PM_{2.5} components including EC, OC,
24 nitrates, iron (Fe), potassium (K), and titanium (Ti) at various lags (US EPA, 2009a, p. 6-195).

25 Limited evidence is available to evaluate the health effects associated with long-term
26 exposures to PM_{2.5} components (US EPA, 2009a, section 7.6.2). The most significant new
27 evidence is provided by a study that evaluated multiple PM_{2.5} components and an indicator of
28 traffic density in an assessment of health effects related to long-term exposure to PM_{2.5} (Lipfert
29 et al., 2006). Using health data from a cohort of U.S. military veterans and PM_{2.5} data from
30 EPA’s CSN, Lipfert et al. (2006) reported positive associations between mortality and long-term
31 exposures to nitrates, EC, Ni and V as well as traffic density and peak O₃ concentrations.
32 Additional evidence from a long-term exposure study conducted in a Dutch cohort provides

³⁷ However, as noted in the ISA, in a sensitivity analysis when selectively removing cities from the overall estimate, the significant association between the PM₁₀ mortality risk estimate and the PM_{2.5} Ni fraction was diminished upon removing New York City from the analysis, which is consistent with the results presented by Dominici et al. (2007) (US EPA, 2009a, section 6.5.2.5; Figure 6-32).

³⁸ We note that New York City was not included in the 25 cities examined by Franklin et al. (2008).

1 supportive evidence that long-term exposure to traffic-related particles is associated with
2 increased mortality (Breelen et al., 2008).

3 With respect to source categories of fine particles associated with a range of health
4 endpoints, the ISA reports that currently available evidence suggests associations between
5 cardiovascular effects and a number of specific PM_{2.5}-related source categories, specifically oil
6 combustion, wood or biomass burning, motor vehicle emissions, and crustal or road dust sources
7 (US EPA, 2009a, section 6.6; Table 6-18). In addition, a few studies have evaluated associations
8 between PM_{2.5}-related source categories and mortality. These studies included a reported
9 association between mortality and a PM_{2.5} coal combustion factor (Laden et al., 2000), while
10 others linked mortality to a secondary sulfate long-range transport PM_{2.5} source (Ito et al., 2006;
11 Mar et al., 2006) (US EPA, 2009a, section 6.6.2.1). There is less consistency in associations
12 observed between PM_{2.5} sources and respiratory health effects, which may be partially due to the
13 fact that fewer studies have evaluated respiratory-related outcomes and measures. However,
14 there is some evidence for associations with secondary SO₄²⁻ and decrements in lung function in
15 asthmatic and healthy adults (US EPA, 2009a, p. 6-211; Gong et al., 2005; Lanki et al., 2006).
16 Respiratory effects relating to the crustal/soil/road dust and traffic sources of PM have been
17 observed in asthmatic children and adults (US EPA, 2009a, p. 6-205; Gent et al., 2009; Penttinen
18 et al., 2006).

19 Recent studies have shown that source apportionment methods have the potential to add
20 useful insights into which sources and/or PM constituents may contribute to different health
21 effects. Of particular interest are several epidemiological studies that compared source
22 apportionment methods and reported consistent results across research groups (US EPA, 2009a,
23 p. 6-211; Hopke et al., 2006; Ito et al., 2006; Mar et al., 2006; Thurston et al., 2005). These
24 studies reported associations between total mortality and secondary sulfate in two cities for two
25 different lag times. The sulfate effect was stronger for total mortality in Washington D.C. and
26 for cardiovascular-related mortality in Phoenix (US EPA, 2009a, p. 6-204). These studies also
27 found some evidence for associations with mortality and a number of source categories (e.g.,
28 biomass/wood combustion, traffic, copper smelter, coal combustion, sea salt) at various lag times
29 (US EPA, 2009a, p. 6-204). Sarnat et al., (2008) compared three different source apportionment
30 methods and reported consistent associations between ED visits for cardiovascular diseases with
31 mobile sources and biomass combustion as well as increased respiratory-related ED visits
32 associated with secondary sulfate (US EPA, 2009a, pp. 6-204 and 6-211; Sarnat et al., 2008).

33 In summary, in considering the currently available evidence for health effects associated
34 with chemical components and source categories of PM_{2.5} as presented in the ISA, we conclude
35 that additional information available in this review continues to provide evidence that many
36 different constituents of the fine particle mixture as well as specific source categories of fine

1 particles are linked to adverse health effects. However, as noted in the ISA, while “[t]here is
2 some evidence for trends and patterns that link particular ambient PM constituents or sources
3 with specific health outcomes...there is insufficient evidence to determine whether these patterns
4 are consistent or robust” (US EPA, 2009a, p. 6-210). Furthermore, the ISA concludes that “the
5 evidence is not yet sufficient to allow differentiation of those constituents or sources that are
6 more closely related to specific health outcomes” (US EPA, 2009a, pp. 2-26 and 6-212).
7 Therefore, we conclude that the currently available evidence is not sufficient to support
8 consideration of a separate indicator for a specific PM_{2.5} component or source category of fine
9 particles. We also conclude that the evidence is not sufficient to support eliminating any
10 component or source from the mix of fine particles included in the PM_{2.5} indicator.

11 Summary

12 In considering whether currently available evidence provides support for retaining,
13 revising, or supplementing the current PM_{2.5} mass-based indicator, we first conclude that it is
14 appropriate to retain PM_{2.5} as the indicator for fine particles. Secondly, we conclude that the
15 currently available evidence does not provide a sufficient basis for supplementing the mass-
16 based PM_{2.5} indicator by considering a separate indicator for ultrafine particles as a subfraction
17 of fine particles. We also conclude that the currently available evidence is too limited to provide
18 support for considering a separate indicator for a specific PM_{2.5} component or source category of
19 fine particles or for eliminating any individual component or source category from the mix of
20 fine particles included in the PM_{2.5} mass-based indicator.

21 **2.3.2 Averaging Times**

22 In 1997, EPA initially set both an annual standard, to provide protection from health
23 effects associated with both long- and short-term exposures to PM_{2.5}, and a 24-hour standard to
24 supplement the protection afforded by the annual standard. In the last review, EPA retained both
25 annual and 24-hour averaging times (62 FR 38667 to 38668, July, 18, 1997).

26 In this review, we consider whether the currently available information provides support
27 for maintaining standards with annual and 24-hour averaging times and whether there is
28 sufficient evidence to support setting standards with other averaging times to address subdaily or
29 seasonal exposures.

30 • **To what extent does the currently available information continue to provide support for** 31 **annual and 24-hour averaging times?**

32 The overwhelming majority of studies conducted since the last review continue to utilize
33 annual and 24-hour averaging times, and largely contribute to the body of evidence for health
34 effects related to both short-term (from less than 1 day to up to several days) and long-term

1 (from a year to several years) measures of PM_{2.5}. Consequently, we conclude that the currently
2 available evidence continues to support annual and 24-hour averaging times.

3 • **To what extent does the currently available scientific evidence provide support for**
4 **considering a standard with an averaging time less than 24 hours to address health**
5 **effects associated with subdaily fine particle exposures?**

6 Relative to information available in the last review, recent studies provide additional
7 evidence for cardiovascular effects associated with subdaily (e.g., one to several hours) exposure
8 to PM, especially effects related to cardiac ischemia, vasomotor function, and more subtle
9 changes in markers of systemic inflammation, hemostasis, thrombosis and coagulation (US EPA,
10 2009a, section 6.2).³⁹ Because these studies have used different indicators (e.g., PM_{2.5}, PM₁₀,
11 PM_{10-2.5}, UFPs), averaging times (e.g., 1, 2, and 4 hours), and health outcomes, it is difficult to
12 draw conclusions about cardiovascular effects associated specifically with subdaily exposures to
13 PM_{2.5}.

14 With regard to respiratory effects associated with subdaily PM_{2.5} exposures, the currently
15 available evidence is much sparser than for cardiovascular effects and continues to be very
16 limited. The ISA concludes that for several studies of hospital admissions or medical visits for
17 respiratory diseases, the strongest associations were observed with 24-hour average or longer
18 exposures, not with less than 24-hour exposures (US EPA, 2009a, section 6.3).

19 Collectively, we conclude that this information is too unclear, with respect to the
20 indicator, averaging time and health outcome, to serve as a basis for establishing a shorter-than-
21 24-hour PM_{2.5} primary standard at this time.

22 • **To what extent does the currently available scientific evidence provide support for**
23 **considering separate standards with distinct averaging times to address effects**
24 **associated with seasonal fine particle exposures?**

25 With regard to health effects associated with PM_{2.5} exposure across varying seasons in
26 this review, Bell et al. (2008) reported higher PM_{2.5} risk estimates for hospitalization for
27 cardiovascular and respiratory diseases in the winter compared to other seasons. In comparison
28 to the winter season, smaller statistically significant associations were also reported between
29 PM_{2.5} and cardiovascular morbidity for spring and autumn, and a positive non-significant
30 association was observed for the summer months. In the case of mortality, Zanobetti and
31 Schwartz (2009) reported a 4-fold higher effect estimate for PM_{2.5} associated mortality for the
32 spring as compared to the winter. Taken together, these results are inconsistent, suggesting

³⁹ A limited number of additional studies have also provided evidence of reported electrocardiogram changes typically representative of cardiac ischemia (S-T segment depression) or reported changes in heart rate variability (HRV) (US EPA, 2009a, sections 6.2.1.2 and 6.2.12.1), however, these changes are often variable and difficult to interpret the PM_{2.5} etiologically relevant mechanism underlying the observed effects.

1 individuals are at greater risk of dying from higher exposures to PM_{2.5} in the warmer months, and
2 at greater risk of PM associated hospitalization for cardiovascular and respiratory diseases during
3 colder months of the year.

4 Overall, we observe that there are few studies presently available to deduce a general
5 pattern in PM_{2.5}-related risk across seasons. In addition, these studies utilized 24-hour exposure
6 periods within each season to assess the PM_{2.5} associated health effects, and do not provide
7 information on health effects associated with a season-long exposure to PM_{2.5}. Due to these
8 limitations in the currently available evidence, we conclude that there is no basis to consider a
9 seasonal averaging time separate from a 24-hour averaging time.

10 Summary

11 We recognize that the currently available evidence informs our understanding of
12 exposure durations of concern and continues to provide strong support for standards that provide
13 protection for both short- and long-term exposures. In considering the possibility of effects
14 associated with subdaily PM_{2.5} exposures (i.e., less than 24-hour exposures), we recognize that
15 there is additional evidence available in this review, primarily focused on cardiovascular effects
16 with more limited evidence for respiratory effects. However, because these studies have used
17 different indicators of PM exposure (e.g., PM_{2.5}, PM₁₀, UFPs), averaging times, and a broad
18 range of health outcomes, it is difficult to use this evidence to serve as a basis for establishing a
19 standard with a shorter-than-24-hour averaging time. With respect to seasonal effects, while we
20 recognize there is some new evidence for PM_{2.5}-related effects differentiated by season, we
21 conclude that this evidence is too limited to use as a basis for establishing a PM_{2.5} standard with a
22 seasonal averaging time. Based on the above considerations, we conclude that the currently
23 available information provides strong support for retaining the current annual and 24-hour
24 averaging times but does not provide support for considering alternative averaging times.

25 **2.3.3 Forms**

26 The “form” of a standard defines the air quality statistic that is to be compared to the
27 level of the standard in determining whether an area attains the standard. In this review, we
28 consider whether currently available information supports consideration of alternative forms for
29 the annual or 24-hour PM_{2.5} standards.

30 **2.3.3.1 Form of the Annual Standard**

31 In 1997, EPA established the form of the annual PM_{2.5} standard as an annual arithmetic
32 mean, averaged over 3 years, from single or multiple community-oriented monitors. This form
33 was intended to represent a relatively stable measure of air quality and to characterize area-wide
34 PM_{2.5} concentrations. The level of the standard was to be compared to measurements made at

1 the community-oriented monitoring site recording the highest level, or, if specific constraints
2 were met⁴⁰, measurements from multiple community-oriented monitoring sites could be
3 averaged (62 FR 38671 to 38672, July 18, 1997). The constraints were intended to ensure that
4 spatial averaging would not result in inequities in the level of protection (62 FR 38672). This
5 approach was consistent with the epidemiological studies on which the PM_{2.5} standard was
6 primarily based, in which air quality data were generally averaged across multiple monitors in an
7 area or were taken from a single monitor that was selected to represent community-wide
8 exposures, not localized “hot spots.”

9 In the last review, EPA tightened the criteria for use of spatial averaging⁴¹ to provide
10 increased protection for vulnerable populations exposed to PM_{2.5}. This change was based in part
11 on an analysis of the potential for disproportionate impacts on potentially vulnerable populations,
12 which found that the highest concentrations in an area tend to be measured at monitors located in
13 areas where the surrounding population is more likely to have lower education and income
14 levels, and higher percentages of minority populations (71 FR 61166/2; US EPA, 2005, section
15 5.3.6.1).

16 In this review, we again consider the potential impact of allowing for spatial averaging,
17 noting that persons from lower socioeconomic strata have been identified as an additional
18 susceptible population (see section 2.2.1).

19 • **Does the currently available evidence provide support for the continued use of spatial**
20 **averaging as part of the form of the annual standard?**

21 In considering the potential for disproportionate impacts on potentially vulnerable
22 populations, we updated analyses conducted for the last review. Specifically, we evaluated
23 whether persons with a lower SES status are more likely than the general population to live in
24 census tracts in which the monitors recording the highest air quality values in an area are located.
25 Data used in this analysis included demographic parameters measured at the census tract level,
26 including income level and percent minority population. Data from the census tract in each area
27 in which the highest air quality value was monitored were compared to the area-wide average
28 value (consistent with the constraints on spatial averaging provided by the current standard) in
29 each area (Schmidt, 2010).

30 Recognizing the limitations of such cross-sectional analyses, we observe that the highest
31 concentrations in an area tend to be measured at monitors located in areas where the surrounding

⁴⁰ The original criteria for spatial averaging included: (1) the annual mean concentration at each site shall be within 20 percent of the spatially averaged annual mean, and (2) the daily values for each monitoring site pair shall yield a correlation coefficient of at least 0.6 for each calendar quarter.

⁴¹ The current criteria for spatial averaging include: (1) the annual mean concentration at each site shall be within 10 percent of the spatially averaged annual mean, and (2) the daily values for each monitoring site pair shall yield a correlation coefficient of at least 0.9 for each calendar quarter (71 FR 61167/2-3, October 17, 2006).

1 populations are more likely to have lower income levels and higher percentage of minority levels.
2 Based upon this analysis, we believe that the existing constraints on spatial averaging, as
3 modified in 2006, may not be adequate to avoid substantially greater exposures in some areas,
4 potentially resulting in disproportionate impacts on persons with lower SES levels and
5 minorities. Therefore, we conclude that it is appropriate to consider a form for the annual PM_{2.5}
6 standard that does not allow for the use of spatial averaging across monitors, such that the level
7 of the annual PM_{2.5} standard would be compared to measurements made at the monitoring site
8 that represents “community-wide air quality” recording the highest PM_{2.5} concentrations.

9 **2.3.3.2 Form of the 24-Hour Standard**

10 In 1997, EPA established the form of the 24-hour PM_{2.5} standard as the 98th percentile of
11 24-hour concentrations at each population-oriented monitor within an area, averaged over three
12 years (62 FR at 38671 to 38674, July 18, 1997). The Agency selected the 98th percentile as an
13 appropriate balance between adequately limiting the occurrence of peak concentrations and
14 providing increased stability which, when averaged over 3 years, facilitated the development of
15 more stable implementation programs. By basing the form of the standard on concentrations
16 measured at population-oriented monitoring sites, EPA intended to provide protection for people
17 residing in or near localized areas of elevated concentrations. In the last review, in conjunction
18 with lowering the level of the 24-hour standard, EPA retained this form based in part on a
19 comparison with the 99th percentile form.⁴²

20 In this review, we have again considered the relative stability of the 98th and 99th
21 percentile forms.

22 • **Does the currently available evidence provide support for the continued use of the 98th** 23 **percentile form of the 24-hour standard?**

24 We recognize that the selection of the appropriate form of the 24-hour standard includes
25 maintaining adequate protection against peak 24-hour concentrations while also providing a
26 stable target for risk management programs, which serves to provide for the most effective
27 public health protection in the long run.⁴³ As in previous reviews, we recognize that a
28 concentration-based form provides proportionally greater weight to days when concentrations are

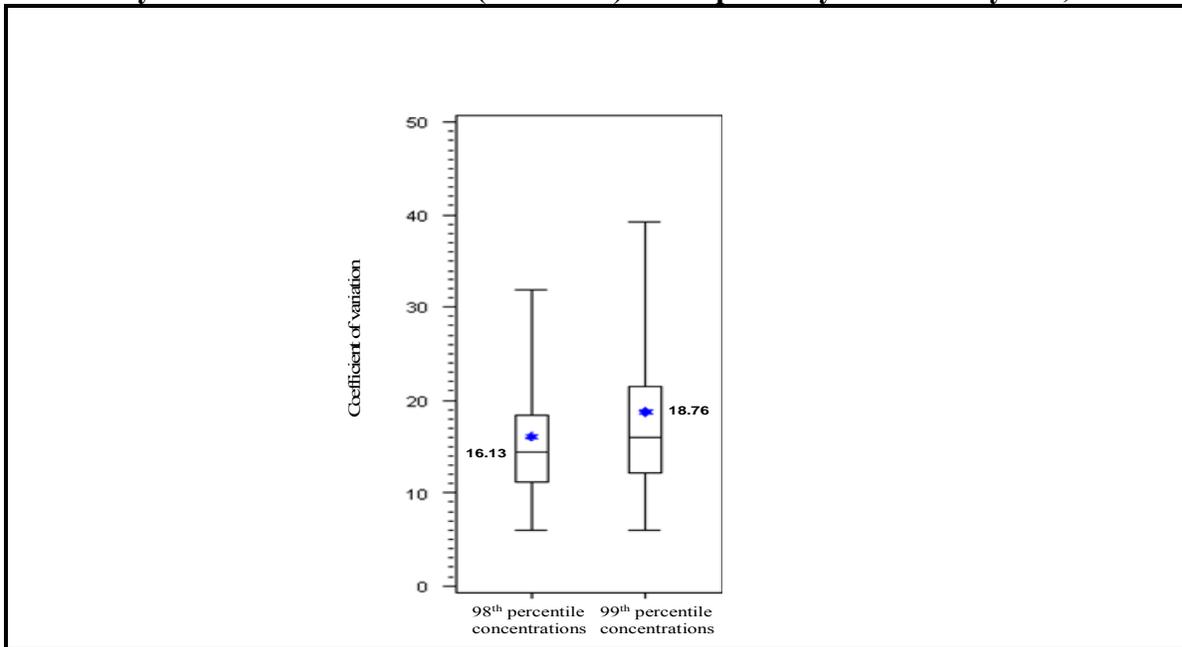
⁴² In reaching this final decision, EPA recognized a technical problem associated with a potential bias in the method used to calculate the 98th percentile concentration for this form. EPA adjusted the sampling frequency requirement in order to reduce this bias. Accordingly, the Agency modified the final monitoring requirements such that areas that are within 5 percent of the standards are required to increase the sampling frequency to every day (71 FR 61164 to 61165, October 17, 2006).

⁴³ See *ATA III*, 283 F.3d at 374-375 which concludes it is legitimate for EPA to consider promotion of overall effectiveness of NAAQS implementation programs, including their overall stability, in setting a standard that is requisite to protect the public health.

1 well above the level of the standard than to days when the concentrations are just above the
2 standard. Further, a concentration-based form compensates for missing data and less-than-every-
3 day monitoring; and, when averaged over 3 years, it has greater stability and, thus, facilitates the
4 development of more stable implementation programs.

5 In revisiting the stability of a 98th versus 99th percentile form for a 24-hour standard
6 intended to provide supplemental protection for a generally controlling annual standard, we
7 consider air quality data reported in 2000 to 2008 to update our understanding of the ratio
8 between peak-to-mean PM_{2.5} concentrations.⁴⁴ As illustrated in Figure 2-2, the 98th percentile
9 value is a more stable metric than the 99th percentile.

10 **Figure 2-2. Distribution of site-level variation in 98th and 99th percentile concentrations, as**
11 **measured by coefficient of variation (SD/Mean)⁴⁵ computed by site across years, 2000-2008**



12
13 On this basis, we conclude that it is appropriate to consider retaining the current 98th
14 percentile form of the 24-hour standard as it represents an appropriate balance between
15 adequately limiting the occurrence of peak concentrations and providing increased stability
16 relative to an alternative 99th percentile form. In addition, by basing the form of the standard on
17 concentrations measured at population-oriented monitoring sites, the standard would continue to
18 focus on providing protection for people residing in or near localized areas of elevated
19 concentrations.

⁴⁴ We consider a coefficient of variation instead of simply the standard deviation because the 99th percentile values have higher concentration levels and dividing by the mean normalizes the data. In focusing on three years of recent air quality (2006 to 2008), we see a similar pattern of peak-to-mean ratios (Schmidt, 2010).

⁴⁵ Coefficient of variation x 100.

1 **2.3.4 Alternative Levels**

2 In reaching staff conclusions for alternative standard levels that are appropriate to
3 consider, we take into account both evidence-based (section 2.3.4.1) and risk-based
4 considerations (section 2.3.4.2) as well as the related limitations and uncertainties associated
5 with this information as presented and discussed more fully in the ISA and RA (US EPA, 2009a;
6 US EPA, 2010a). Alternative levels are discussed in conjunction with staff conclusions on other
7 elements of the standard presented above, notably, retaining PM_{2.5} as the indicator for fine
8 particles (see section 2.3.1); retaining the current annual and 24-hour averaging times (see
9 section 2.3.2); modifying the current form of the annual standard (see section 2.3.3.1) and
10 retaining the current form of the 24-hour standard (see section 2.3.3.2). Specifically, we address
11 the following overarching question:

12 **What alternative standard levels are appropriate to consider to provide requisite**
13 **protection for long- and short-term PM_{2.5} exposures?**

14 Staff conclusions based on the integration of the evidence-based and risk-based
15 approaches are presented in section 2.3.4.3.

16 **2.3.4.1 Evidence-based Considerations**

17 In translating information from epidemiological studies into the basis for reaching staff
18 conclusions on alternative standard levels, we apply the policy framework outlined in section
19 2.1.3. In doing so, we focus on identifying levels for a generally controlling annual standard and
20 a 24-hour standard that provides supplemental protection against days with high peak
21 concentrations especially in areas with high peak-to-mean ratios, possibly associated with strong
22 local or seasonal sources, or for potential PM_{2.5}-related effects that may be associated with
23 shorter-than-daily exposure periods, that might not be well controlled by an annual standard.

24 We address a series of specific questions beginning with consideration of the relative
25 weight to place on different evidence. Using the available epidemiological evidence, we then
26 consider two approaches for identifying alternative annual standard levels: (1) looking directly
27 at confidence bounds on C-R relationships reported in long- and short-term exposure studies and
28 (2) exploring different statistical metrics that characterize air quality distributions from multi-city
29 epidemiological studies. We also consider information on peak-to-mean air quality ratios in
30 reaching conclusions regarding 24-hour standard levels that are appropriate to consider.

- 31 • **What factors do we weigh in placing emphasis on epidemiological evidence to translate**
32 **this information into staff conclusions on alternative standard levels?**

33 As discussed in section 2.1.3, we initially focus on long- and short-term PM_{2.5} multi-city
34 exposure studies conducted in the U.S. and Canada and place the greatest weight on associations

1 that have been judged in the ISA to be causal or likely causal. We also consider the evidence for
2 a broader range of health outcomes judged in the ISA to have suggestive evidence of a causal
3 association, specifically studies that focus on effects in susceptible populations, to evaluate
4 whether this evidence provides support for considering lower alternative levels.

5 We take several factors into account in placing relative weight on the body of available
6 epidemiological studies, for example, study characteristics, including study design (e.g., time
7 period of air quality monitoring, control for potential confounders); strength of the study (in
8 terms of statistical significance and precision of result); and availability of air quality distribution
9 data. We place greatest weight on information from multi-city epidemiological studies. These
10 studies have a number of advantages compared to single-city studies⁴⁶ that include providing
11 representation of ambient PM_{2.5} concentrations and potential health impacts across a range of
12 diverse locations providing spatial coverage for different regions across the country, reflecting
13 differences in PM_{2.5} sources, composition, and potentially other exposure-related factors which
14 might impact PM_{2.5}-related risks; lack of ‘publication bias’ (US EPA, 2004a, p. 8-30); and
15 consideration of larger study populations that afford the possibility of generalizing to the broader
16 national population and provide higher statistical power than single-city studies to detect
17 potentially statistically significant associations with relatively more precise effect estimates.

18 • **To what extent have confidence bounds on concentration-response relationships**
19 **reported in epidemiological studies been characterized?**

20 As outlined in section 2.1.3, to the extent analyses characterizing confidence bounds on
21 concentration-response (C-R) relationships have been published from both long- and short-term
22 exposure studies we have explored this evidence in reaching staff conclusions on the range of
23 annual standard levels that is appropriate to consider. We note that although epidemiological
24 studies reporting C-R functions and associated 95% confidence intervals provide information on
25 the precision of the effect estimates across the air quality distribution, these analyses do not
26 provide evidence of a concentration below which the confidence interval becomes notably wider
27 and uncertainty in a C-R relationship substantially increases. If an analysis characterizing the
28 confidence bounds on the C-R relationships is available and indicates an unacceptable degree of
29 uncertainty about the existence of a continuing C-R relationship, the possibility that an effects
30 threshold may exist becomes more likely.

⁴⁶ As discussed in section 2.2.1, we recognize that single-city studies provide ancillary evidence to multi-city studies in support of calling into question the adequacy of the current suite of standards. However, in light of the mixed findings reported in single-city short-term PM_{2.5} exposure studies, and the likelihood that these results are influenced by localized events and not representative of air quality across the country, we place comparatively greater weight on the results from multi-city studies in considering alternative annual standard levels.

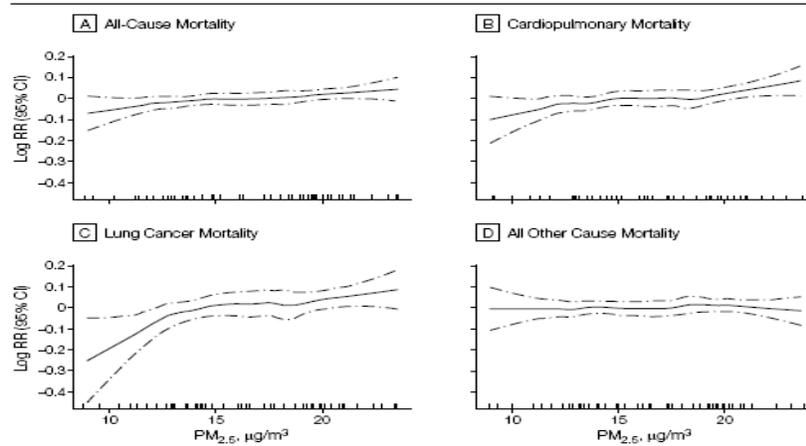
1 Although we have identified a number of PM₁₀ studies reporting confidence bounds
2 around C-R functions (e.g., Daniels et al., 2004; Samoli et al., 2005; Wong et al., 2008), we have
3 identified only three PM_{2.5} studies reporting confidence bounds around C-R functions (i.e., Pope
4 et al., 2002; Miller et al., 2007; Schwartz et al., 2008; see Figure 2-3). We found that these
5 studies analyzed the C-R function not to characterize the confidence bounds associated with the
6 C-R function, but primarily to determine if a linear curve most appropriately represented the C-R
7 relationship. In particular, one long-term exposure study utilized a nonparametric smoothing
8 model to depict the shape of the C-R function and associated confidence intervals using mortality
9 data from the ACS cohort (Pope et al., 2002). Goodness-of-fit testing indicated these data
10 supported a log-linear C-R relationship. We observe (and observed in the last review) that there
11 was an appreciable widening of confidence bounds on the smoothing plot for all-cause mortality
12 at approximately 13 to 12 µg/m³ (US EPA, 2005, p. 3-56; Figure 3-4) which is somewhat below
13 where the bulk of the air quality distribution in the study occurs (i.e., 14 µg/m³ is one standard
14 deviation below the long-term mean concentration of 17.7 µg/m³). The widening of the
15 confidence bounds in the smoothing plot is likely a consequence of the relative paucity of air
16 quality data at lower PM_{2.5} concentrations in this study, and does not suggest the possibility that
17 an effects threshold may exist at lower PM_{2.5} concentrations.

18 Additional evidence continues to support log-linear C-R relationships (US EPA, 2009a,
19 section 2.4.3). A recent long-term exposure study also provides information on the shape of the
20 C-R function and associated confidence intervals using cardiovascular mortality and morbidity
21 data from the WHI cohort (Miller et al., 2007). This analysis of cardiovascular events in relation
22 to long-term PM_{2.5} exposure is indicative of a continued reduction in risk at lower concentrations
23 of PM_{2.5}. However, since the reference value for the C-R function in this analysis is 11µg/m³, we
24 are unable to ascertain a point on the C-R relationship where there is an appreciable widening of
25 confidence bounds and greater uncertainty in the C-R relationship in this study.

26 Statistical analyses have been conducted to examine the PM-mortality C-R relationship
27 and whether a threshold exists. Schwartz et al. (2008) analyzed the shape of the concentration
28 response relationship associated with long-term PM_{2.5} exposure using a variety of statistical
29 methods. Similar to the above noted studies, the C-R function reported by Schwartz et al. (2008)
30 was found to be linear, with "...little evidence for a threshold in the association between
31 exposure to fine particles and the risk of death...". Although confidence bounds were provided
32 in Schwartz et al. (2008) for the association between PM_{2.5} and mortality, widening of
33 confidence bounds occurred just below the long-term mean concentration of 17.5 µg/m³, and
34 continued broadening of confidence bounds at lower PM_{2.5} concentrations is likely attributable to
35 the comparative lack of data at the lower end of the air quality distribution for this analysis (i.e.,

Figure 2-3. Confidence Bounds on PM_{2.5} Concentration-Response Relationships from Multi-city Epidemiological Studies

Figure 2. Nonparametric Smoothed Exposure Response Relationship



Vertical lines along x-axes indicate rug or frequency plot of mean fine particulate pollution; PM_{2.5}, mean fine particles measuring less than 2.5 µm in diameter; RR, relative risk; and CI, confidence interval.

Source: Pope et al., 2002

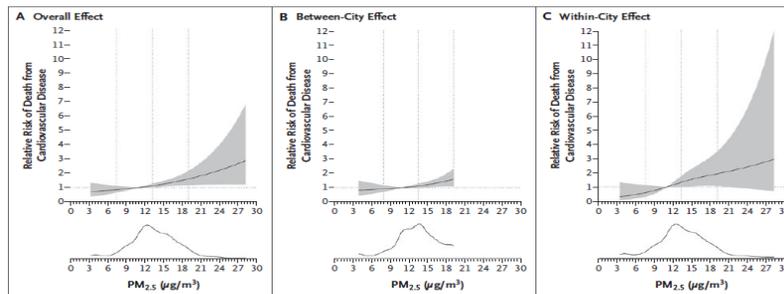


Figure 1. Level of Exposure to Fine Particulate Matter and the Risk of Death from Cardiovascular Causes in Women. The graphs demonstrate the observed relationship between the risk of death from cardiovascular disease and the level of particulate matter of less than 2.5 µm in aerodynamic diameter (PM_{2.5}), including both definite and possible deaths from coronary heart disease or cerebrovascular disease. Panel A shows the overall relationship between the PM_{2.5} level and death, Panel B the effects between metropolitan areas, and Panel C the effects within metropolitan areas, with an indicator variable used to adjust for each city. These results suggest a generally linear relationship between exposure and risk, though the 95% confidence intervals (shaded areas) are wide at the extremes of exposure. Risk is depicted in comparison with a reference value of 11 µg per cubic meter. The histogram in each panel illustrates the density of exposure distribution for air pollution. All estimates are adjusted for age, race or ethnic group, educational level, household income, smoking status, systolic blood pressure, body-mass index, and presence or absence of a history of diabetes, hypertension, or hypercholesterolemia.

Source: Miller et al., 2007

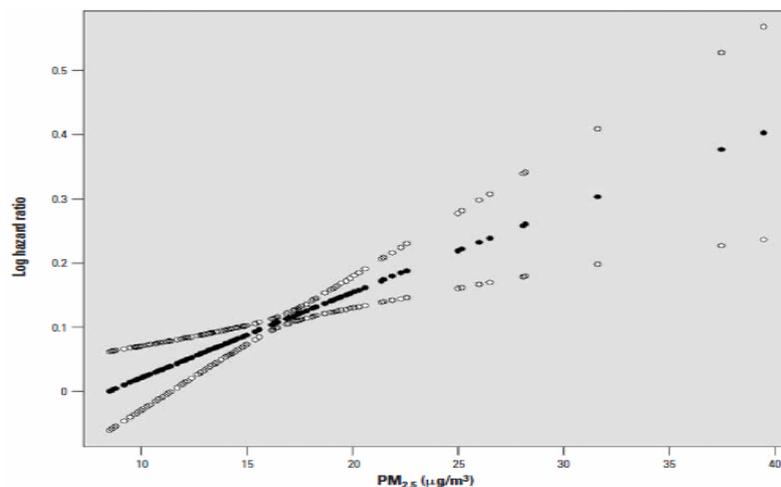


Figure 1. The estimated concentration-response relation between PM_{2.5} and the risk of death in the Six Cities Study, using a penalized spline with 18 knots. Also shown are the pointwise 95% CIs.

Source: Schwartz et al., 2008

1 10.7 $\mu\text{g}/\text{m}^3$ is one standard deviation below the long-term mean concentration). Therefore we
2 are unable to identify from this study where uncertainty in the C-R relationship substantially
3 increases at lower $\text{PM}_{2.5}$ concentrations and the potential increases for an effects threshold.

4 While these long-term studies provide information on the lack of any discernible
5 threshold and on the precision of $\text{PM}_{2.5}$ -associated health effects, we have found no new
6 evidence to inform our understanding of the confidence intervals around the estimated $\text{PM}_{2.5}$ C-R
7 functions. Consequently, without sufficient information available at this time we are unable to
8 use this type of approach as the basis for reaching conclusions on a range of alternative $\text{PM}_{2.5}$
9 standard levels that would be appropriate for consideration in this review.

10 In the absence of this type of evidence, we focus on the second approach by addressing
11 the following questions:

- 12 • **How do we consider different statistical air quality metrics for identifying alternative**
13 **levels to be considered for a generally controlling annual standard?**

14 We first recognize that health effects may occur over the full range of concentrations
15 observed in the long- and short-term epidemiological studies and that the ISA concluded no
16 discernible threshold for any effects can be identified based on the currently available evidence
17 (US EPA 2009a, section 2.4.3). As outlined in section 2.1.3, there is no single criterion that is
18 recognized as the “correct” approach to translate air quality metrics from epidemiological studies
19 into the basis for considering alternative standard levels, therefore, we explore various statistical
20 metrics that are reasonable to consider using two air quality distributions that are relevant to
21 inform alternative standard levels.

22 We first consider air quality distributions based on concentrations averaged across
23 ambient monitors within each area included in a given study (i.e., *composite monitor*) and then
24 averaged across study areas for an overall study mean concentration. The composite monitor
25 distribution is representative of the air quality data typically used in epidemiological analyses
26 and provides a direct link between $\text{PM}_{2.5}$ concentrations and the observed health effects.

27 We then consider a second air quality distribution based on concentrations estimated at
28 the monitor within each area that records the highest concentration (i.e., *maximum monitor*) and
29 then averaged across study areas for an overall study mean concentration. The maximum
30 monitor distribution is relevant because it reflects the distribution that is generally used to
31 determine whether a given standard is met in an area, and determines the extent to which
32 ambient $\text{PM}_{2.5}$ concentrations need to be reduced in order to bring an area into attainment with
33 the standard.

34 In identifying alternative standard levels that are appropriate to consider, we recognize
35 that the protection provided depends on whether these levels are set just below or substantially
36 below the long-term mean concentrations reported in the epidemiological studies and whether a

1 composite or maximum monitor distribution is used as the basis for selecting the levels.
2 Statistical metrics based upon composite monitor distributions are below, and in some places
3 well below, the same statistical metrics based upon maximum monitor distributions. A policy
4 approach that uses data based on composite monitor distributions to identify alternative standard
5 levels, and then compares those levels to concentrations at maximum monitors to determine if an
6 area meets a given standard, inherently builds in a margin of safety. In recognition of this
7 margin of safety, we believe it is appropriate to consider a policy approach that places
8 appreciably greater weight on focusing on alternative levels that are just somewhat below the
9 long-term mean concentrations reported in the epidemiological studies using the composite
10 monitor distributions.⁴⁷

11 Alternatively, a policy approach that uses maximum monitor distributions to identify
12 alternative standard levels that are then compared to concentrations at maximum monitors has no
13 inherent margin of safety. Therefore, to identify alternative standard levels using maximum
14 monitor distributions that provide a margin of safety, staff believes it is appropriate to consider a
15 policy approach that places greater weight on identifying alternative levels substantially below
16 the long-term mean concentrations from long- and short-term exposure studies. Using this
17 approach, we believe it is reasonable to focus on the lower end of the region within which the C-
18 R relationship is strongest, in which the bulk of the data reside. In doing so, we believe that
19 focusing on concentrations down to approximately one standard deviation below the long-term
20 mean concentrations using the maximum monitor distribution is also appropriate.

21 For the reasons discussed above, we focus our consideration of the long-term means and
22 associated standard deviations from multi-city epidemiological studies (see Figures 2-4 through
23 2-6). The studies included in these figures represent long- and short-term exposure studies that
24 evaluated endpoints classified in the ISA as having a causal or likely causal association at long-
25 term mean concentrations close to or below the level of the current annual PM_{2.5} standard (i.e.,
26 below 17 µg/m³).⁴⁸ In addition, Figure 2-5 includes long-term exposure studies of children that
27 evaluated effects classified as having evidence suggestive of an association below the level of the
28 current annual standard. These figures present long-term mean concentrations and associated
29 standard deviations to the extent they are available for the composite monitor and maximum
30 monitor distributions.

31 Beyond looking at standard deviations, we also have considered distributions from
32 epidemiological studies that provide a more continuous measure of the density of the data from

⁴⁷ A similar approach was followed in the SO₂ primary NAAQS review (75 FR 35547 to 35548, June 22, 2010).

⁴⁸ We note that additional studies presented and assessed in the ISA report effects at higher long-term mean PM_{2.5} concentrations.

Figure 2-4. Summary of Effect Estimates (per 10 µg/m³) and Air Quality Distributions for Long-term PM_{2.5} Exposure Multi-City Studies of the General Population and Older Adults

Study	Cite	Geographic Area	Years of Air Quality Data	Endpoint	Air Quality Data (µg/m ³)						Effect Estimate (95% CI)				
					Author Reported Data			EPA Analysis (Max Monitor) ^a							
					Mean	Mean - 1SD	Range	Mean	Mean - 1SD	Range					
<i>General Population</i>															
WHI	Miller et al. (2007)	36 US cities	2000	Mortality-CV	13.5	10.2	3.4 - 28.3	16.8	12.5	5-28					
				All CVD											
				Incident MI											
				Revascularization											
				Stroke											
				CBVD											
Cystic Fibrosis	Goss et al. (2004) ^b	6 US regions (NE, SE, NC, SC, NW, SW)	2000	Mortality-All-cause	13.7	9.5	11.8-15.9 (IQR)	-	-	-					
				Pulmonary exacerbation											
ACS-Reanalysis II	Krewski et al. (2009)	51 US MSAs	1999-2000	Mortality-all cause	14.0	11.0	5.8 - 22	15.4	11.3	9-25					
				Mortality-IHD											
				Mortality-CPD											
				Mortality-Lung cancer											
VA	Lipfert et al. (2006)		1999-2001	Mortality-all cause	14.3	11.3	5.0 - ?	-	-	-					
Harvard Six Cities (SCS)-Extended	Laden et al. (2006)	6 US cities (Northeast/Midwest)	1979-1998	Mortality-all cause	16.4 ^c	10.8	10-22	-	-	-					
				Mortality-CV											
				Mortality-Respiratory											
				Mortality-Lung cancer											
<i>Older Adults</i>															
MCAPS-Western US	Zeger et al. (2008)	62 US counties	2000-2005	Mortality-all cause	13.1 ^d	-	10.4-18.5 (IQR)	-	-	-					
Medicare-ACS	Eftim et al. (2008)	51 US MSAs	2000-2002	Mortality-all cause	13.6	10.8	6.0-25.1	14.9	11.3	9-24					
MCAPS-Eastern US	Zeger et al. (2008)	421 US counties	2000-2005	Mortality-all cause	14.0 ^d	-	12.3-15.3 (IQR)	-	-	-					
Medicare -SCS	Eftim et al. (2008)	6 US cities	2000-2002	Mortality-all cause	14.1	11.0	9.6-19.1	-	-	-					

08 1 1.2 1.4 1.6 1.8 2 2.2 2.4

^aMaximum monitor calculations noted in Hassett-Sipple et al. (2010)

^b cohort included persons with cystic fibrosis age 6 and older, mean age: 18.4 yrs

^cEstimated from data provided by study author

^dMedian (IQR: Interquartile range); overall US reported median (IQR) of 13.2 µg/m³ (11.1-14.9)

Figure 2-5. Summary of Effect Estimates (per 10 µg/m³) and Air Quality Distributions for Long-term PM_{2.5} Exposure Multi-City Studies of Children

Children								
Study	Cite	Geographic Area	Years of Air Quality Data	Endpoint	Air Quality Data (µg/m ³)			Effect Estimate (95% CI)
					Mean	Mean - 1SD	Range	
	Bell et al. (2007)	CT,MA	1998-2002	Low Birth Weight	11.9	10.3		
	Liu et al. (2007)	3 Canadian cities	1985-1999	IUGR - 1 st trimester	12.2	-	6.3-15	
IUGR - 2 nd trimester								
IUGR - 3 rd trimester								
	Parker and Woodruff (2008)	Continental US	2000-2003	Low Birth Weight	13.5	-	10.9-16.1 (IQR)	
S CA CHS	McConnell et al. (2003)	12 communities - S CA	1996-1999	Bronchitic Symptoms	13.8	6.1	6-29	
24-Cities	Dockery et al. (1996)	24 communities - US, Canada	1988-1991	Bronchitis	14.5	10.3	5.8-20.7	
	Woodruff et al. (2008)	96 US counties	1999-2002	Infant mortality	14.9 ^a	--	12.0-18.6 (IQR)	

^amedian for all cause mortality; median (IQR: interquartile range) for survivors = 14.8 (11.7-18.7) µg/m³

0.8 1 1.2 1.4 1.6 1.8 2

Figure 2-6. Summary of Effect Estimates (per 10 µg/m³) and Air Quality Distributions for Short-term PM_{2.5} Exposure Multi-City Studies of the General Population and Older Adults

Study/Cite	Geographic Area	Years of Air Quality Data	Endpoint	Air Quality Data (µg/m ³)						Effect Estimate (95% CI)	
				Author Reported Data			EPA Analysis (Max Monitor) ^a				
				Mean	Mean - 1SD	Range	Mean	Mean - 1SD	Range		
<i>General Population</i>											
Burnett et al. (2004)	12 Canadian Cities	1981-1999	Nonaccidental mortality	12.8	-	-	-	-	-	-	
Zanobetti & Schwartz (2009)	112 US counties	1999-2005	Nonaccidental mortality	13.2 ^b	10.3 ^b	6.6-24.7	14.1	10.9	6.7-26.4	-	
Burnett & Goldberg (2003)	8 Canadian Cities	1986-1996	Nonaccidental mortality	13.3	3.9 ^c	-	-	-	-	-	
Harvard Six Cities/ Klemm and Mason (2003)	6 US cities (Northeast/ Midwest)	1979-1988	Nonaccidental mortality	14.7 ^d	-	9-23 (IQR)	-	-	-	-	
Franklin et al. (2008)	25 US communities	2000-2005	Nonaccidental mortality	14.8 ^b	-	9.9-27.4 ^b	-	-	-	-	
Franklin et al. (2007)	27 US communities	1997-2002	Nonaccidental mortality	15.6 ^b	-	8.8-23.9	-	-	-	-	
<i>Older Adults</i>											
MCAPS/Bell et al. (2008)	202 US counties	1999-2005	CVD HA	12.9 ^b	10.2 ^b	4-20	13.4	10.3	5-26	-	
			Resp HA								
MCAPS/Dominici et al. 2006	204 US counties	1999-2002	IHD HA	13.4 ^b	10.5 ^b	4-23	13.9	10.5	5-29	-	
			CHF HA								
			Dysrhythmia HA								
			CBVD HA								
			PVD HA								
			COPD HA								
			RTI HA								

^aMaximum monitor calculations noted in Hassett-Sipple et al. (2010)

^bEstimated from data provided by study author or published study

^cEstimated from coefficient of variation reported in original study by Burnett et al. (2000)

^dMean value not reported in study, median presented from original study by Schwartz et al. (1996)

IQR: interquartile range

1 multi-city studies as a function of annual mean concentrations (from both composite and
2 maximum monitor distributions) across the cities in each study, as shown in Figures 2-7 and 2-8.
3 Specifically, we consider distributions of air quality data as well as distributions of population
4 data across air quality concentrations, which serve as a surrogate for the density of the health
5 effects data. Taken together, statistics from these distributions can inform our understanding of
6 the relative density of air quality and health effects data in the lower region of the C-R
7 relationship. We focus on the lower quartile of these distributions, and consider the range from
8 the 25th to 10th percentiles as a range within which the data become more sparse and that the C-R
9 relationship becomes appreciably more uncertain.⁴⁹ In so doing, we recognize that focusing on
10 any specific percentile from such distributions is somewhat arbitrary, although we believe the
11 25th and 10th percentiles are reasonable to consider. In staff's view, placing weight on PM_{2.5}
12 concentrations from air quality and population distributions below the range from the 25th to 10th
13 percentiles would be a highly uncertain basis for selecting alternative standard levels for
14 consideration.

15 We note the interrelatedness of these distributional statistics and a range of one standard
16 deviation around the mean, which contains approximately 68% of normally distributed data, in
17 that one standard deviation below the mean falls between the 25th and 10th percentiles of the air
18 quality distribution. Having considered both these distributional statistics and the concentrations
19 that are one standard deviation below study mean concentrations, we conclude that they provide
20 similar information to inform selection of alternative standard levels for consideration. In the
21 following discussion, we focus on the standard deviation statistic as being a more comparable
22 statistical measure across studies, rather than focusing on what are somewhat arbitrary
23 distributional percentiles, to inform our translation of the epidemiological evidence into the basis
24 for alternative standard levels that are appropriate to consider.

25 Composite Monitor Distributions

26 Information on composite monitor distributions is available for each epidemiological
27 study considered and, therefore, represents the most robust data set available to inform our
28 conclusions regarding alternative standard levels. In addition, these data were used in the

⁴⁹ In looking at the distributions between the 25th and 10th percentiles of either the air quality or population data, we note that the difference between the maximum and composite monitor distributions is much smaller for the three short-term exposure studies (Figure 2-8) than for the three long-term exposure studies (Figure 2-7). This difference is most likely associated with the greater number of years of air quality data considered in the short-term exposure studies (3 to 6 years) compared to the long-term exposure studies (1 to 3 years) as well as the greater number of study areas considered in the short-term exposure studies (112 to 204 counties) compared to the long-term exposure studies (36 to 51 cities/MSAs).

Figure 2-7. Air Quality and Population Distributions: Long-Term PM2.5 Exposure Studies

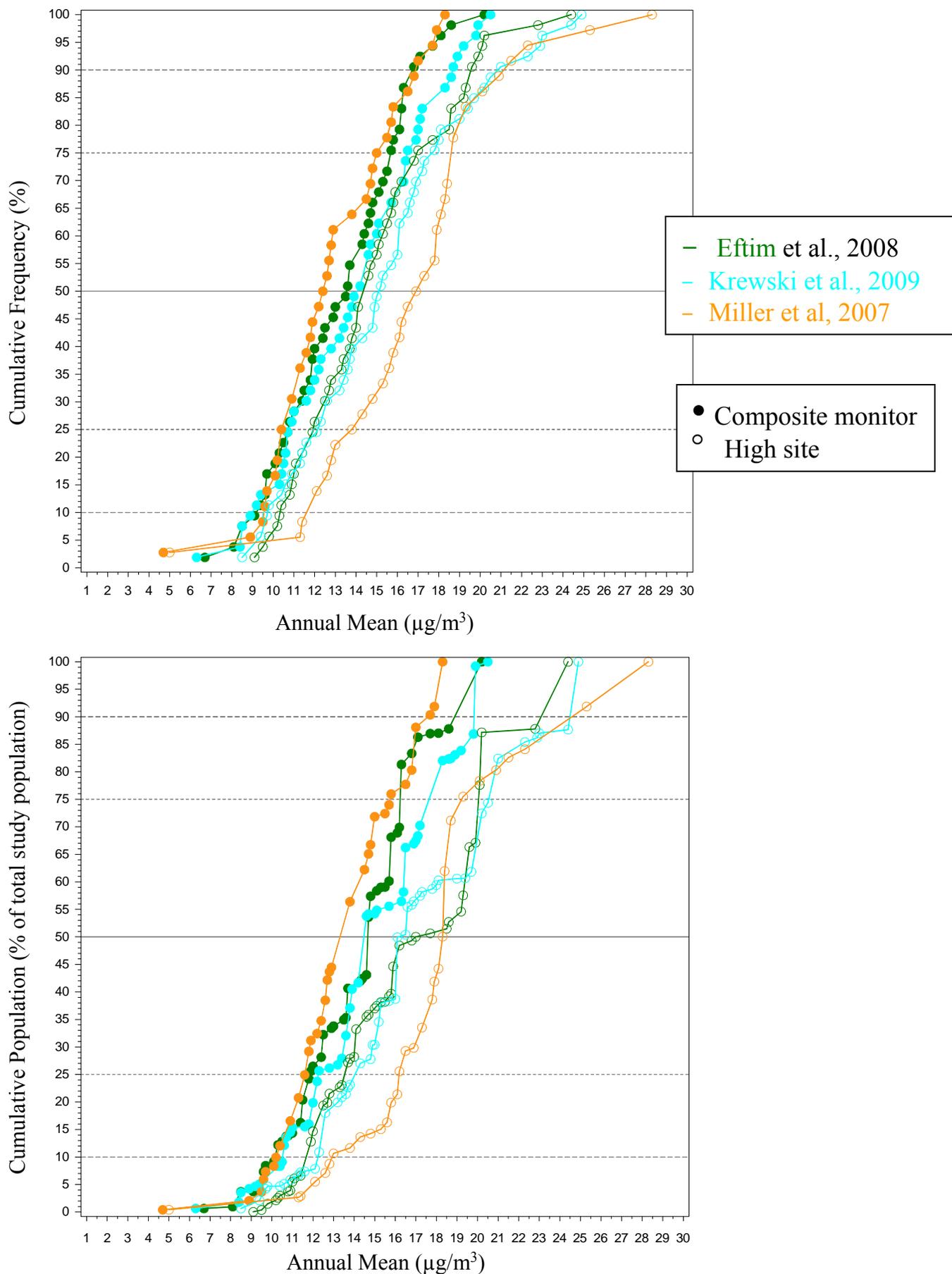
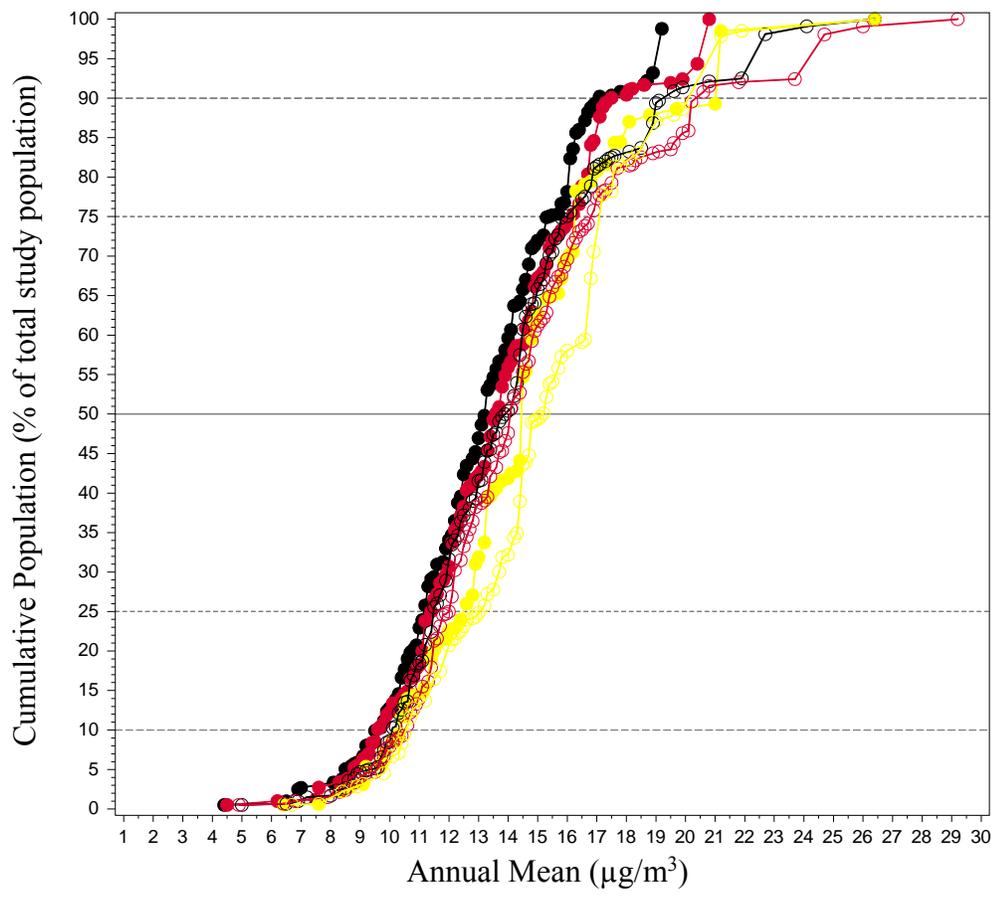
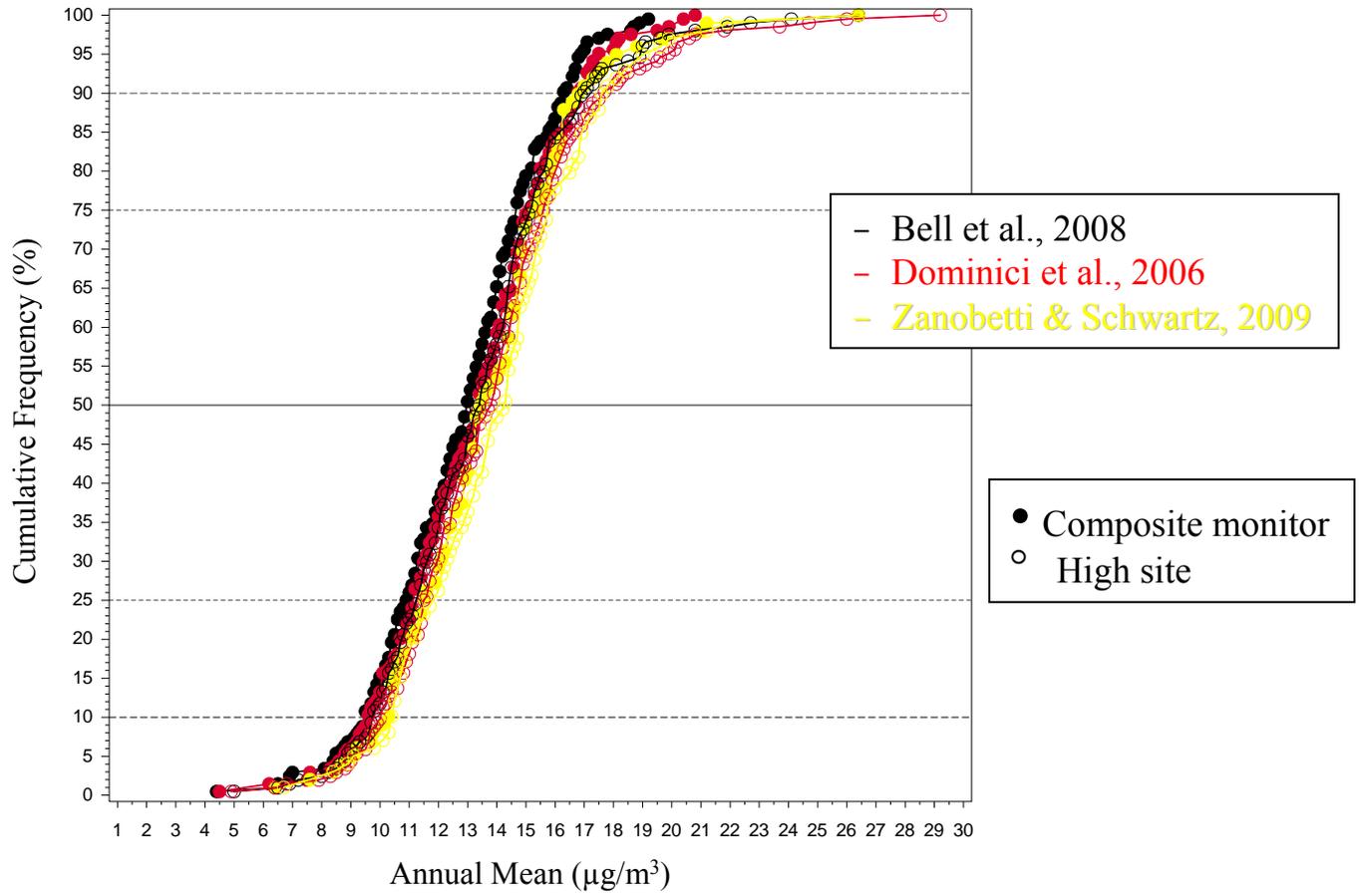


Figure 2-8. Air Quality and Population Distributions: Short-Term PM_{2.5} Exposure Studies



1 epidemiological studies to generate PM_{2.5} effect estimates and, therefore, provide a direct link to
2 health evidence.

3 By applying the policy approach discussed above, we focus on alternative levels that are
4 just somewhat below the long-term mean concentrations reported in the epidemiological studies
5 using the composite monitor distributions. We initially consider studies that evaluated mortality
6 (causal association), cardiovascular effects (causal association) or respiratory effects (likely
7 causal association). In the long-term exposure studies, we observe positive and statistically
8 significant associations at long-term mean concentrations ranging from 16.4 to 13.5 µg/m³
9 (Laden et al., 2006; Lipfert et al., 2006; Krewski et al., 2009; Goss et al., 2004; Miller et al.;
10 2007; Zeger et al., 2008; Eftim et al., 2008; Dockery et al., 1996; McConnell et al., 2003; see
11 Figures 2-4 and 2-5). In then considering information from short-term exposure studies
12 reporting positive and statistically significant associations with these same broad health effect
13 categories, we observe positive and statistically significant associations at long-term mean
14 concentrations in a somewhat lower range of 15.6 to 12.8 µg/m³ (Franklin et al., 2007, 2008;
15 Klemm and Mason, 2003; Burnett and Goldberg, 2003; Zanobetti and Schwartz, 2009; Burnett et
16 al., 2005; Bell et al., 2008; Dominici et al., 2006; See Figure 2-6).⁵⁰

17 In taking into consideration specific susceptible populations (i.e., children), we expand
18 our evaluation of the long-term exposure studies to include a broader range of health outcomes
19 judged in the ISA to have suggestive evidence of a causal association. This evidence is taken
20 into account to evaluate whether it provides support for considering lower alternative levels than
21 if weight were only placed on studies for which associations have been judged in the ISA to be
22 causal or likely causal. We make note of studies that provide emerging evidence for low birth
23 weight and infant mortality, especially related to respiratory causes during the post-neonatal
24 period (see Figure 2-5). This more limited body of evidence indicates positive and often
25 statistically significant effects associated with long-term mean concentrations in the range of
26 14.9 to 11.9 µg/m³ (Woodruff et al., 2008; Liu et al., 2009; Bell et al., 2007). As illustrated in
27 this figure, although Parker and Woodruff (2008) did not observe an association between
28 quarterly estimates of exposure to PM_{2.5} and low birth weight in a multi-city U.S. study, other
29 U.S. and Canadian studies did report positive and statistically significant associations between
30 PM_{2.5} and low birth weight at lower ambient concentrations (Bell et al., 2007; Liu et al., 2007).⁵¹

⁵⁰ When integrating evidence from short-term exposure studies reporting positive and statistically significant associations with mortality, cardiovascular, and respiratory effects, the ISA concluded that these associations are generally consistent and precise at long-term mean PM_{2.5} concentrations of 12.8 µg/m³ and above (US EPA, 2009a, pp. 2-10 to 2-11)

⁵¹ As noted in section 7.4 of the ISA. Parker et al. (2005) reported that over a 9 month exposure period (mean concentrations of 15.4 µg/m³) a significant decrease in birth weight was associated with infants in the highest quartile of PM_{2.5} exposure as compared to infants exposed in the lowest quartile.

1 There remain significant limitations (e.g., identifying the etiologically relevant time period) in
2 the evaluation of evidence on the relationship between PM_{2.5} exposures and birth outcomes (US
3 EPA, 2009a, pp. 7-48 and 7-56); nonetheless, we believe it is important to give some weight to
4 these studies in taking into account potential impacts on specific susceptible populations and in
5 considering alternative standard levels that provide protection with a margin of safety.

6 With respect to carcinogenicity, mutagenicity, and genotoxicity (suggestive evidence of a
7 causal association), the strongest evidence currently available is from long-term prospective
8 cohort studies that report positive relationships between PM_{2.5} and lung cancer mortality. At this
9 time, the PM_{2.5} concentrations reported in studies evaluating these effects reported ambient
10 concentrations that are equal to or greater than ambient concentrations observed in studies that
11 reported mortality and cardiovascular and respiratory effects (US EPA, 2009a, section 7.5).
12 Therefore, in selecting alternative levels, we note that in providing protection for mortality and
13 cardiovascular and respiratory effects, it is reasonable to anticipate that protection will also be
14 provided for carcinogenicity, mutagenicity, and genotoxicity effects.

15 A different policy approach might place greater weight on selecting alternative levels that
16 are substantially below the long-term mean concentrations reported in the epidemiological
17 studies based on composite monitor distributions. In focusing on effects classified as having a
18 causal or likely causal association with long- or short-term PM_{2.5} exposures, we observe
19 concentrations one standard deviation below the long-term means for long-term exposure studies
20 generally in the range of about 11 to 9.5 µg/m³ (Laden et al., 2006; Lipfert et al., 2006; Krewski
21 et al., 2009; Goss et al., 2004; Miller et al., 2007; Eftim et al., 2008; Dockery et al., 1996; see
22 Figures 2-4 and 2-5).⁵² As presented in Figure 2-6, one standard deviation below the long-term
23 mean concentrations for short-term exposure studies is generally about 10 µg/m³.⁵³ In expanding
24 our consideration to include effects classified as having evidence suggestive of a causal
25 association with long-term PM_{2.5} exposures, information is available for one study which
26 reported one standard deviation below the mean at 10.3 µg/m³ (Bell et al., 2007; see Figure 2-5).
27 We believe focusing on statistical metrics looking at levels that are substantially below the long-
28 term mean concentrations reported in the epidemiological studies using the composite monitor

⁵² With respect to considering air quality concentrations reported in the extended follow-up to the Southern California CHS, we recognize that one standard deviation below the long-term mean concentration aggregated across the 12 study communities is a significantly lower concentration than looking at one standard deviation below the long-term means reported in other long-term exposure studies, if fact, this concentration is close to or below the LML reported in the majority of the long-term exposure studies. This reflects the large standard deviation reported in this study related to the wide range of ambient concentrations considered in the 12 study communities. (McConnell et al., 2003).

⁵³ Burnett and Goldberg (2003) reported a comparatively greater variability in the air quality distribution across 8 Canadian cities: estimated SD from coefficient of variation reported in original study by Burnett et al.(2000) is 9.4 µg/m³.

1 distributions could potentially result in standard levels that are lower than necessary to provide
2 appropriate protection given the margin of safety already provided by the approach discussed
3 above.

4 Therefore, in focusing on composite monitor distributions, staff believes it is most
5 appropriate and reasonable to consider an alternative annual PM_{2.5} standard level just somewhat
6 below the lowest long-term mean concentrations reported in the multi-city epidemiological
7 studies, that is, in the range of 13 to 11 µg/m³.

- 8 • An alternative level of 13 µg/m³ would be somewhat below the lowest long-term
9 mean concentrations reported in the multi-city *long-term exposure studies* that
10 provide evidence of positive and statistically significant associations with health
11 effects classified as having a *causal or likely causal association*. This level would
12 also be below the long-term mean concentrations reported in many, but not all, *short-*
13 *term exposure studies* that provided similar evidence.
- 14 • An alternative level of 12 µg/m³ would be somewhat below the lowest long-term
15 mean concentrations reported in the multi-city *long- and short-term exposure studies*
16 that provide evidence of positive and statistically significant associations with health
17 effects classified as having a *causal or likely causal association*. This level would
18 also be below the long-term mean concentrations reported in two of the three long-
19 term exposure studies that provide *suggestive evidence* of positive and often
20 statistically significant associations with low birth weight and infant mortality.
- 21 • An alternative level of 11 µg/m³ would place appreciably greater weight on
22 information from multi-city, long-term exposure studies for which we have
23 *suggestive evidence* of low birth weight and infant mortality by considering a level
24 somewhat below the lowest long-term mean concentrations reported in such studies
25 that provide evidence of positive and statistically significant associations.

26 Maximum Monitor Distributions

27 With regard to maximum monitor distributions, we first note that this data set represents
28 a far less robust metric compared to the data set using composite monitor distributions because it
29 is based upon far fewer epidemiological studies. Furthermore, as previously noted, maximum
30 monitor distributions are not used in analyses of associations between air quality data and PM_{2.5}-
31 associated health effects in epidemiological studies.

32 Maximum monitor distributions were calculated by OAQPS for studies for which we
33 could reasonably match geographic study areas and air quality data from recent years assessed in
34 epidemiological studies that considered mortality (causal association), cardiovascular effects
35 (causal association) or respiratory effects (likely causal association) (Hassett-Sipple et al., 2010).
36 This data set includes consideration of three long-term exposure studies, specifically the
37 extended follow-up analyses of the ACS study (Krewski et al., 2009), the WHI study (Miller et
38 al., 2007), and the MCAPS cohort using the ACS study sites (Eftim et al., 2008) as well as three

1 short-term exposure studies including the MCAPS cohort (Dominici et al., 2006; Bell et al.,
2 2008), and a national mortality study (Zanobetti and Schwartz, 2009). Information for maximum
3 monitor distributions for studies that evaluated health outcomes judged in the ISA to have
4 suggestive evidence of a causal association were not calculated and, thus, are not considered in
5 the discussion below.

6 By applying the policy approach discussed above, in using the maximum monitor
7 distributions, we focus on alternative levels that reflect consideration of concentrations
8 substantially below the long-term mean concentrations reported in the epidemiological studies to
9 provide a margin of safety. In considering concentrations that are one standard deviation below
10 the long-term mean concentrations for the three long-term exposure studies, we observe
11 concentrations of 12.5, 11.3 and 11.3 $\mu\text{g}/\text{m}^3$ (Miller et al., 2007; Krewski et al., 2009; Eftim et
12 al., 2008; see Figure 2-4) from these studies that report positive and statistically significant
13 associations. In considering this same statistical metric for the three short-term exposure studies,
14 we observe somewhat lower concentrations of 10.9, 10.3, and 10.5 $\mu\text{g}/\text{m}^3$ (Zanobetti and
15 Schwartz, 2009; Bell et al., 2008; Dominici et al., 2006; see Figure 2-6).

16 A different policy approach might focus on alternative levels that are only just somewhat
17 below the long-term mean concentrations for epidemiological studies using the maximum
18 monitor distributions. For the three long-term exposure studies, positive and statistically
19 significant effects were associated with long-term mean concentrations of 16.8, 15.4, and 14.9
20 $\mu\text{g}/\text{m}^3$ (Miller et al., 2007; Krewski et al., 2009; Eftim et al., 2008; see Figure 2-3). For the
21 three short-term exposure studies, positive and primarily statistically significant effects were
22 reported at lower long-term mean concentrations of 14.1, 13.4, and 13.9 $\mu\text{g}/\text{m}^3$ ((Zanobetti and
23 Schwartz, 2009; Bell et al., 2008; Dominic et al., 2006; Figure 2-5). In staff's view, focusing on
24 statistical metrics looking at levels that are just below the long-term mean concentrations
25 reported in the epidemiological studies using the maximum monitor distributions would
26 potentially result in standard levels that are too high to provide protection with a margin of
27 safety.

28 Therefore, in focusing on maximum monitor distributions, we consider it most
29 appropriate to focus on a policy approach that considers alternative annual $\text{PM}_{2.5}$ standard levels
30 that are substantially below the long-term mean concentrations reported in the multi-city
31 epidemiological studies, specifically, at 11 or 10 $\mu\text{g}/\text{m}^3$.

- 32 • An alternative level of 11 $\mu\text{g}/\text{m}^3$ would be approximately *one standard deviation*
33 *below the long-term mean concentrations* reported in the multi-city *long-term*
34 *exposure studies* reporting positive and statistically significant associations with
35 health effects classified as having a *causal or likely causal association*.

- An alternative level of $10 \mu\text{g}/\text{m}^3$ would be approximately *one standard deviation below the long-term mean concentrations* reported in the multi-city *long- and short-term exposure studies* reporting positive and statistically significant associations with health effects classified as having a *causal or likely causal association*.

Summary of Evidence-based Considerations

In considering the currently available evidence, we have identified a set of alternative annual standard levels that we conclude are appropriate to consider. Based upon composite monitor distributions, we conclude it is appropriate to consider alternative annual standard levels from 13 to $11 \mu\text{g}/\text{m}^3$. Alternatively, based upon maximum monitor distributions, we conclude it is appropriate to consider alternative annual standard levels of 11 or $10 \mu\text{g}/\text{m}^3$. We also observe that these alternative standard levels are generally within the range of levels that would be identified based on considering distributional statistics, within the range of the 25th to 10th percentiles, of distributions of air quality data and population data across air quality concentrations, as discussed above.

While we believe alternative annual standard levels based on either composite monitor and maximum monitor distributions are appropriate to consider, we think it is reasonable to place more weight on an approach based on composite monitor distributions. As noted above, information on composite monitor distributions is available for each epidemiological study considered and, therefore, represents the most robust data set available to inform our conclusions regarding alternative standard levels. Further, this distribution represents the air quality data typically used in epidemiological analyses and provides a direct link between $\text{PM}_{2.5}$ concentrations and the observed health effects. In contrast, maximum monitor distributions represent a far less robust metric because they are available for only a few studies. Moreover, while this distribution is generally used to determine whether a given standard is met in an area, it is not used to link air quality data with $\text{PM}_{2.5}$ -associated health effects in the epidemiological evidence.

Therefore, in considering the emphasis to place on the two policy approaches considered, staff concludes it is most appropriate to place weight on the composite monitor approach and focus on alternative levels that are just somewhat below the lowest long-term mean concentrations reported in the multi-city studies. As noted above, by basing this level on the composite monitor distribution but by applying it at monitors that represent the maximum monitor distribution, this approach inherently builds in a margin of safety. Thus, based upon the currently available evidence, we conclude alternative annual standard levels in the range of 13 to $11 \mu\text{g}/\text{m}^3$ are most strongly supported by the available evidence and are appropriate to consider.

1 We next take into account evidence-based considerations and air quality data to inform
2 alternative standard levels for the 24-hour PM_{2.5} standard by addressing the following question:

- 3 • **What alternative standard levels are appropriate to consider for setting a 24-hour**
4 **standard intended to supplement the protection afforded by a generally controlling**
5 **annual standard?**

6 As recognized above, a generally controlling annual standard intended to serve as the
7 primary means for providing protection for effects associated with both long- and short-term
8 PM_{2.5} exposures cannot be expected to offer an adequate margin of safety against the effects of
9 all short-term PM_{2.5} exposures, especially in areas with high peak-to-mean ratios, possibly
10 associated with strong local or seasonal sources, or for potential PM_{2.5}-related effects that may be
11 associated with shorter-than-daily exposure periods. As a result, in conjunction with such an
12 annual standard, we believe it is appropriate to consider alternative 24-hour PM_{2.5} standard levels
13 that would provide supplemental protection while still maintaining the annual standard as the
14 generally controlling standard. Such a 24-hour standard can reasonably be based on considering
15 air quality information based on site-level 24-hour and annual design values. Such information is
16 presented in Figure 2-9 in terms of distributions of the ratios of 98th percentile design values to
17 annual mean design values by region across the U.S. A different way of looking at such
18 information is presented in Figure 2-10, which is a scatter plot of 24-hour and annual design
19 values in urban areas across the U.S., color-coded by region.

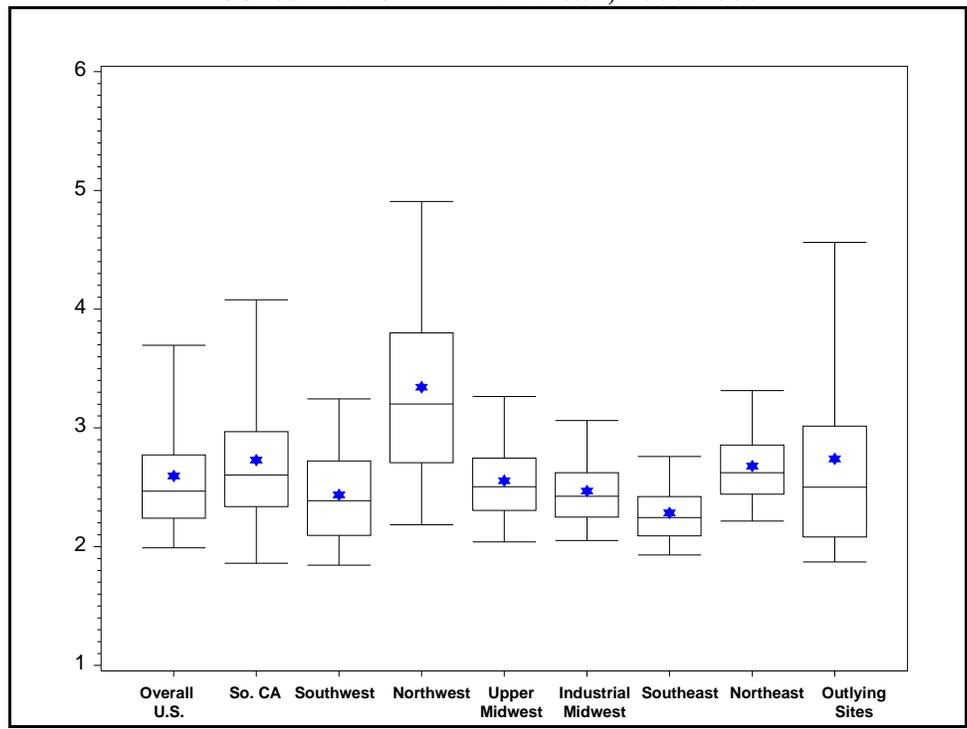
20 In general, as illustrated in Figure 2-9, 98th percentile PM_{2.5} concentrations are on average
21 approximately 2.5 times greater than annual concentrations in almost all regions of the U.S. with
22 the notable exception of the Northwest, where the ratio is approximately 3.5. In narrowing this
23 analysis to three recent years of air quality data (2006 to 2008), we see the same patterns
24 throughout the U.S. (Schmidt, 2010). Based on this information, we believe it is reasonable to
25 focus on 24-hour standard levels that are at least 2.5 times higher than the annual standard level.
26 Based on this consideration, we conclude that it is appropriate to consider retaining the current
27 24-hour standard level of 35 µg/m³ in conjunction with annual standard levels of 13 to 11 µg/m³.

28 We have also considered whether it is also appropriate to consider a lower 24-hour
29 standard level in conjunction with any annual standard levels within this range, while still
30 maintaining the annual standard as the generally controlling standard. As illustrated in Figure 2-
31 10, we have examined the implications of a 24-hour standard level of 30 µg/m³ as compared to a
32 level of 35 µg/m³ in conjunction with an annual standard of 12 µg/m³. As seen in Figure 2-10,
33 combining an annual standard of 12 µg/m³ with a 24-hour standard of 30 µg/m³ will result in
34 many more areas outside the Northwest in which the 24-hour standard would become the
35 controlling standard. This can be seen by looking to the left of the “12/30” and “12/35” lines,

1 which is the region on the graph in which the 24-hour standard would be controlling.⁵⁴ In
 2 contrast, based on the same type of consideration, we observe that a 24-hour standard level of 30
 3 $\mu\text{g}/\text{m}^3$ in conjunction with an annual standard of 11 $\mu\text{g}/\text{m}^3$ would not result in the 24-hour
 4 standard becoming the controlling standard in an appreciable number of areas. Based on these
 5 considerations, we conclude that it is appropriate to also consider a 24-hour standard level of 30
 6 $\mu\text{g}/\text{m}^3$ in conjunction with an annual standard of 11 $\mu\text{g}/\text{m}^3$.

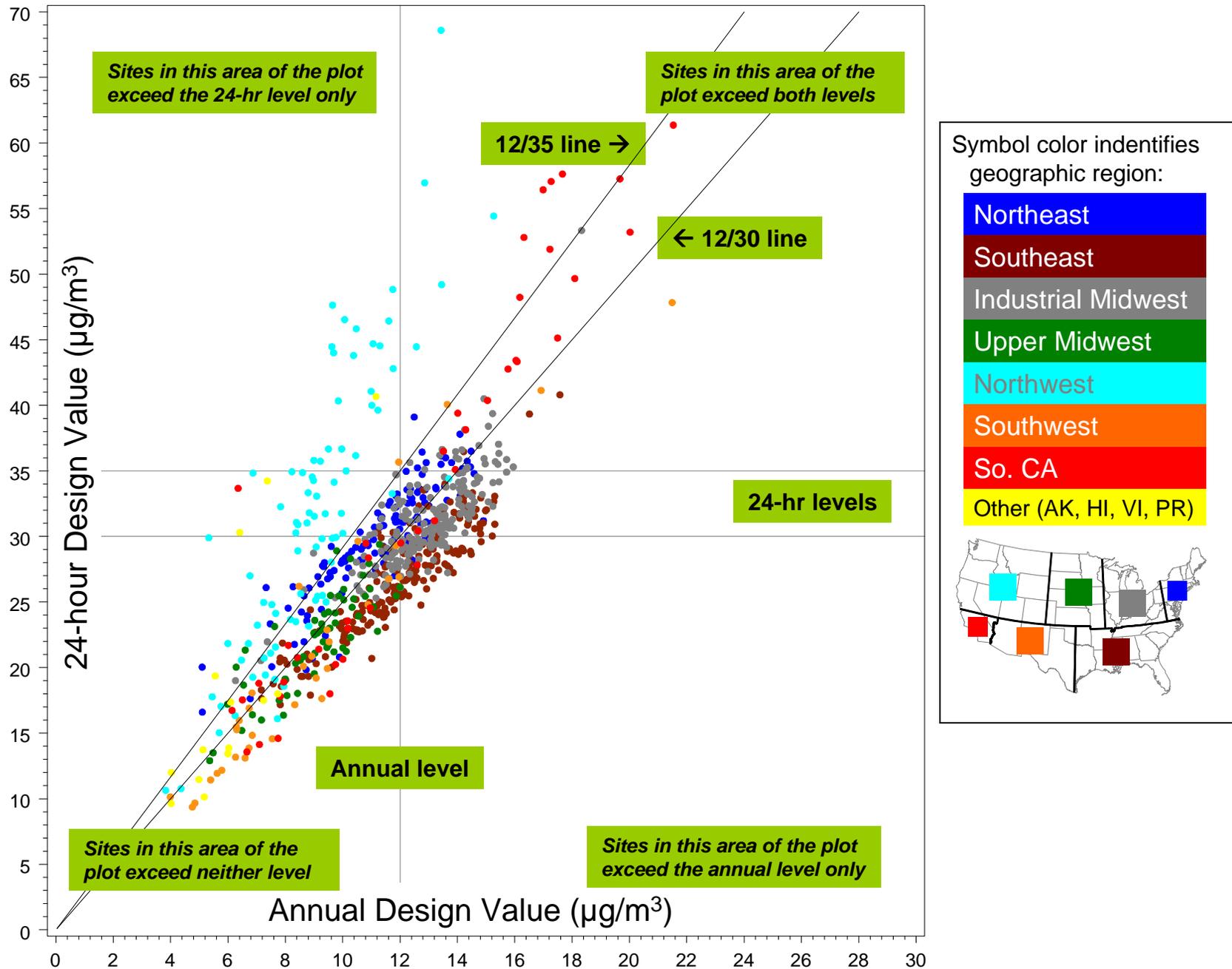
7 In summary, based on the above considerations, we conclude that it is appropriate to
 8 consider retaining the current 24-hour standard level of 35 $\mu\text{g}/\text{m}^3$ in conjunction with annual
 9 standard levels of 13 to 11 $\mu\text{g}/\text{m}^3$, and that consideration could also be given to an alternative 24-
 10 hour standard level of 30 $\mu\text{g}/\text{m}^3$ particularly in conjunction with an annual standard level of 11
 11 $\mu\text{g}/\text{m}^3$. Such combinations of 24-hour and annual standards levels would be expected to result in
 12 the annual standard being the generally controlling standard, except in areas in the Northwest
 13 with particular high peak-to-mean ratios, to protect against $\text{PM}_{2.5}$ -related health effects
 14 associated with long- and short-term exposures, with the 24-hour standard providing appropriate
 15 supplemental protection.

16 **Figure 2-9. Distribution of Site-Level Ratio of Annual 98th Percentile**
 17 **Concentration/Annual Mean, 2000-2008**



⁵⁴Figures similar to Figure 2-10 are presented in the RA (US EPA 2010a, section 4.5.1) together with a more detailed discussion of interpreting the information in such figures.

Figure 2-10. Site-level 24-hour DVs versus Annual DVs, 2006-2008



2.3.4.2 Risk-based Considerations

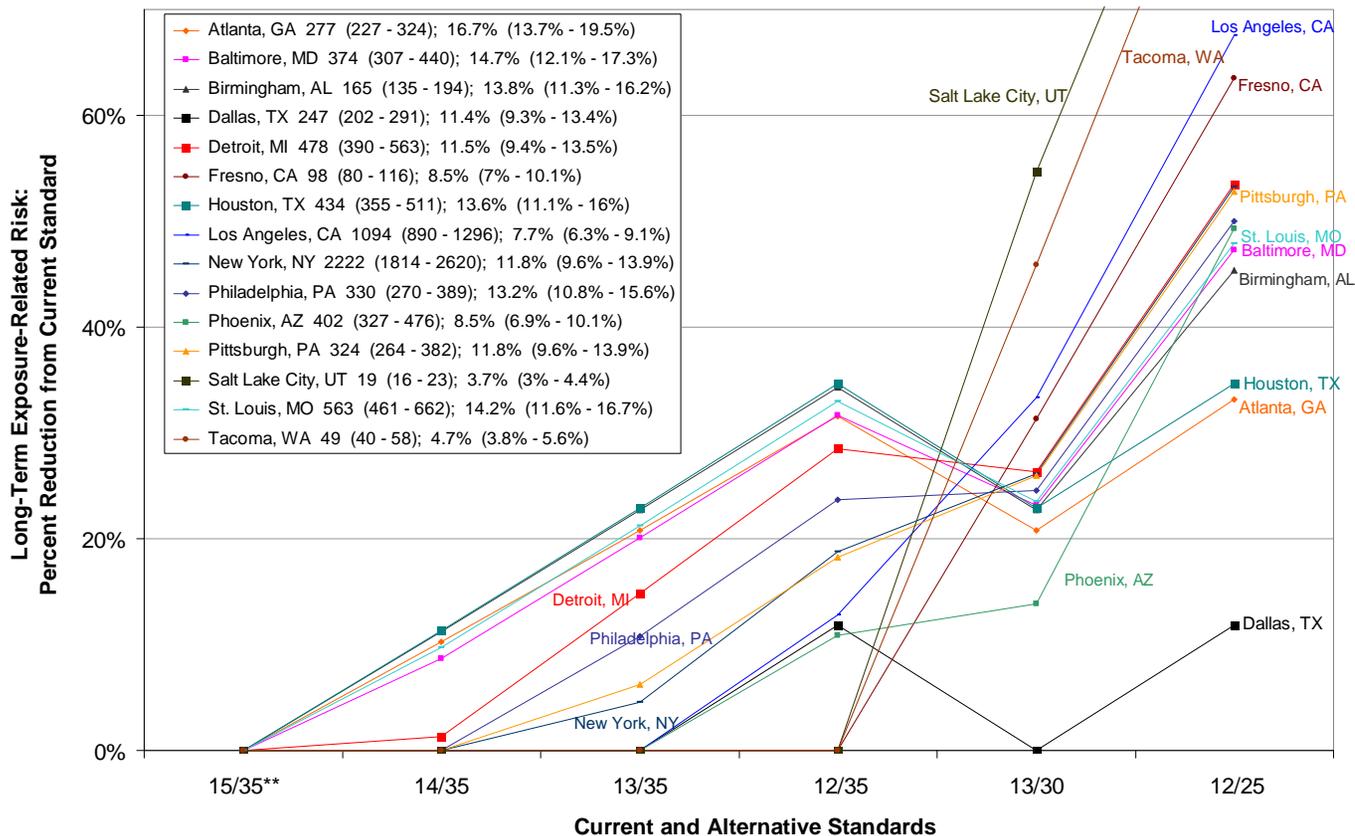
Beyond looking directly at the relevant epidemiologic evidence, staff has also considered the extent to which specific levels of alternative PM_{2.5} standards are likely to reduce both long-term exposure-related mortality risk and short-term exposure-related mortality and morbidity risk. In addition to considering the nature and magnitude of PM_{2.5}-attributable risk remaining under each set of alternative standards, we have also considered the nature and magnitude of risk reductions under the alternative standards considered. These risk estimates for the set of alternative standard levels considered are based on the same methodology used in estimating risk for the current suite of standard levels (see section 2.2.2).

The RA initially included analyses of alternative annual standard levels of 14, 13, and 12 µg/m³ paired with either the current 24-hour standard level of 35 µg/m³ or with alternative 24-hour standard levels of 30 and 25 µg/m³. The specific combinations of alternative standard levels assessed in the RA included: (a) suites focusing on alternative annual standard levels alone including combinations of alternative standard levels denoted by 14/35, 13/25 and 12/35 and (b) combinations of alternative annual and 24-hour standard levels including combinations denoted by 13/30 and 12/25. In addition, subsequent to the release of the second draft RA, we expanded the range of alternative annual standard levels evaluated to include a level of 10 µg/m³. In simulating ambient PM_{2.5} levels associated with these alternative standard levels, we included a more regional spatial pattern of reductions (reflected in the use of a proportional rollback approach) as well as more localized spatial patterns of reductions (reflected in the use of a hybrid approach and to an even greater extent in the use of a locally focused rollback approach) (see U.S. EPA, 2010a, section 3.2.3). While the proportional rollback approach was used in generating the core risk estimates, the other two more localized rollback approaches were considered as part of sensitivity analyses.

Results for the alternative suites of standard levels considered are presented in Figures 2-11 and 2-12, which depict patterns in risk reduction for long-term exposure-related risk (Figure 2-11) and short-term exposure-related risk (Figure 2-12) using different combinations of alternative standard levels relative to the risk under the current standard. These figures include results for each of the 15 urban study areas, thereby allowing patterns in risk reduction across alternative standard levels and urban study areas to be considered together.⁵⁵ The discussion

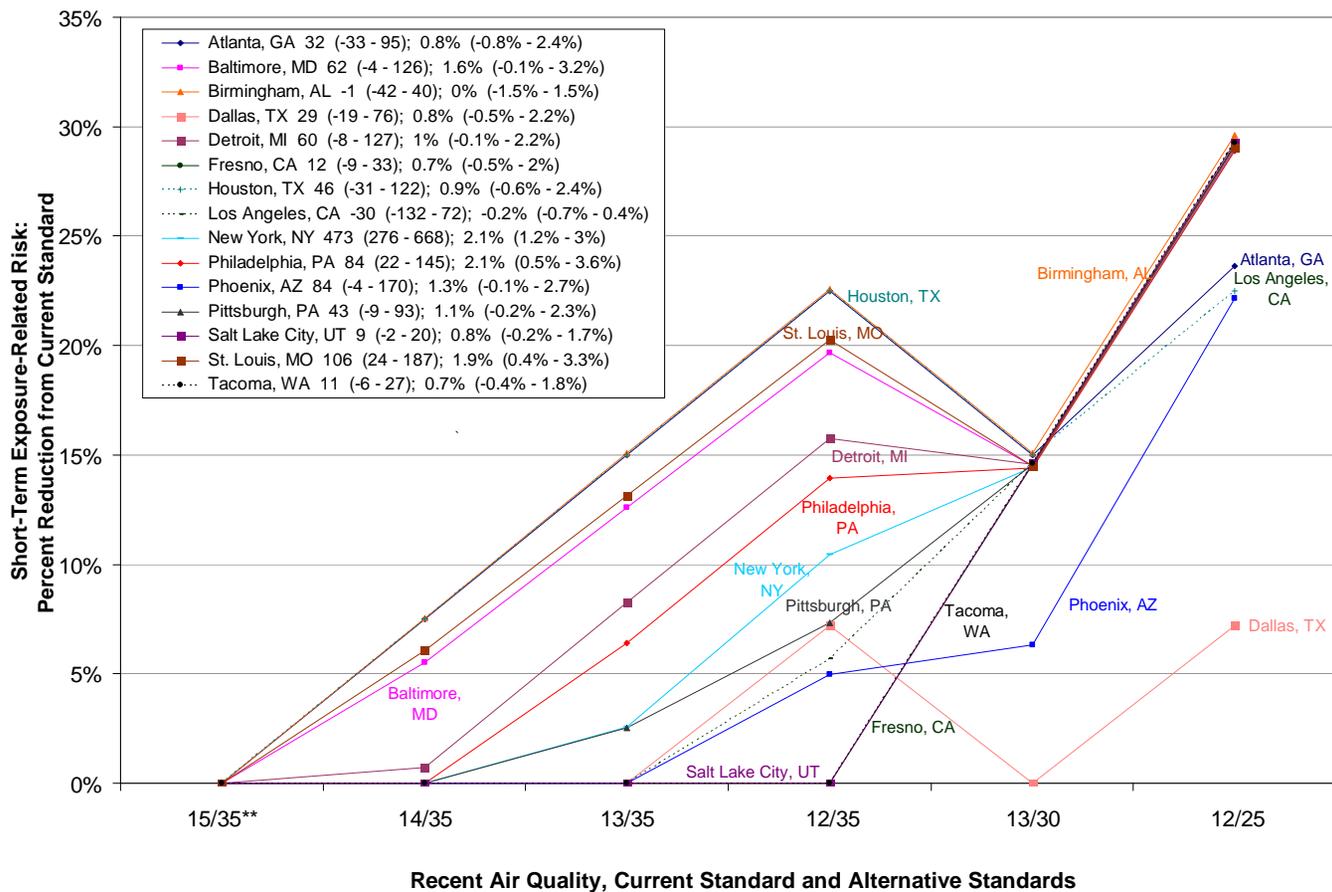
⁵⁵ Patterns of risk reduction across alternative annual standard levels (in terms of percent change relative to risk for the current annual standard level) are similar for all health endpoints modeled for a particular exposure duration (i.e., patterns of percent risk reduction will be similar for long-term exposure related all-cause, IHD and cardiopulmonary mortality). This reflects the fact that the C-R functions used in the quantitative RA are close to linear across the range of ambient air concentrations evaluated..

1 **Figure 2-11 Percent reduction in long-term exposure-related mortality risk** (alternative standards relative to the current standard)
 2 (Note: inset shows PM_{2.5} related incidence and percent of total incidence for IHD mortality under the current suite of standards*)
 3



4
 5
 6 *Based on Krewski et al. (2009), exposure period from 1999 – 2000. The legend contains, for each urban area, the incidence estimate (and 95% CI) and the
 7 estimate of percent of total incidence (and 95% CI) under the current standards. While incidence and percent of total incidence estimates are provided
 8 specifically for IHD-related mortality, the percent reduction plots provided in the figure apply to all long-term exposure-related mortality categories assessed –
 9 see text.
 10 **The current standards consist of an annual standard of 15 µg/m³ and a daily standard of 35 µg/m³. Combinations of an annual standard (n) and a daily standard
 11 (m) are denoted n/m in this figure. Note, that the percent reductions for Salt Lake City and Tacoma at the 12/25 standard are 100% and 93%, respectively.

1 **Figure 2-12. Percent Reduction in Short-term Exposure-related Mortality and Morbidity Risk** (alternative standards relative to the
 2 current standards) (Note: inset shows PM_{2.5} related incidence and percent of total incidence for CV mortality under the current suite of
 3 standards*)
 4



5
 6
 7 *Based on Zanobetti and Schwartz (2009). The legend contains, for each urban area, the incidence estimate (and 95% CI) and the estimate of percent of total
 8 incidence (and 95% CI) under the current standards. While incidence and percent of total incidence estimates are provided specifically for CV-related mortality,
 9 the percent reduction plots provided in the figure apply to all short-term exposure-related mortality and morbidity categories assessed – see text.
 10 **The current standards consist of an annual standard of 15 µg/m³ and a daily standard of 35 µg/m³. Combinations of an annual standard (n) and a 24-hour
 11 standard (m) are denoted n/m in this figure.

1 below of the magnitude of risk remaining under simulated attainment of the alternative standard
2 levels is based on risk estimates presented in U.S.EPA, 2010a, section 4.2.2.

3 • **What is the nature and magnitude of risk associated with just meeting the alternative
4 annual PM_{2.5} standards considered?**

5 In characterizing estimates of PM_{2.5}-related risk associated with simulation of the
6 alternative annual standards combined with the current 24-hour standard level (i.e., 14/35, 13/35
7 and 12/35), we estimated both the magnitude of risk reductions (relative to risk remaining upon
8 just meeting the current suite of standards), as well as the risk estimated to remain upon just
9 meeting the alternative standards. Our analysis included the assessment of risk associated with
10 an alternative annual standard level of 10 µg/m³. In discussing these risks, we focus on the set
11 of urban study areas experiencing risk reductions under each alternative annual standard. Key
12 policy-relevant observations associated with these risk estimates include:

- 13 • *Magnitude of estimated reductions in long-term exposure-related mortality risk:*
14 Upon simulation of just meeting the alternative annual standard levels considered (14,
15 13, and 12 µg/m³) in conjunction with the current 24-hour standard (denoted as 14/35,
16 13/35 and 12/35 suites of standards), the core analysis estimates reductions in long-
17 term exposure-related mortality for 12 of the 15 urban study areas, with the degree of
18 estimated risk reduction increasing incrementally across the alternative standard
19 levels (both in terms of the number of study areas experiencing risk reduction and the
20 magnitude of those reductions). For the alternative annual standard level of 12 µg/m³
21 (in conjunction with the current 24-hour standard), the core analysis estimates that
22 these study areas have reductions in risk (relative to risk remaining upon just meeting
23 the current suite of standards) ranging from about 11 to 35%.

24 For some of those areas in which the 24-hour standard is the generally controlling
25 standard, larger risk reductions would have been estimated in this case (12/35 suite of
26 standards) if the locally-focused rollback approach had been used to simulate just
27 meeting the current suite of standards. This result would be expected since the
28 magnitude of risk remaining upon just meeting the current suite of standards would
29 have been higher than that estimated based on the proportional rollback approach
30 used in the core analysis. Therefore, while the absolute risks would not change, the
31 percentage difference would have been greater if we had started with higher risks
32 related to simulation of just meeting the current annual standard.

- 33 • *Long-term exposure-related mortality risk remaining:* For an annual standard level of
34 14 µg/m³, the percent of total incidence of long-term exposure-related IHD mortality
35 attributable to PM_{2.5} (i.e., risk remaining) in the 5 urban study areas experiencing risk
36 reductions ranges from an estimate of 9 to 15%. For an alternative annual standard of
37 12 µg/m³, estimated risk remaining in the 12 urban study areas experiencing risk
38 reductions ranges from 6 to 11% in terms of PM_{2.5}-attributable long-term exposure-
39 related mortality. This translates into estimates of between 90 and 300 cases per year
40 attributable to long-term PM_{2.5} exposure for those study areas experiencing the

1 greatest reductions in risk under the lowest alternative annual standard level
2 simulated.

- 3 • *Short-term exposure-related mortality and morbidity risk:* For the alternative annual
4 standard level of 12 $\mu\text{g}/\text{m}^3$ (in conjunction with the current 24-hour standard), the
5 core analysis estimates that reductions in both short-term exposure-related CV
6 mortality and morbidity risk ranged from 5 to 23%.⁵⁶ In terms of risk remaining upon
7 simulation of 12 $\mu\text{g}/\text{m}^3$ (in conjunction with the current 24-hour standard), the urban
8 study areas with the greatest percent reduction have CV-related mortality estimates
9 ranging from 25 to 50 deaths per year.
- 10 • *Simulation of risks for an alternative annual standard level below 12 $\mu\text{g}/\text{m}^3$:*
11 Simulation of risks for an alternative annual standard of 10 $\mu\text{g}/\text{m}^3$ suggests that
12 additional risk reductions could be expected with alternative annual standards below
13 12 $\mu\text{g}/\text{m}^3$. However, we recognize that there is potentially greater uncertainty
14 associated with these risk estimates compared with estimates generated for the higher
15 alternative annual standards considered in the RA, since these estimates require
16 simulation of relatively greater reductions in ambient $\text{PM}_{2.5}$ concentrations. As lower
17 ambient $\text{PM}_{2.5}$ concentrations are simulated (i.e., ambient concentrations further from
18 recent conditions), potential variability in such factors as the spatial pattern of
19 ambient $\text{PM}_{2.5}$ reductions (rollback) increases, thereby introducing greater uncertainty
20 into the simulation of composite monitor annual mean $\text{PM}_{2.5}$ concentrations and,
21 consequently, risk estimates (US EPA, 2010a, Appendix J).
- 22 • *Substantial variability in magnitude of estimated risk reduction across urban study*
23 *areas:* While there is a consistent pattern of estimated risk reduction across the
24 alternative annual standards with lower alternative standard levels resulting in more
25 urban study areas experiencing increasingly larger risk reductions, there is
26 considerable variability in the magnitude of these reductions across study areas for a
27 given alternative annual standard level. This variability in estimated risk reflects
28 differing degrees of reduction in annual mean concentrations across the study areas,
29 which results, in part, because the study areas began with varying annual mean $\text{PM}_{2.5}$
30 concentrations after simulating just meeting the current suite of standards. Therefore,
31 even if study areas have similar “ending” annual mean $\text{PM}_{2.5}$ concentrations after
32 simulation of just meeting a specific alternative annual standard, because the starting
33 point in the calculation (the annual mean $\text{PM}_{2.5}$ concentrations upon just meeting the
34 current suite of standards) can be variable, the overall reduction in annual mean $\text{PM}_{2.5}$
35 concentrations across the standards can also be variable. This translates into variation
36 in reductions in long-term exposure-related risk upon just meeting alternative annual
37 standard levels across the study areas.
- 38 • *The nature of the spatial pattern in $\text{PM}_{2.5}$ reductions (reflected in the rollback*
39 *approach used) can impact the magnitude of estimated risk reductions:* The
40 sensitivity analysis involving application of locally focused rollback revealed that the
41 pattern of reductions in ambient $\text{PM}_{2.5}$ concentrations upon just meeting the current

⁵⁶ Because the same air quality metric (annual distributions of 24-hour $\text{PM}_{2.5}$ concentrations) was used in generating short-term exposure-related mortality and morbidity endpoints, patterns of risk reduction (as a percent of risk under the current suite of standards) are similar for both sets of endpoints (see US EPA, 2010a, section 4.2.2).

1 suite of standards can impact the magnitude of additional risk reductions estimated
2 for just meeting alternative (lower) annual standard levels. Specifically, for those
3 study areas with more peaky PM_{2.5} distributions, application of the locally focused
4 rollback approach resulted in higher annual mean PM_{2.5} concentrations remaining
5 upon just meeting the current suite of standards. If a proportional rollback approach
6 was then used to simulate just meeting alternative annual standard levels, a greater
7 degree of reduction in composite monitor annual mean PM_{2.5} concentrations will
8 result, since the “starting point” for the calculation (annual mean PM_{2.5} concentrations
9 upon just meeting the current suite of standards) would be higher. These findings
10 highlight the important roll played by variability in the spatial pattern of ambient
11 PM_{2.5} concentrations in influencing the magnitude of risk reductions under alternative
12 annual standard levels.

13 • *Based on consideration of the composite monitor annual mean PM_{2.5} concentrations*
14 *involved in estimating long-term exposure-related mortality, we have varying levels*
15 *of confidence in risk estimates generated for the three alternative annual standard*
16 *levels considered:* With the exception of one study area, those study areas estimated
17 to have risk reductions under the alternative annual standards of 14 and 13 µg/m³
18 have simulated composite monitor annual mean PM_{2.5} concentrations ranging from
19 just below 10.6 to over 13.3 µg/m³ (see US EPA, 2010a, Table 3-4). In other words,
20 these composite monitor annual mean PM_{2.5} concentrations generally fall well within
21 the range of ambient PM_{2.5} concentrations considered in fitting the C-R functions
22 used (i.e., within one SD of the mean PM_{2.5} concentration from 1999-2000 ACS
23 dataset). The urban study areas estimated to have risk reductions under the lower
24 alternative annual standard level of 12 µg/m³ have lower composite monitor annual
25 mean values ranging from 9.0 to over 11.4 µg/m³. These values generally extend to
26 below one SD of the mean of the ACS dataset and therefore, we have somewhat
27 lower confidence in these risk estimates, relative to those generated for the two higher
28 alternative annual standards. By contrast, urban study areas estimated to have risk
29 reductions under the alternative standard level of 10 µg/m³ (paired with the current
30 24-hour standard) have simulated composite monitor annual estimates ranging from
31 7.6 to 8.9 µg/m³ (see US EPA, 2010a, Table 3-4). These concentrations are towards
32 the lower end of the range of ACS data used in fitting the C-R functions (in some
33 cases approaching the LML) and, therefore, we have substantially less confidence in
34 these risk estimates, compared with those for the higher alternative annual standards
35 assessed.

36 • **What is the nature and magnitude of risk associated with simulating different**
37 **combinations of alternative annual and 24-hour PM_{2.5} standards?**

38 In characterizing PM_{2.5}-related risks associated with simulation of alternative annual
39 standards combined with alternative 24-hour standards (13/30 and 12/25), we estimated both the
40 magnitude of risk remaining upon just meeting these alternative standards, as well as the
41 magnitude of risk reductions (relative to risk remaining upon just meeting the current suite of
42 standards). While the alternative 24-hour standard levels considered did result in estimated risk

1 reductions, we have lower confidence in these risk estimates because of the relatively low
2 annual-average PM_{2.5} concentrations associated with simulation of these standard levels.

3 Of the 11 urban study areas estimated to have risk reductions under the alternative 24-
4 hour standard of 30 µg/m³ (with the 24-hour standard controlling – see US EPA 2010a, Table 3-
5 4), composite monitor annual mean PM_{2.5} concentrations ranged from 6.6 to 11.3 µg/m³ with
6 most of the urban study areas having concentrations in the 8 to 10 µg/m³ range. These
7 concentrations extend into the lower range of PM_{2.5} concentrations considered in the ACS study
8 to fit the C-R functions and therefore, we have somewhat lower confidence in these estimates.

9 When we consider composite monitor concentrations for urban study areas assessed to
10 have risk reductions under the alternative 24-hour standard level of 25 µg/m³ (again, where the
11 24-hour standard is controlling), we observed composite monitor annual mean PM_{2.5}
12 concentrations that are even lower, ranging from 5.6 to 11.2 µg/m³ with most study areas having
13 concentrations in the range of 7 to 9 µg/m³. Because this range extends well into the lower range
14 of PM_{2.5} concentrations considered in the ACS study to fit the C-R functions (in some cases
15 extending below the LML), we have substantially lower confidence in these risk estimates.
16 Furthermore, we find that those urban study areas with the greatest degree of estimated risk
17 reduction under these alternative 24-hour standard levels also had the lowest composite monitor
18 annual average PM_{2.5} levels, and therefore we have the lowest overall confidence in these results.

19 **2.3.4.3 Integration of Evidence-based and Risk-based Considerations**

20 In considering the epidemiological evidence, estimates of risk reductions associated with
21 just meeting alternative annual and/or 24-hour standards, air quality analyses, and related
22 limitations and uncertainties, staff concludes that there is clear support for considering revisions
23 to the suite of current PM_{2.5} standards to provide additional protection against health effects
24 associated with long- and short-term exposures. We recognize that health effects may occur over
25 the full range of concentrations observed in the long- and short-term epidemiological studies and
26 that no discernible threshold for any effects can be identified based on the currently available
27 evidence. In reaching staff conclusions regarding appropriate alternative standard levels to
28 consider, we have examined where the evidence of associations is strongest and, conversely,
29 where we have substantially less confidence in C-R relationships and in quantitative estimates of
30 risk.

31 As an initial matter, based on the available evidence and insights from the quantitative
32 risk assessment, we conclude that it is appropriate to consider alternative annual and 24-hour
33 standard levels consistent with the goal of establishing a suite of standards that includes a
34 “generally controlling” annual standard and a 24-hour standard that provides supplemental
35 protection, especially for areas with high peak-to-mean ratios, possibly associated with strong

1 local or seasonal sources, or for potential PM_{2.5}-related effects that may be associated with
2 shorter-than-daily exposure periods. Based upon the currently available evidence, we conclude
3 alternative annual standard levels in the range of 13 to 11 µg/m³ are appropriate to consider.
4 This conclusion reflects the much stronger body of scientific evidence available in this review
5 supporting a causal association between long- and short-term PM_{2.5} exposures and mortality and
6 cardiovascular effects and a likely causal association between long- and short-term PM_{2.5}
7 exposures and respiratory effects, as well as evidence that is suggestive of a causal association
8 with other health outcomes such as low birth weight and infant mortality and cancer,
9 mutagenicity, and genotoxicity effects. In conjunction with an annual standard within this range,
10 we conclude it is appropriate to consider retaining the current 24-hour standard level of 35
11 µg/m³, as well as considering an alternative 24-hour standard level of 30 µg/m³, particularly in
12 conjunction with consideration of an annual standard level of 11 µg/m³.

13 Beyond evidence-based considerations, we have also considered the extent to which the
14 quantitative risk assessment supports consideration of these alternative standard levels or
15 provides support for lower levels. We first conclude that risks estimated to remain upon
16 simulation of just meeting the current suite of standards are important from a public health
17 perspective, considering both the severity and estimated magnitude of effects. In considering
18 simulations of just meeting alternative annual standard levels within the range of 13 to 11 µg/m³
19 in conjunction with the current 24-hour standard level of 35 µg/m³, we conclude that important
20 public health improvements are associated with risk reductions estimated for standard levels of
21 13 and 12 µg/m³, noting that the level of 11 µg/m³ was not included in the quantitative risk
22 assessment. Our overall confidence in the quantitative risk estimates is strongest for the
23 alternative annual standard level of 13 µg/m³. We have somewhat lower confidence in risk
24 estimates for the alternative annual standard of 12 µg/m³. We also estimated risks likely to
25 remain upon just meeting an annual standard level of 10 µg/m³, although we have substantially
26 lower confidence in those estimates. We also have somewhat lower confidence in our risk
27 estimates for an alternative 24-hour standard level of 30 µg/m³, and substantially lower
28 confidence in our estimates of risks for an alternative 24-hour standard level of 25 µg/m³. Based
29 on the above considerations, we conclude that the quantitative risk assessment provides support
30 for considering an alternative annual standard within a range of 13 to 11 µg/m³, in conjunction
31 with a 24-hour standard of 35 or 30 µg/m³, but does not provide strong support for considering
32 lower alternative levels.

33 In identifying this range of annual standard levels, staff has taken into consideration the
34 importance of balancing the strength of the currently available evidence and risk-based
35 information with the remaining uncertainties and limitations associated with this information.
36 The upper end of this range of annual standard levels (13 µg/m³) reflects placing appreciably

1 more weight on the uncertainties and limitations in the information which would serve to reduce
2 the potential to overestimate public health risks and protection likely to be associated with just
3 meeting a standard set at this level. This policy option would reflect placing greater weight on
4 the remaining uncertainties in the evidence, including uncertainties associated with
5 understanding the heterogeneity observed in the epidemiological studies such as those associated
6 with the role of specific components, sources, and subfractions (e.g., UFPs) within the current
7 PM_{2.5} mass-based indicator, the role of fine particles and co-pollutants within the broader
8 ambient mixture, and exposure-related factors that influence the magnitude and duration of fine
9 particle exposures. The lower end of this range (11 µg/m³) reflects placing much less weight on
10 uncertainties and limitations in the information which would serve to reduce the potential to
11 underestimate public health improvements likely to be associated with just meeting a standard
12 set at this level. This policy option would reflect placing considerably more weight on limited
13 evidence of serious effects in susceptible populations such as potential developmental effects,
14 while recognizing that significant limitations remain in assessing the relationship between PM_{2.5}
15 exposures and these effects, specifically, understanding the nature of the association and
16 exposure windows of concern.

17 To provide some perspective on the implications of applying various combinations of
18 alternative annual and 24-hour standards, staff assessed (based on 2006 to 2008 air quality data)
19 the percentage of counties, and the population in those counties, that would not likely attain
20 various alternative suites of PM_{2.5} standards. This assessment, shown in Appendix 2A, Table 2A-
21 1, was not considered as a basis for the above staff conclusions.

22 **2.4 STAFF CONCLUSIONS ON ALTERNATIVE FINE PARTICLE STANDARDS**

23 In reaching conclusions on potential alternative standards to provide requisite protection
24 for health effects associated with long- and short-term fine particle exposures, staff has
25 considered these standards in terms of the basic elements of the NAAQS: indicator, averaging
26 time, form, and level. In considering the scientific and technical information, we reflect upon the
27 information available in the last review integrated with information that is newly available as
28 assessed and presented in the ISA and RA (US EPA, 2009a; US EPA, 2010a) and as summarized
29 in sections 2.2 and 2.3. We also consider the issues raised by the court in its remand of the
30 primary annual PM_{2.5} standard.

31 As outlined in section 2.1.3, our approach to reaching conclusions about the adequacy of
32 the current suite of PM_{2.5} standards and potential alternative standards that are appropriate for
33 consideration is broader and more integrative than approaches used in past reviews. Our
34 approach integrates a much expanded body of health effects evidence, more extensive air quality
35 data and analyses, and a more comprehensive quantitative risk assessment, and considers the

1 combined protection against PM_{2.5}-related mortality and morbidity effects associated with both
2 long- and short-term exposures afforded by the annual and 24-hour standards.

3 We recognize that selecting from among alternative standards will necessarily reflect
4 consideration of the qualitative and quantitative uncertainties inherent in the relevant evidence
5 and in the assumptions that underlie the quantitative risk assessment. In identifying these
6 alternative suites of primary standards and ranges of levels for consideration, we are mindful that
7 the Clean Air Act (CAA) requires standards to be set that are requisite to protect public health
8 with an adequate margin of safety, such that the standards are to be neither more nor less
9 stringent than necessary. Thus, the CAA does not require that the NAAQS be set at zero-risk
10 levels, but rather at levels that avoid unacceptable risks to public health.

- 11 (1) Consideration should be given to revising the current PM_{2.5} primary standards to provide
12 increased public health protection from the effects of both long- and short-term exposures to
13 fine particles in the ambient air. This conclusion is based in general on the evaluation in the
14 ISA of the currently available epidemiological, toxicologic, dosimetric, and exposure-related
15 evidence, and on air quality information and analyses related to the epidemiological
16 evidence, together with judgments as to the public health significance of the estimated
17 incidence of effects upon just meeting the current suite of standards.
- 18 (2) The indicator for fine particle standards should continue to be PM_{2.5}. We conclude that the
19 available evidence does not provide a sufficient basis for replacing or supplementing the
20 PM_{2.5} indicator with an indicator defined in terms of UFPs or for any specific fine particle
21 component or source category of fine particles, nor does it provide a basis for excluding any
22 component or source category from the mix of particles included in the PM_{2.5} indicator.
- 23 (3) Averaging times for PM_{2.5} standards should continue to include annual and 24-hour averages
24 to protect against health effects associated with long-term (seasons to years) and short-term
25 (hours to days) exposure periods. Consideration of other averaging times, including an
26 averaging time less than 24 hours to address health effects associated with subdaily
27 exposures or a longer averaging time to address effects associated with seasonal exposures,
28 was limited by the relatively small amount of relevant information available.
- 29 (4) Consideration should be given to revising the form of the annual standard to one based on the
30 highest community-oriented monitor in an area rather than a form that allows averaging
31 across monitors (e.g., spatial averaging). The 98th percentile form of the current 24-hour
32 standard should be retained.
- 33 (5) Consideration should be given to alternative suites of PM_{2.5} standards to provide protection
34 against effects associated with both long- and short-term exposures, taking into account both
35 evidence-based and risk-based considerations.
 - 36 (a) Consideration should be given to setting the levels of the annual and 24-hour
37 standards such that the annual standard would be the “generally controlling”
38 standard to provide protection for both long- and short-term PM_{2.5} exposures in
39 conjunction with a 24-hour standard to provide supplemental protection against
40 days with high peak concentrations to limit peak concentrations in areas with high
41 peak-to-mean ratios, possibly associated with strong local or seasonal sources, or

1 for potential PM_{2.5}-related effects that may be associated with shorter-than-daily
2 exposure periods. We conclude that this is the most effective and efficient way to
3 reduce total population risk associated with both long- and short-term exposures
4 and provide requisite protection in areas across the country.

5 (b) Consideration should be given to alternative annual standard levels in the range of
6 13 to 11 µg/m³. We conclude that this range is most strongly supported by the
7 currently available scientific evidence and risk-based information

8 (c) Consideration should be given to retaining the current 24-hour standard level of
9 35 µg/m³ in conjunction with annual standard levels in this range; consideration
10 could also be given to an alternative 24-hour standard level of 30 µg/m³
11 particularly in conjunction with an annual standard level of 11 µg/m³.

12 **2.5 KEY UNCERTAINTIES AND AREAS FOR FUTURE RESEARCH AND DATA** 13 **COLLECTION**

14 In this section, we briefly discuss key uncertainties and areas for future health-related
15 research, model development, and data collection activities. These efforts, if undertaken, could
16 provide important evidence for informing future PM NAAQS reviews and, in particular,
17 consideration of possible alternative indicators, averaging times, and/or levels. In some cases,
18 research in these areas can go beyond aiding standard setting to informing the development of
19 more efficient and effective control strategies. We note, however, that a full set of research
20 recommendations to meet standards implementation and strategy development needs is beyond
21 the scope of this discussion.

22 As has been presented and discussed in the PM ISA, particularly in Chapters 4 through 7,
23 the scientific body of evidence informing our understanding of health effects associated with
24 long- and short-term exposures to fine particles has been broadened and strengthened since the
25 last review. In reviewing the adequacy of the current suite of primary PM_{2.5} standards and in
26 evaluating alternative health-based fine particle standards appropriate for consideration, we
27 identify the following key uncertainties and areas for future research and data collection efforts
28 that have been highlighted in this review. We recognize that some research could be available to
29 inform the next PM NAAQS review, while other research may require longer-term efforts.

30 Interpretation of Epidemiological Evidence

31 Additional research focused on identifying the most important factors contributing to the
32 observed heterogeneity in the epidemiological evidence could provide insights for interpreting
33 these studies. We encourage research and data collection efforts directed at improving our
34 understanding of the nature of the exposures contributing to the observed health effects, for
35 example, the role of specific components, sources, and subfractions (e.g., UFPs) within the
36 current PM_{2.5} mass-based indicator and the role of fine particles and co-pollutants within the
37 broader ambient mixture, as well as improving our understanding of exposure-related factors that

1 influence the magnitude and duration of fine particle exposures. Much of this research may
2 depend on the availability of increased monitoring data, as discussed below.

- 3 • Components/Sources. The currently available scientific evidence continues to be largely
4 indexed by aggregate PM_{2.5} mass-based concentrations which vary in composition both
5 regionally and seasonally. Source characterization, exposure, epidemiological, and
6 toxicological research focused on improving our understanding of the relative toxicity of
7 different fine particle components, properties, and sources that may be more closely linked
8 with various health effects. Critical to this better understanding of the impacts of
9 components and their associated sources are data that refines the temporal and spatial
10 variability of the fine particle mixture.. This research would reduce the uncertainties in
11 estimating risks. It could also inform consideration of alternative indicators in future PM
12 NAAQS reviews as well as aid in the development of efficient and effective source control
13 strategies for reducing health risks.
- 14 • Ultrafine Particles (UFPs). Additional monitoring methods development work, health
15 research, and ambient monitoring data collection efforts are needed to expand the currently
16 available scientific data base for UFPs. UFP measurements should include surface area as
17 well as number, mass and composition. It would be most useful for a UFPs monitoring
18 network to be designed to inform our understanding of the spatial and temporal variability of
19 these particles, including in near-roadway environments. This information would improve
20 our ability to explore consideration of a separate indicator for UFPs as a subfraction of fine
21 particles in future PM NAAQS reviews.
- 22 • Co-pollutant Exposures. Research focused on furthering our understanding of the extent to
23 which an association between fine particles and specific health effects can be modified by
24 one or more co-pollutants would inform our ability to discern the role of PM in the complex
25 ambient mixture. For example, does the magnitude of a PM_{2.5}-related effect estimate differ
26 on days when O₃ levels are higher compared to days when O₃ levels are lower?
- 27 • Exposure-related Factors. Additional research and analyses would be useful to provide
28 insights on population exposures, specifically in improving our understanding of intra-city
29 and inter-city differences related to various PM components, source contributions and
30 personal and building-related factors (e.g., air conditioning use; residence near roadways)
31 that may enhance our interpretation of the epidemiological evidence. This research could
32 focus on different size fractions in PM_{2.5} (i.e., UFPs) as well as components. Coordination
33 between exposure and health studies could advance our understanding of exposure-related
34 factors. For example, epidemiological panel studies might use measured personal exposure
35 measurements to explore differences in personal exposures related to (1) indoor generated
36 fine particles, (2) fine particle exposures measured by community monitors, and (3) fine
37 particle exposures not captured by community monitors (i.e., personal exposures during
38 commuting).

39 Health Outcomes, Exposure Durations of Concern, and Susceptible Populations

40 New information available in this review reinforces and expands the evidence of
41 associations between long- and short-term PM_{2.5} exposures and mortality and cardiovascular and
42 respiratory effects. Less evidence is available to understand other health effects (e.g.,

1 developmental/reproductive effects; central nervous system effects). Additional research could
2 expand our understanding of the associations between PM_{2.5} and a broader range of health
3 outcomes; reduce uncertainties associated with our current understanding of concentration-
4 response relationships; improve our understanding of exposure durations of concern; and
5 improve our understanding of the potential public health impacts of fine particle exposures in
6 susceptible populations.

- 7 • Health Effects. Research on a broader range of cardiovascular and respiratory endpoints
8 could improve our understanding of the mechanisms by which these effects occur. In
9 addition, future research could expand the scientific data base for health effects that are
10 currently less understood including effects categorized within the ISA as having evidence
11 suggestive of a causal relationship or for which currently available evidence is inadequate.
12 To the extent that research supports a link between fine particles and adverse effects on the
13 nervous system, reproduction, development, or other endpoints, such effects could play an
14 increased role for informing future PM NAAQS reviews including expanding the health
15 endpoints that could potentially be evaluated in future quantitative risk assessments.
- 16 • Concentration-Response Relationships. Research focused on improving our understanding
17 of the shape of the C-R relationships, especially at lower ambient fine particle
18 concentrations, as well as the confidence bounds around these C-R relationships, could
19 reduce uncertainties associated with estimating and characterizing risks throughout the full
20 range of air quality distributions. As more information becomes available on fine particle
21 components and sources, it will be important to understand the C-R relationships for key
22 constituents of the fine particle mixture, as well.
- 23 • Exposure Durations of Concern. Research should be directed at broadening the scientific
24 data base to improve our understanding of health effects associated with short-term, peak
25 exposures, such as those related to traffic-related sources, wildfires, agricultural burning, or
26 other episodic events, as well as to improve our understanding of health effects associated
27 with seasonal-length exposures, such as those related to wintertime wood-burning emissions.
28 Additional quantitative measures of exposure might take into account factors including the
29 magnitude and duration of sub-daily and seasonal length PM_{2.5} exposures and the frequency
30 of health impacts associated with repeated peak exposures. More research is needed to better
31 understand effects that occur at longer lag times than have historically been studied (e.g., 0 to
32 2 day lags).
- 33 • Susceptible Populations. Improving our understanding of the populations that are more
34 likely to experience adverse health effects related to fine particle exposures and the
35 concentrations at which these effects may occur is important for informing future PM
36 NAAQS reviews and for developing programs to reduce related public health risks. This
37 evidence may also provide insights into the biologic modes of action for toxicity.
 - 38 ○ Pre-existing Health Conditions. While currently identified susceptible populations
39 include persons with pre-existing cardiovascular and respiratory disease, evidence
40 continues to emerge related to additional health conditions that may increase
41 susceptibility to fine particle exposures (e.g., diabetes, obesity, neurological
42 disorders). Research to replicate or extend these findings would enhance our
43 understanding of these and other potentially susceptible populations.

- 1 ○ Children. Epidemiological and toxicological studies provide evidence that children
2 are more susceptible to PM exposures, primarily for respiratory-related effects.
3 Evidence of developmental effects associated with PM exposures continues to
4 emerge. Additional research exploring issues to better understand key windows of
5 development impacted by PM exposures could enhance our understanding of this
6 important susceptible population.
- 7 ○ Genetic Susceptibility. Research to expand our understanding of genetic
8 susceptibility could inform our understanding of potentially susceptible populations
9 and provide additional information for identifying the specific pathways and
10 mechanisms of action by which PM initiates health effects.
- 11 ○ Socioeconomic status (SES). Additional research is needed to identify what factors
12 (e.g., general health status, diet, medication, stress, unmeasured pollution) cause SES
13 differences in response to pollution measured at a community site.

14 Data Collection Needs and Methods Development Activities

15 Additional research and data collection efforts focused on expanding current monitoring
16 methods and networks as well as continued development of exposure models expand data
17 available for health studies could improve our understanding of potential alternative indicators,
18 averaging times, and levels to consider in future PM NAAQS reviews. In particular, staff
19 encourages work to enhance our understanding of the temporal and spatial variability of PM_{2.5},
20 PM_{2.5} components, and subfractions (e.g., UFPs).

- 21 • Monitoring Measurements. In order to improve our understanding of the association
22 between fine particles and health effects, more frequent measurement data could be collected.
23 This would provide information that could inform our understanding of alternative lags.
 - 24 ○ PM_{2.5} Components. With respect to improving our understanding of the impacts of
25 PM_{2.5} components, enhancements to the Chemical Speciation Network (CSN),
26 including more frequent measurement schedules and the development and
27 deployment of continuous monitoring methods for specific fine particle components
28 (e.g., EC/OC, sulfates), could enhance our understanding of the temporal and spatial
29 variability of specific components. Furthermore, identifying chemical species within
30 the mix of OC would improve our understanding of the artifacts associated with semi-
31 volatile PM components and aid in designing toxicological experiments.
 - 32 ○ Ultrafine Particles. In order to improve our understanding of the public health
33 impacts of UFPs, consideration should be given to establishing a Federal Reference
34 Method (FRM) for UFPs and establishing a national UFP monitoring network.
 - 35 ○ Source Apportionment. Composition data with better time resolution (e.g., 1 to 6
36 hour) and better size resolution (e.g., UFPs, accumulation mode particles, coarse
37 particles in PM_{2.5} and PM_{10-2.5}) could provide more precise and accurate information
38 on sources of fine particles to inform health research as well as development of more
39 efficient and effective control strategies.
 - 40 ○ Areas of Peak Exposures. Some portion of the required PM_{2.5} monitoring network
41 could be dedicated to improving our ability to characterize community-wide air

1 quality at a neighborhood or urban scale that represents where we expect to find peak
2 exposures. A portion of the required network could be targeted to monitor short-term
3 PM_{2.5} concentrations in areas that are not affirmatively designated as “community-
4 wide” air quality. Requiring monitors in areas not designated as “community-wide”
5 would ensure protection for those populations whose ambient exposures may be
6 affected by unique local sources or terrain which are often located in areas with
7 smaller spatial scales. These areas, such as population-oriented micro- and middle-
8 scale “hot spots,” are not required to be monitored in the current network design.

- 9 • Model Development. Continuing work to improve models for estimating PM_{2.5} mass and
10 composition in areas with only every third or sixth day measurements, and by space, where
11 measurements are not available could enhance our understanding of the temporal and spatial
12 variability of fine particles. Refinement of these models to finer spatial scales may improve
13 exposure estimates in epidemiological studies as well as in quantitative risk and exposure
14 assessments
- 15 • Air Quality Distributions Reported in Epidemiological Studies. Most epidemiological
16 studies provide some information on the distribution of ambient measurement data evaluated,
17 however, published information is often generally limited in scope and the descriptive
18 statistics reported vary from one study to another. Understanding the air quality distributions
19 at which effects have been observed is important for informing consideration of the adequacy
20 of the current NAAQS as well as potential alternative indicators, averaging times, and levels
21 to consider. Working with intramural and extramural research groups, we plan to encourage
22 a more comprehensive and more consistent reporting of air quality data.

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3 REVIEW OF THE PRIMARY STANDARD FOR THORACIC COARSE PARTICLES

This chapter presents staff conclusions with regard to the adequacy of the current primary PM₁₀ standard, which is intended to protect public health against exposures to thoracic coarse particles (PM_{10-2.5}), and potential alternative primary standards for consideration in this review. Our assessment of these issues is framed by a series of key policy-relevant questions, which expand upon those presented in the IRP (US EPA, 2008a). The answers to these questions will inform decisions on whether, and if so how, to revise the current PM₁₀ standard.

Our approach for reviewing the primary PM₁₀ standard is presented in section 3.1. Our considerations and conclusions regarding the adequacy of the current PM₁₀ standard are presented in section 3.2. Section 3.3 presents our considerations and conclusions with respect to potential alternative standards, focusing on each of the basic elements of the standards: pollutant indicator (section 3.3.1), averaging time (section 3.3.2), form (section 3.3.3), and level (section 3.3.4). Section 3.4 summarizes staff conclusions on the current and potential alternative standards. Section 3.5 discusses key uncertainties and suggested future research areas and data collection efforts.

3.1 APPROACH

Staff's approach for reviewing the current primary PM₁₀ standard builds upon the approaches used in previous PM NAAQS reviews. The past and current approaches described below are all based most fundamentally on using information from epidemiologic studies to inform the selection of PM standards that, in the Administrator's judgment, protect public health with an adequate margin of safety. Evidence-based approaches to using information from epidemiologic studies to inform decisions on PM standards are complicated by the recognition that no population threshold, below which it can be concluded with confidence that PM-related effects do not occur, can be discerned from the available evidence (ISA, section 2.4.3). As a result, any approach to reaching decisions on what standards are appropriate requires judgments about how to translate the information available from the epidemiologic studies into a basis for appropriate standards, which includes consideration of how to weigh the uncertainties in reported associations across the distributions of PM concentrations in the studies. Our approach to informing these decisions, discussed more fully below, recognizes that the available health effects evidence reflects a continuum consisting of ambient levels at which scientists generally agree that health effects are likely to occur through lower levels at which the likelihood and magnitude of the response become increasingly uncertain. Such an approach is consistent with

1 setting standards that are neither more nor less stringent than necessary, recognizing that a zero-
2 risk standard is not required by the CAA.

3 **3.1.1 Reviews Completed in 1987 and 1997**

4 The PM NAAQS have always included some type of a primary standard to protect
5 against effects associated with exposures to thoracic coarse particles. In 1987, when EPA first
6 revised the PM NAAQS, EPA changed the indicator for PM from Total Suspended Particles
7 (essentially applicable to particles smaller than 25-45 micrometers) to focus on inhalable
8 particles, those which can penetrate into the trachea, bronchi, and deep lungs (52 FR 24634).
9 EPA changed the PM indicator to PM₁₀ based on evidence that the risk of adverse health effects
10 associated with particles of 10 micrometers or less was significantly greater than that associated
11 with larger particles (52 FR at 24639).

12 In the 1997 review, in conjunction with establishing new fine particle (i.e., PM_{2.5})
13 standards (see above, sections 1.2.3, 2.1.1), EPA concluded that continued protection remained
14 warranted against potential effects associated with thoracic coarse particles in the size range of
15 2.5 to 10 µm. This conclusion was based on particle dosimetry, toxicological information, and
16 on limited epidemiologic evidence from studies that measured PM₁₀ in areas where coarse
17 particles were likely to dominate the distribution (62 FR 38677, July 18, 1997). Thus, EPA
18 concluded that the existing PM₁₀ standards would provide requisite protection against effects
19 associated with particles in the size range of 2.5 to 10 µm. Although EPA considered a more
20 narrowly defined indicator for thoracic coarse particles in that review (i.e., PM_{10-2.5}), EPA
21 concluded that it was more appropriate, based on existing evidence, to continue to use PM₁₀ as
22 the indicator. This decision was based, in part, on the recognition that the only studies of clear
23 quantitative relevance to health effects most likely associated with thoracic coarse particles used
24 PM₁₀ in areas where the coarse fraction was the dominant fraction of PM₁₀, namely two studies
25 conducted in areas that substantially exceeded the 24-hour PM₁₀ standard (62 FR 38679). In
26 addition, there were only very limited ambient air quality data then available specifically for
27 thoracic coarse particles, in contrast to the extensive monitoring network already in place for
28 PM₁₀. Therefore, it was judged more administratively feasible to use PM₁₀ as an indicator. EPA
29 also stated that the PM₁₀ standards would work in conjunction with the PM_{2.5} standards by
30 regulating the portion of particulate pollution not regulated by the PM_{2.5} standards.

31 As explained in chapter 1, in May 1998, a three-judge panel of the U.S. Court of Appeals
32 for the District of Columbia Circuit found "ample support" for EPA's decision to regulate coarse
33 particle pollution, but vacated the 1997 PM₁₀ standards, concluding that EPA had failed to
34 adequately explain its choice of PM₁₀ as the indicator for thoracic coarse particles pointing to the
35 lack of reasoned explanation for the variable level of allowable concentrations of thoracic coarse

1 particles (varying by levels of PM_{2.5}) and the consequent double regulation of PM_{2.5}. *American*
2 *Trucking Associations v. EPA* , 175 F. 3d 1027, 1054-56 (D.C. Cir. 1999). The court also
3 rejected considerations of administrative feasibility as justification for a NAAQS, which are to
4 be based exclusively on health and welfare considerations. *Id.* at 1054. Pursuant to the court's
5 decision, EPA removed the vacated 1997 PM₁₀ standards from the Code of Federal Regulations
6 (CFR) (69 FR 45592, July 30, 2004) and deleted the regulatory provision (at 40 CFR section
7 50.6(d)) that controlled the transition from the pre-existing 1987 PM₁₀ standards to the 1997
8 PM₁₀ standards (65 FR 80776, December 22, 2000). The pre-existing 1987 PM₁₀ standards
9 remained in place. *Id.* at 80777.

10 **3.1.2 Review Completed in 2006**

11 In the review of the PM NAAQS that concluded in 2006, EPA considered the growing,
12 but still limited, body of evidence supporting associations between health effects and thoracic
13 coarse particles measured as PM_{10-2.5}.¹ The new studies available in the 2006 review included
14 epidemiologic studies that reported associations with health effects using direct measurements of
15 PM_{10-2.5}, as well as dosimetric and toxicological studies. In considering this growing body of
16 PM_{10-2.5} evidence, as well as evidence from studies that measured PM₁₀ in locations where the
17 majority of PM₁₀ was in the PM_{10-2.5} fraction (US EPA, 2005, section 5.4.1), staff concluded that
18 that the level of protection afforded by the existing 1987 PM₁₀ standard remained appropriate
19 (US EPA, 2005, p. 5-67), but recommended that the indicator for the standard be revised.
20 Specifically, staff recommended urban coarse particles in the size range of 10-2.5 micrometers as
21 the indicator, thus focusing on those thoracic coarse particles that are generally present in urban
22 environments (US EPA, 2005, p. 5-71). The agency proposed to retain a standard for a subset of
23 thoracic coarse particles, proposing a qualified PM_{10-2.5} indicator to focus on the mix of thoracic
24 coarse particles generally present in urban environments. More specifically, the proposed
25 revised thoracic coarse particle standard would have applied only to an ambient mix of PM_{10-2.5}
26 dominated by resuspended dust from high-density traffic on paved roads and/or by industrial and
27 construction sources. The proposed revised standard would not have applied to any ambient mix
28 of PM_{10-2.5} dominated by rural windblown dust and soils. In addition, agricultural sources,
29 mining sources, and other similar sources of crustal material would not have been subject to
30 control in meeting the standard (71 FR 2667 to 2668, January 17, 2006).

31 The Agency received a large number of comments overwhelmingly opposed to the
32 proposed qualified PM_{10-2.5} indicator (71 FR 61188 to 61197). After careful consideration of the

¹The PM Staff Paper (US EPA, 2005) also presented results of a quantitative assessment of health risks for PM_{10-2.5}. However, staff concluded that the nature and magnitude of the uncertainties and concerns associated with this risk assessment weighed against its use as a basis for recommending specific levels for a thoracic coarse particle standard.

1 scientific evidence and the recommendations contained in the 2005 Staff Paper, the advice and
2 recommendations from CASAC, the public comments received regarding the appropriate
3 indicator for coarse particles, and after extensive evaluation of the alternatives available to the
4 Agency, the Administrator decided it would not be appropriate to adopt a qualified PM_{10-2.5}
5 indicator. Underlying this determination was the decision that it was requisite to provide
6 protection from exposure to all thoracic coarse PM, regardless of its origin, rejecting arguments
7 that there are no health effects from community-level exposures to coarse PM in non-urban areas
8 (71 FR 61189). The EPA concluded that dosimetric, toxicological, occupational and
9 epidemiologic evidence supported retention of a primary standard for short-term exposures that
10 included all thoracic coarse particles (i.e. both urban and non-urban), consistent with the Act's
11 requirement that primary NAAQS provide an adequate margin of safety. At the same time, the
12 agency concluded that the standard should target protection toward urban areas, where the
13 evidence of health effects from exposure to PM_{10-2.5} was strongest (71 FR at 61193, 61197). The
14 proposed indicator was not suitable for that purpose. Not only did it inappropriately provide no
15 protection at all to many areas, but it failed to identify many areas where the ambient mix was
16 dominated by coarse particles contaminated with urban/industrial types of coarse particles for
17 which evidence of health effects was strongest (71 FR 61193).

18 The agency ultimately concluded that the existing indicator, PM₁₀, was most consistent
19 with the evidence. Although PM₁₀ includes both coarse and fine PM, the Agency concluded that
20 it remained an appropriate indicator for thoracic coarse particles because, as reported by Schmidt
21 et al. (2005), fine particle levels are generally higher in urban areas and, therefore, a PM₁₀
22 standard set at a single unvarying level will generally result in lower allowable concentrations of
23 thoracic coarse particles in urban areas than in non-urban areas. EPA considered this to be an
24 appropriate targeting of protection given that the strongest evidence for effects associated with
25 thoracic coarse particles came from epidemiologic studies conducted in urban areas and that
26 elevated fine particle concentrations in urban areas could result in increased contamination of
27 coarse fraction particles by PM_{2.5}, potentially increasing the toxicity of thoracic coarse particles
28 in urban areas (71 FR 61195-96). Given the evidence that the existing PM₁₀ standard afforded
29 requisite protection with an ample margin of safety, the Agency retained the level and form of
30 the 24-hour PM₁₀ standard.²

31 The Agency also revoked the annual PM₁₀ standard, in light of the conclusion in the PM
32 Criteria Document (US EPA, 2004) that the available evidence does not suggest an association
33 with long-term exposure to PM_{10-2.5} and the conclusion in the Staff Paper (US EPA, 2005) that
34 there is no quantitative evidence that directly supports an annual standard.

²Thus, the standard is met when a PM₁₀ concentration of 150 µg/m³ is not exceeded more than once per year on average over a three-year period.

1 In the same rulemaking, EPA also included a new Federal Reference Method (FRM) for
2 the measurement of PM_{10-2.5} in the ambient air (71 FR 61212 to 61213). Although the standard
3 for thoracic coarse particles does not use a PM_{10-2.5} indicator, the new FRM for PM_{10-2.5} was
4 established to provide a basis for approving Federal Equivalent Methods (FEMs) and to promote
5 the gathering of scientific data to support future reviews of the PM NAAQS.

6 **3.1.3 Litigation of 2006 Final Rule for Thoracic Coarse Particles**

7 A number of groups filed suit in response to the final decisions made in the 2006 review.
8 See *American Farm Bureau Federation and the National Pork Producers Council v. EPA* (DC
9 Cir. 2009). Among the petitions for review were challenges from industry groups on the
10 decision to retain the PM₁₀ indicator and the level of the PM₁₀ standard and from environmental
11 and public health groups on the decision to revoke the annual PM₁₀ standard. The court upheld
12 both the daily PM₁₀ standard and the decision to revoke the annual standard.

13 First, the court upheld EPA's decision for a standard to cover all thoracic coarse PM,
14 both of urban and non-urban origin. The court rejected arguments that the evidence showed
15 there are no risks from exposure to non-urban coarse PM. The court further found that EPA had
16 a reasonable basis not to set separate standards for urban and non-urban coarse PM, namely the
17 inability to reasonably define what ambient mixes would be included under either 'urban' or
18 'non-urban;' and that the evidence in the record supported EPA's cautious decision to provide
19 "some protection from exposure to thoracic coarse particles... in all areas." 559 F. 3d at 532-33.
20 Specifically, the court stated,

21
22 Although the evidence of danger from coarse PM is, as EPA recognizes,
23 "inconclusive," (71 FR 61193, October 17, 2006), the agency need not wait for
24 conclusive findings before regulating a pollutant it reasonably believes may pose
25 a significant risk to public health. The evidence in the record supports the EPA's
26 cautious decision that "some protection from exposure to thoracic coarse particles
27 is warranted in all areas." *Id.* As the court has consistently reaffirmed, the CAA
28 permits the Administrator to "err on the side of caution" in setting NAAQS.
29 559 F. 3d at 533.
30

31 The court also upheld EPA's decision to retain the level of the standard at 150 µg/m³ and
32 to use PM₁₀ as the indicator for a standard meant to protect against exposures to thoracic coarse
33 particles. In upholding the level of the standard, the court referred to the conclusion in the Staff
34 Paper that there is "little basis for concluding that the degree of protection afforded by the
35 current PM₁₀ standards in urban areas is greater than warranted, since potential mortality effects
36 have been associated with air quality levels not allowed by the current 24-hour standard, but
37 have not been associated with air quality levels that would generally meet that standard, and

1 morbidity effects have been associated with air quality levels that exceeded the current 24-hour
2 standard only a few times.” 559 F. 3d at 534. The court also rejected arguments that a PM₁₀
3 standard established at an unvarying level will result in arbitrarily varying levels of protection
4 given that the level of coarse PM would vary based on the amount of fine PM present. The court
5 agreed that the variation in allowable coarse PM accorded with the strength of the evidence:
6 typically less coarse PM would be allowed in urban areas (where levels of fine PM are typically
7 higher), in accord with the strongest evidence of health effects from coarse particles. 559 F. 3d
8 at 535-36. In addition, such regulation would not impermissibly double regulate fine particles,
9 since any additional regulation of fine particles (beyond that afforded by the primary PM_{2.5}
10 standard) would be for a different purpose: to prevent contamination of coarse particles by fine
11 particles. 559 F. 3d at 535, 536. These same explanations explained the choice of PM₁₀ as an
12 indicator, and provided the reasoned explanation for that choice lacking in the record for the
13 1997 standard. 559 F. 3d at 536.

14 With regard to the challenge from environmental and public health groups, the court
15 upheld EPA’s decision to revoke the annual PM₁₀ standard. Specifically, the court stated the
16 following:

17 The EPA reasonably decided that an annual coarse PM standard is not necessary
18 because, as the Criteria Document and the Staff Paper make clear, the latest
19 scientific data do not indicate that long-term exposure to coarse particles poses a
20 health risk. The CASAC also agreed that an annual coarse PM standard is
21 unnecessary. 559 F. 3d at 538-39.
22

23 **3.1.4 Current Approach**

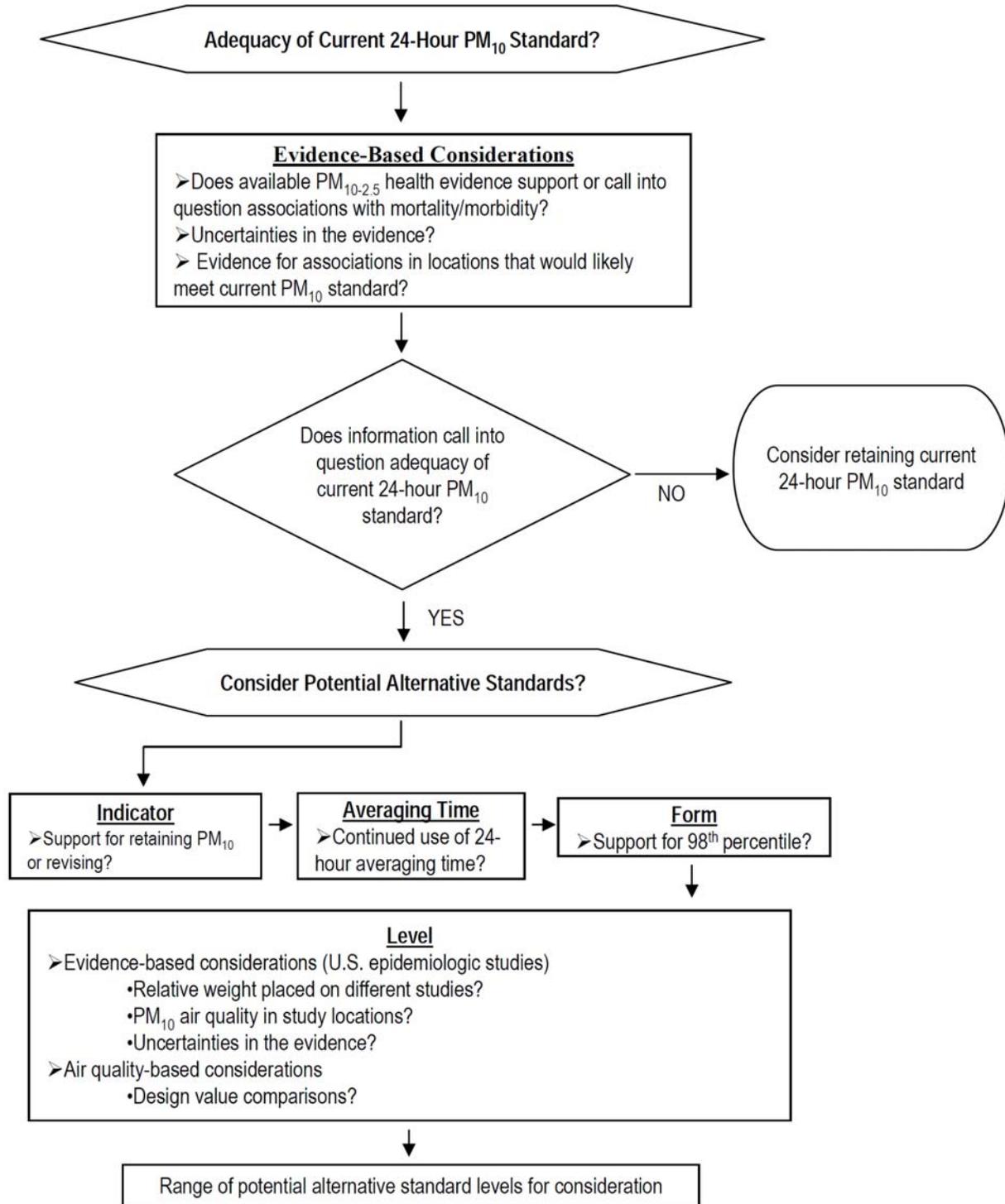
24 Our approach relies most heavily on the health evidence, primarily the epidemiologic
25 evidence, assessed in the ISA (US EPA, 2009a) and on available PM air quality information. As
26 discussed in more detail in the *Quantitative Health Risk Assessment for Particulate Matter –*
27 *Final* (US EPA, 2010a), we have not conducted a quantitative assessment of health risks
28 associated with PM_{10-2.5}. Staff concluded that limitations in the monitoring network and in the
29 health studies that rely on that monitoring network, which would be the basis for estimating
30 PM_{10-2.5} health risks, would introduce significant uncertainty into a PM_{10-2.5} risk assessment such
31 that the risk estimates generated would be of limited value in informing review of the standard.
32 Therefore, staff concluded in the RA that a quantitative risk assessment for PM_{10-2.5} is not
33 supportable at this time (US EPA, 2010a, p. 2-6).

34 For purposes of this policy assessment, we seek to provide as broad an array of options
35 for consideration as is supportable by the available evidence and air quality information,
36 recognizing that the final decisions on the primary PM₁₀ standard will reflect the judgments of
37 the Administrator. In developing these options for consideration, we consider the available

1 evidence and air quality information that informs overarching questions related to: (1) the
2 adequacy of the current 24-hour PM₁₀ standard to protect against effects associated with
3 exposures to thoracic coarse particles and (2) what potential alternative standard(s), if any,
4 should be considered in this review. In addressing these broad questions, we have organized the
5 discussions below around a series of more specific questions reflecting different aspects of each
6 overarching question. When evaluating the health protection afforded by the current or potential
7 alternative standards, we have taken into account the four basic elements of the NAAQS (e.g.,
8 indicator, averaging time, form, and level).

9 Figure 3-1 provides an overview of the policy-relevant questions that frame our review,
10 as discussed more fully below. We believe the approach outlined here, when presented in the
11 final Policy Assessment, will provide a comprehensive basis to help inform the judgments
12 required of the Administrator in reaching decisions about the current and potential alternative
13 primary standards meant to protect public health against exposures to thoracic coarse particles.

1 **Figure 3-1. Approach to Considering the Current and Potential Alternative Thoracic**
 2 **Coarse Particle Standards**



3

3.2 ADEQUACY OF THE EXISTING PM₁₀ STANDARD

In considering the adequacy of the current 24-hour PM₁₀ standard to protect against effects associated with exposures to thoracic coarse particles, we address the following overarching question:

Does the available scientific evidence, as reflected in the ISA, support or call into question the adequacy of the protection afforded by the current 24-hour PM₁₀ standard against effects associated with exposures to thoracic coarse particles?

To inform our consideration of this overarching question, we consider the scientific evidence for associations between PM_{10-2.5} and mortality and morbidity, evidence linking PM_{10-2.5} toxicity to specific sources/locations, uncertainties in the evidence, and available PM₁₀ air quality concentrations in PM_{10-2.5} study locations (section 3.2.1). Evidence for populations that are particularly susceptible to PM exposures is discussed in detail in section 2.2.1 above, and is not repeated here. Staff conclusions regarding the adequacy of the current standard are presented in section 3.2.2.

3.2.1 Evidence-Based Considerations

In considering the currently available body of scientific evidence for health effects of thoracic coarse particles, we consider the following question:

- To what extent does the available scientific evidence support or call into question associations between ambient thoracic coarse particle exposures and adverse health effects?**

Since the conclusion of the last review, the Agency has developed a more formal framework for reaching causal inferences from the body of scientific evidence. As discussed above in section 2.2.1, this framework uses a five-level hierarchy that classifies the overall weight of evidence using the following categorizations: causal relationship, likely to be a causal relationship, suggestive of a causal relationship, inadequate to infer a causal relationship, and not likely to be a causal relationship (ISA, section 1.5, Table 1-3). Applying this framework to thoracic coarse particles, the ISA concludes that the existing evidence is suggestive of a causal relationship between short-term PM_{10-2.5} exposures and mortality, cardiovascular effects, and respiratory effects (US EPA, 2009a, section 2.3.3; see Table 3-1 below). In contrast, the ISA concludes that available evidence is *inadequate to infer a causal relationship* between long-term PM_{10-2.5} exposures and various health effects (US EPA, 2009a, section 2.3; Table 3-1 below). Similar to the judgment made in the AQCD regarding long-term exposures (US EPA, 2004), the ISA states, “To date, a sufficient amount of evidence does not exist in order to draw conclusions regarding the health effects and outcomes associated with long-term exposure to PM_{10-2.5}” (US

1 EPA, 2009a, section 2.3.4). Given these weight of evidence conclusions in the ISA, our
 2 evidence-based considerations regarding the adequacy of the current 24-hour PM₁₀ standard
 3 focus on effects that have been linked with short-term exposures to PM_{10-2.5}.

4 As noted above, in the last review of the PM NAAQS, PM₁₀ studies conducted in
 5 locations where PM₁₀ is comprised predominantly of PM_{10-2.5} were also considered (US EPA,
 6 2005, pp. 5-49 to 5-50). However, PM₁₀ studies are difficult to interpret within the context of a
 7 standard meant to protect against exposures to PM_{10-2.5} because PM₁₀ is comprised of both fine
 8 and coarse particles, even in locations with the highest concentrations of PM_{10-2.5} (see below). In
 9 light of the considerable uncertainty in the extent to which PM₁₀ effect estimates reflect
 10 associations with PM_{10-2.5}, together with the availability in this review of recent studies that
 11 evaluate associations with PM_{10-2.5} specifically and the fact that the ISA weight of evidence
 12 conclusions for thoracic coarse particles were based on studies of PM_{10-2.5}, we focus in this
 13 Policy Assessment on studies that have specifically evaluated PM_{10-2.5}. The evidence supporting
 14 a link between short-term thoracic coarse particle exposures and adverse health effects is
 15 discussed in detail in the ISA (US EPA, 2009a, Chapter 6) and is summarized briefly below for
 16 mortality, cardiovascular effects, and respiratory effects.

17 **Table 3-1. Summary of Causality Determinations for PM_{10-2.5}**

Exposure Duration	Outcome	Causal Determination
Short-term	Cardiovascular Effects	Suggestive
	Respiratory Effects	Suggestive
	Mortality	Suggestive
	Central Nervous System Effects	Inadequate
Long-term	Cardiovascular Effects	Inadequate
	Respiratory Effects	Inadequate
	Mortality	Inadequate
	Reproductive and Developmental Effects	Inadequate
	Cancer Mutagenicity, Genotoxicity Effects	Inadequate

18 Source: adapted from US EPA, 2009a; Table 2-6

19 Short-Term PM_{10-2.5} and Mortality

20 The ISA assesses a number of multi-city and single-city epidemiologic studies that have
 21 evaluated associations between mortality and short-term PM_{10-2.5} concentrations (US EPA,
 22 2009a, Figure 6-30 presents PM_{10-2.5} mortality studies assessed in the last review and the current
 23 review). Different studies have used different approaches to estimate ambient PM_{10-2.5}. Some
 24 studies have used the difference between PM₁₀ and PM_{2.5} mass, either measured at co-located

1 monitors (e.g., Lipfert et al., 2000; Mar et al., 2003; Ostro et al., 2003; Sheppard et al., 2003;
2 Wilson et al., 2007) or as the difference in county-wide average concentrations (Zanobetti and
3 Schwartz, 2009), while other studies have measured PM_{10-2.5} directly with dichotomous samplers
4 (e.g., Burnett and Goldberg, 2003; Fairley et al., 2003; Burnett et al., 2004; Klemm et al., 2004).
5 Despite differences in the approaches used to estimate ambient PM_{10-2.5} concentrations, the
6 majority of multi- and single-city studies have reported positive associations between PM_{10-2.5}
7 and mortality, though most of these associations were not statistically significant (US EPA,
8 2009a, Figure 6-30). When considered as a whole, the ISA concluded that epidemiologic studies
9 have reported consistent, positive associations between short-term PM_{10-2.5} and mortality (US
10 EPA, 2009a, section 6.5.2.3).

11 In considering specific mortality studies, we note that the U.S. multi-city study by
12 Zanobetti and Schwartz (2009) reported positive and statistically significant associations with
13 PM_{10-2.5} for all-cause, cardiovascular, and respiratory mortality (US EPA, 2009a, section 6.5.2.3)
14 while other multi-city studies have reported positive, but not statistically significant, PM_{10-2.5}
15 effect estimates (US EPA, 2009a, Figure 6-30, Burnett and Goldberg, 2003; Klemm et al., 2003;
16 Burnett et al., 2004). In the study by Zanobetti and Schwartz, the effect estimates for all-cause
17 and respiratory mortality remained statistically significant in co-pollutant models that included
18 PM_{2.5}, while the effect estimate for cardiovascular mortality remained positive but not
19 statistically significant. When risk estimates in this study were evaluated by climatic region (US
20 EPA, 2009a, Figure 6-28), the “dry continental” region, which included areas with relatively
21 high PM_{10-2.5} concentrations (see US EPA, 2009a, Figure 6-29; Schmidt and Jenkins, 2010; and
22 discussion of regional differences in PM_{10-2.5} concentrations below) such as Salt Lake City,
23 Provo, and Denver, showed the largest risk estimates. However, the “dry” region, which
24 included Phoenix and Albuquerque, two locations that also have relatively high PM_{10-2.5}
25 concentrations, did not show positive associations with all-cause or respiratory mortality and
26 only a relatively small positive association for cardiovascular mortality. In addition, the
27 “Mediterranean” region (which included cities in California, Oregon, and Washington) did not
28 show positive associations while the other three regions (i.e., “hot summer, continental,” “warm
29 summer, continental,” and “humid, subtropical and maritime”), which included cities that
30 correspond to the mid-west, northeast, and southeast geographic regions, all showed positive
31 associations (US EPA, 2009a, Figure 6-28).

32 The ISA also presents empirical Bayes-adjusted city-specific effect estimates for cities
33 evaluated by Zanobetti and Schwartz (US EPA, 2009a, Figure 6-29). City-specific estimates
34 were positive, though generally not statistically significant, for cardiovascular mortality in all 47
35 cities evaluated. Effect estimates were positive for all-cause and respiratory mortality in all
36 cities except Los Angeles (negative for all-cause and respiratory) and Phoenix (negative for

1 respiratory) (US EPA, 2009a, Figure 6-29). In addition, positive and statistically significant
2 associations between mortality (all-cause, cardiovascular, and/or respiratory) and PM_{10-2.5} were
3 reported for six locations (St. Louis, MO; Salt Lake City, UT; Chicago, IL; Pittsburgh, PA;
4 Detroit, MI; and Birmingham, AL).

5 In considering single-city PM_{10-2.5} mortality studies, we note that all of the studies
6 included in Figure 6-30 of the ISA (US EPA, 2009a) reported positive PM_{10-2.5} effect estimates,
7 with three single-city studies reporting effect estimates that were statistically significant (Mar et
8 al., 2003; Ostro et al., 2003; Wilson et al., 2007). One study reported a negative PM_{10-2.5} effect
9 estimate for respiratory mortality (Villeneuve et al., 2003), though effect estimates for all-cause
10 and cardiovascular mortality were positive in this study (US EPA, 2009a, Figure 6-30).

11 Short-Term PM_{10-2.5} and Cardiovascular Effects

12 The ISA assesses a number of studies that have evaluated the link between short-term
13 ambient concentrations of thoracic coarse particles and cardiovascular effects. In considering the
14 available epidemiologic evidence, the ISA concludes that single- and multi-city epidemiologic
15 studies generally report positive associations between short-term PM_{10-2.5} concentrations and
16 hospital admissions or emergency department visits for cardiovascular causes (US EPA, 2009a,
17 section 2.3.3, 6.2.12.2). Some of these studies have reported positive and statistically significant
18 PM_{10-2.5} effect estimates in co-pollutant models while others report that PM_{10-2.5} effect estimates
19 remain positive, but not statistically significant (US EPA, 2009a, Figure 6-5).

20 These studies include a recent U.S. multi-city study evaluating hospital admissions and
21 emergency department visits for cardiovascular disease in Medicare patients (Peng et al., 2008).
22 In this study of older adults, the authors reported a positive and statistically significant
23 association between 24-hour PM_{10-2.5} concentrations and cardiovascular disease hospitalizations
24 in a single pollutant model using air quality data for 108 U.S. counties with co-located PM₁₀ and
25 PM_{2.5} monitors. The effect estimate was reduced only slightly in a two-pollutant model that
26 included PM_{2.5}, but it was no longer statistically significant (US EPA, 2009a, sections 2.3.3,
27 6.2.10.9). Regional and/or county-specific analyses were not conducted for the locations
28 evaluated by Peng. Therefore, it is not possible, based on the data from this study, to consider
29 differences in PM_{10-2.5} effect estimates in specific locations or regions. In addition to this U.S.
30 multi-city study, positive associations reported for short-term PM_{10-2.5} and cardiovascular
31 morbidity reached statistical significance in a multi-city study in France (Host et al., 2007) and
32 single-city studies in Detroit and Toronto (US EPA, 2009a, Figures 6-2, 6-3). In contrast,
33 associations were positive but not statistically significant in single-city studies conducted in
34 Atlanta and Boston (and for some endpoints in Detroit) (US EPA, 2009a, Figures 6-1 to 6-3, 6-
35 5).

1 The plausibility of the positive associations reported for PM_{10-2.5} and cardiovascular-
2 related hospital admissions and emergency department visits is supported by a small number of
3 controlled human exposure studies that have reported alterations in heart rate variability
4 following exposure to PM_{10-2.5}; by short-term PM_{10-2.5} epidemiologic studies reporting positive
5 associations with cardiovascular mortality; by a small number of recent epidemiologic studies
6 that have examined dust storm events and reported increases in cardiovascular-related
7 emergency department visits and hospital admissions (see below); and by associations with other
8 cardiovascular effects including heart rhythm disturbances and changes in heart rate variability
9 (US EPA, 2009a, sections 2.3.3, 6.2.12.2). The few toxicological studies that examined the
10 effect of PM_{10-2.5} on cardiovascular health effects used intratracheal instillation and, as a result,
11 provide only limited evidence on the biological plausibility of PM_{10-2.5} induced cardiovascular
12 effects (US EPA, 2009a, sections 2.3.3, 6.2.12.2).

13 Short-Term PM_{10-2.5} and Respiratory Effects

14 The ISA also assesses a number of studies that have evaluated the link between short-
15 term ambient concentrations of thoracic coarse particles and respiratory effects. This includes
16 recent studies conducted in the U.S., Canada, and France (US EPA, 2009a, section 6.3.8),
17 including the U.S. multi-city study of Medicare patients by Peng et al. (2009). As discussed
18 above, Peng estimated PM_{10-2.5} concentrations as the difference between PM₁₀ and PM_{2.5}
19 concentrations measured by co-located monitors. The authors reported a positive, but not
20 statistically significant, PM_{10-2.5} effect estimate for respiratory-related hospital admissions.
21 Single-city studies have reported positive, and in some cases statistically significant, PM_{10-2.5}
22 effect estimates for respiratory-related hospital admissions and emergency department visits (US
23 EPA, 2009a, Figures 6-10 to 6-15). Some of these PM_{10-2.5} respiratory morbidity studies have
24 reported positive and statistically significant PM_{10-2.5} effect estimates in co-pollutant models that
25 included gaseous pollutants while others reported that PM_{10-2.5} effect estimates remain positive,
26 but not statistically significant, in such co-pollutant models (US EPA, 2009a, Figure 6-15).

27 A limited number of epidemiologic studies have focused on specific respiratory
28 morbidity outcomes and reported both positive and negative, but generally not statistically
29 significant, associations between PM_{10-2.5} and lower respiratory symptoms, wheeze, and
30 medication use (US EPA, 2009a, sections 2.3.3.1 and 6.3.1.1; Figures 6-7, 6-8, 6-9). Although
31 controlled human exposure studies have not observed an effect on lung function or respiratory
32 symptoms in healthy or asthmatic adults in response to short-term exposure to PM_{10-2.5}, healthy
33 volunteers have exhibited an increase in markers of pulmonary inflammation. Toxicological
34 studies using inhalation exposures are still lacking, but pulmonary injury and inflammation has
35 been reported in animals after intratracheal instillation exposure (US EPA, 2009a, section
36 6.3.5.3) and, in some cases, PM_{10-2.5} was found to be more potent than PM_{2.5}.

1 PM_{10-2.5} Toxicity: Impacts of Sources and Composition

2 As discussed above, positive, and in some cases statistically significant, associations
3 between short-term PM_{10-2.5} concentrations and mortality and morbidity have been reported in a
4 number of different locations. Little is known about how PM_{10-2.5} composition varies across
5 these locations and how that variation could affect particle toxicity (US EPA, 2009a, sections
6 2.3.3, 2.3.4, 2.4.4). However, the limited available evidence suggests that specific components
7 of thoracic coarse particles tend to comprise different fractions of PM_{10-2.5} mass in different
8 environments (e.g., urban versus rural environments) (US EPA, 2009a, section 3.5.1.1; Schmidt
9 et al., 2005; Edgerton et al., 2009). It is possible that such differences in particle composition
10 could affect particle toxicity, though the ISA concludes that existing evidence is not sufficient to
11 draw distinctions in toxicity based on composition and notes that recent studies have reported
12 that PM from different sources, including crustal sources, is associated with adverse health
13 effects (US EPA, 2009a, section 2.4.4). The evidence for associations with particles originating
14 from different types of sources and in different locations is discussed briefly below.

15 As discussed above, most PM_{10-2.5} epidemiologic studies have been conducted in urban
16 locations in the U.S., Canada, and Europe while a small number of studies have examined the
17 health impacts of dust storm events (US EPA, 2009a, sections 6.2.10.1, 6.5.2.3). Although these
18 dust storm studies do not link specific particle constituents to health effects, it is useful to
19 consider them within the context of the toxicity of particles of non-urban crustal origin. Several
20 studies have reported positive and statistically significant associations between dust storm events
21 and morbidity or mortality, including the following:

- 22 • Middleton et al. (2008) reported that dust storms in Cyprus were associated with a
23 statistically significant increase in risk of hospitalization for all causes and a non-
24 significant increase in hospitalizations for cardiovascular disease.
- 25 • Chan et al. (2008) studied the effects of Asian dust storms on cardiovascular hospital
26 admissions in Taipei, Taiwan and reported a statistically significant increase associated
27 with 39 Asian dust events. Evaluating the same data, Bell et al. (2008) also reported
28 positive and statistically significant associations between hospitalization for ischemic
29 heart disease and PM_{10-2.5}.
- 30 • Perez et al. (2008) tested the hypothesis that outbreaks of Saharan dust exacerbate the
31 effects of PM_{10-2.5} on daily mortality in Spain. During Saharan dust days, the PM_{10-2.5}
32 effect estimate was larger than on non-dust days and it became statistically significant,
33 whereas it was not statistically significant on non-dust days.

34 In contrast to the studies noted above, some dust storm studies have reported associations that
35 were not statistically significant. Specifically, Bennett et al. (2006) reported on a dust storm in
36 the Gobi desert that transported PM across the Pacific Ocean, reaching western North America in

1 the spring of 1998. The authors reported no excess risk of cardiac or respiratory hospital
2 admissions associated with the dust storm in the population of British Columbia’s Lower Fraser
3 Valley (Bennett et al., 2006). In addition, Yang et al. (2009) reported that hospitalizations for
4 congestive heart failure were elevated during or immediately following 54 Asian dust storm
5 events, though effect estimates were not statistically significant. The implications of these
6 studies for the current review, for consideration of potential standard indicators, are discussed
7 below.

8 Next we consider uncertainties associated with the evidence by addressing the following
9 question:

- 10 • **What are the important uncertainties associated with the currently available scientific**
11 **evidence that should be considered in evaluating the adequacy of the current PM₁₀**
12 **standard?**

13 The majority of the health evidence supporting the link between short-term thoracic
14 coarse particle exposures and mortality and morbidity comes from epidemiologic studies.
15 Although new studies have become available since the last review and have expanded our
16 understanding of the association between PM_{10-2.5} and adverse health effects (see above and U.S.
17 EPA, 2009a, Chapter 6), important uncertainties remain. These uncertainties, and their
18 implications for interpreting the scientific evidence, are discussed below.

19 The ISA (sections 2.3.3, 2.3.4) concludes that an important uncertainty in the PM_{10-2.5}
20 epidemiologic literature is that associated with the air quality estimates used in these studies.
21 Specifically, the ISA concludes that there is greater error in estimating ambient exposures to
22 PM_{10-2.5} than to PM_{2.5} and that such uncertainty is a particularly relevant consideration when
23 interpreting PM_{10-2.5} epidemiologic studies. Contributing to this uncertainty is the relatively
24 limited spatial coverage provided by existing PM_{10-2.5} monitors (US EPA, 2009a, sections 2.2.3,
25 2.3.3, 2.3.4, 3.5.1.1 and see above). As discussed above, a national network to monitor PM_{10-2.5}
26 is not in place, limiting the spatial area over which PM_{10-2.5} concentrations are measured. In
27 addition, based on the limited available evidence, the ISA concluded that “there is greater spatial
28 variability in PM_{10-2.5} concentrations than PM_{2.5} concentrations, resulting in increased exposure
29 error for the larger size fraction” (US EPA, 2009a, p. 2-8) and that available measurements do
30 not provide sufficient information to adequately characterize the spatial distribution of PM_{10-2.5}
31 concentrations (US EPA, 2009a, section 3.5.1.1). The net effect of these uncertainties on
32 epidemiologic studies of PM_{10-2.5} is to bias the results of such studies toward the null hypothesis.
33 That is, as noted in the ISA, these limitations in estimates of ambient PM_{10-2.5} concentrations
34 “would tend to increase uncertainty and make it more difficult to detect effects of PM_{10-2.5} in
35 epidemiologic studies” (US EPA 2009a, p. 2-21).

1 Given these limitations in PM_{10-2.5} monitoring, different epidemiologic studies have
2 employed different approaches to estimating PM_{10-2.5} concentrations, further contributing to
3 uncertainty in interpreting these studies. For example, as discussed above, the multi-city study
4 by Peng et al. (2008) estimated PM_{10-2.5} by taking the difference between collocated PM₁₀ and
5 PM_{2.5} monitors while the study by Zanobetti and Schwartz (2009) used the difference between
6 county average PM₁₀ and PM_{2.5} concentrations. A small number of studies have directly
7 measured PM_{10-2.5} concentrations with dichotomous samplers (e.g., Burnett et al., 2004;
8 Villeneuve et al., 2003; Klemm et al., 2004). It is not clear how computed PM_{10-2.5}
9 measurements, such as those used by Zanobetti and Schwartz, compare with the PM_{10-2.5}
10 concentrations obtained in other studies either by direct measurement with a dichotomous
11 sampler or by calculating the difference using co-located samplers (US EPA, 2009a, section
12 6.5.2.3).³ Given the use of these different approaches to estimating PM_{10-2.5} concentrations
13 across studies, and their inherent limitations, the distributions of thoracic coarse particle
14 concentrations over which reported health outcomes occur remain highly uncertain.

15 The ISA also notes that the potential for confounding by co-occurring pollutants,
16 particularly PM_{2.5}, has been addressed in only a relatively small number of PM_{10-2.5}
17 epidemiologic studies, introducing additional uncertainty into the interpretation of these studies
18 (US EPA, 2009a, section 2.3.3). This is a particularly important consideration given the
19 relatively limited body of experimental evidence available to support the plausibility of
20 associations between PM_{10-2.5} itself and health effects reported in epidemiologic studies. As
21 discussed above, many epidemiologic studies that have evaluated co-pollutant models have
22 reported that PM_{10-2.5} effect estimates remain positive, but lose precision and are not statistically
23 significant in these models (US EPA, 2009a, Figures 6-5, 6-9, 6-15). The net effect of this
24 limitation in the number of epidemiologic studies that have evaluated co-pollutant models,
25 combined with the limited number of supporting experimental studies, is to increase the
26 uncertainty associated with estimates of the extent to which PM_{10-2.5} itself, rather than one or
27 more co-occurring pollutants, is responsible for reported health effects.

28 Another uncertainty results from the relative lack of information on the chemical and
29 biological composition of PM_{10-2.5}, and the effects associated with the various components (ISA,
30 section 2.3.4). As discussed above, a few recent studies have evaluated associations between
31 health effects and particles of non-urban, crustal origin by evaluating the health impacts of sand
32 storm events. Though these studies provide some information on the health effects of particles
33 that likely differ in composition from the particles of urban origin that are typically studied,

³In addition, when the difference between PM_{2.5} and PM₁₀ is calculated, the potential for differences among operational flow rates and temperatures for PM₁₀ and PM_{2.5} monitors add to the potential for exposure misclassification.

1 without more information on the chemical speciation of PM_{10-2.5}, the apparent variability in
2 associations with health effects across locations is difficult to characterize (US EPA, 2009a,
3 section 6.5.2.3).

4 As discussed above, a 24-hour PM₁₀ standard is in place to protect the public health
5 against exposures to thoracic coarse particles. Therefore, in further considering the adequacy of
6 the current PM₁₀ standard, we ask the following question:

- 7 • **To what extent does the available scientific evidence report associations between**
8 **PM_{10-2.5} and morbidity and mortality in areas that would likely meet the current PM₁₀**
9 **standard?**

10 In addressing this question, we have used EPA's Air Quality System (AQS)⁴ to
11 characterize PM₁₀ concentrations in U.S. locations where both single-city and multi-city PM_{10-2.5}
12 studies have been conducted (see U.S. EPA, 2009a, Figures 6-1 to 6-30 for studies). When
13 compared to single-city studies, we note that multi-city studies assess PM_{10-2.5}-associated health
14 effects among larger study populations, providing enhanced power to detect PM_{10-2.5}-associated
15 health effects. In addition, multi-city studies often provide spatial coverage for different regions
16 across the country, reflecting differences in PM_{10-2.5} sources, composition, and potentially other
17 factors that could impact PM_{10-2.5}-related effects. These factors make multi-city studies
18 particularly important when drawing conclusions about health effect associations. However,
19 multi-city studies often present overall effect estimates rather than single-city effect estimates,
20 while short-term air quality can vary considerably across cities. Therefore, the extent to which
21 effects reported in multi-city studies are associated with the short-term air quality in any
22 particular location is uncertain, especially when considering short-term concentrations at the
23 upper end of the distribution of daily concentrations for pollutants with relatively heterogeneous
24 spatial distributions such as PM_{10-2.5} and PM₁₀ (US EPA, 2009a, section 2.1.1.2). In contrast,
25 single-city studies are more limited in terms of power and geographic coverage but the link
26 between reported health effects and the short-term air quality in a given city is more
27 straightforward to establish. As a result, in considering 24-hour PM₁₀ concentrations in locations
28 of epidemiologic studies, we have focused below primarily on single-city studies (Figures 3-2
29 and 3-3) and single-city analyses of the locations evaluated in the multi-city study by Zanobetti
30 and Schwartz (US EPA, 2009a, Figure 6-29).

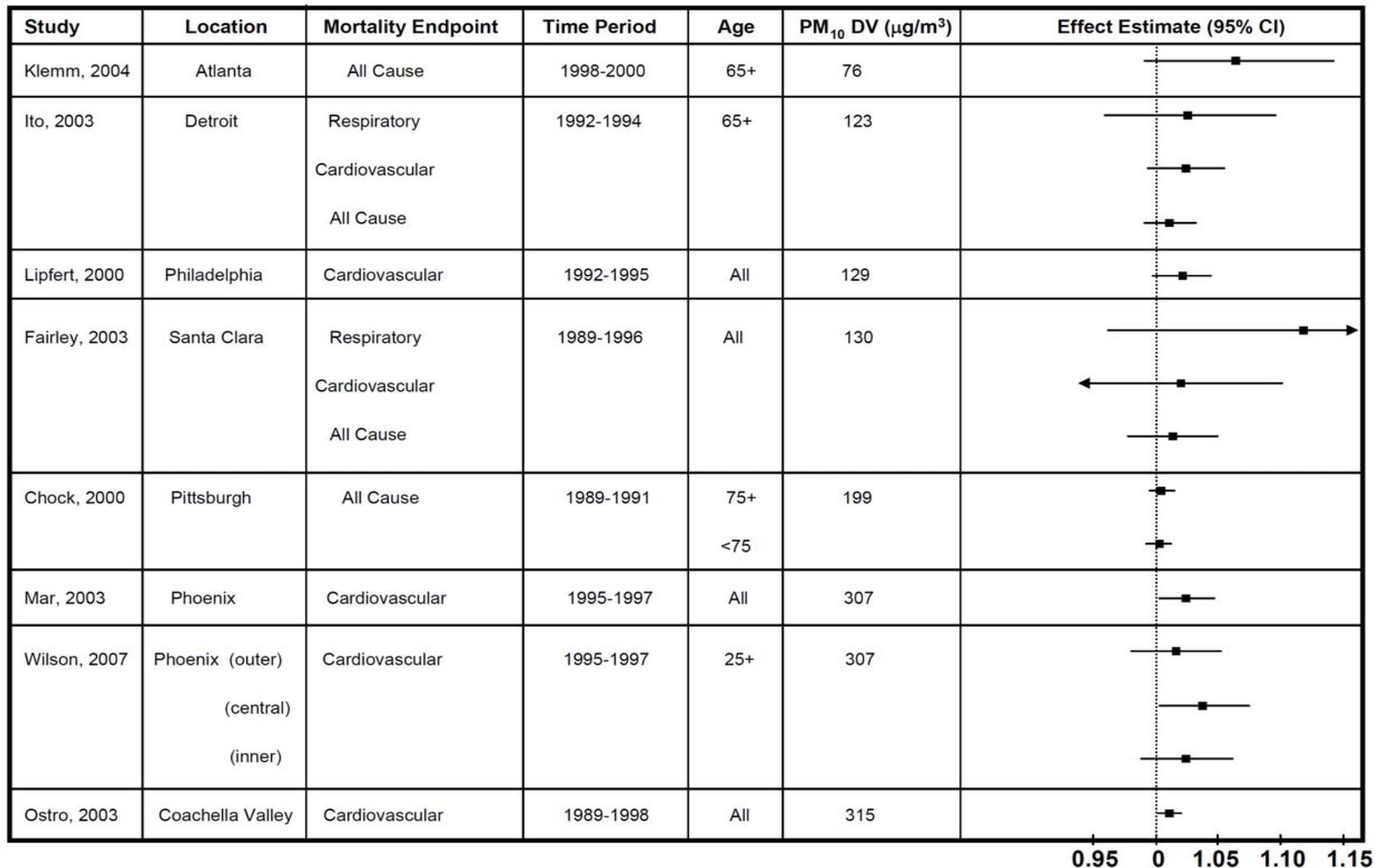
31 The current PM₁₀ standard has a form of one-expected-exceedance per year, averaged
32 over 3 years. In order to compare PM₁₀ concentrations in study locations to the level of the
33 current standard, we have identified the PM₁₀ 3-year expected exceedance concentration-
34 equivalent design value for each study period (labeled "DV" in Figures 3-2 and 3-3 below) using

⁴ <http://www.epa.gov/ttn/airs/airsaqs/>

1 the protocol specified in the PM₁₀ State Implementation Plan (SIP) Development Guidelines (US
2 EPA, 1987).⁵ Some studies (indicated by a * in Figure 3-3) covered time periods of less than
3 three years. For these study areas, to characterize ambient PM₁₀ concentrations relative to
4 concentrations allowed under the current PM₁₀ standard, we averaged the second highest 24-hour
5 PM₁₀ concentrations for each year of the study (i.e., second highest concentration measured at
6 the single monitor in the study area recording the highest such concentration). The identification
7 of concentration-equivalent design values and second highest PM₁₀ concentrations for each study
8 area are described in more detail in the memo by Schmidt and Jenkins (2010).

⁵Specifically, the PM₁₀ 3-year expected exceedance concentration-equivalent design value is identified as the highest 24-hour average concentration (i.e., from a single monitor in the study area) over a 3-year period when there are 347 or fewer samples reported for that time-frame, the second highest 24-hour average concentration when there are 348 to 695 samples in the 3-year period, the third highest 24-hour average concentration when there are 696 to 1042 samples in the 3-year period, and the fourth highest 24-hour average concentration when there are 1043 or more samples reported over the 3-year period. Concentration-equivalent design values were not identified for study periods less than 3 years.

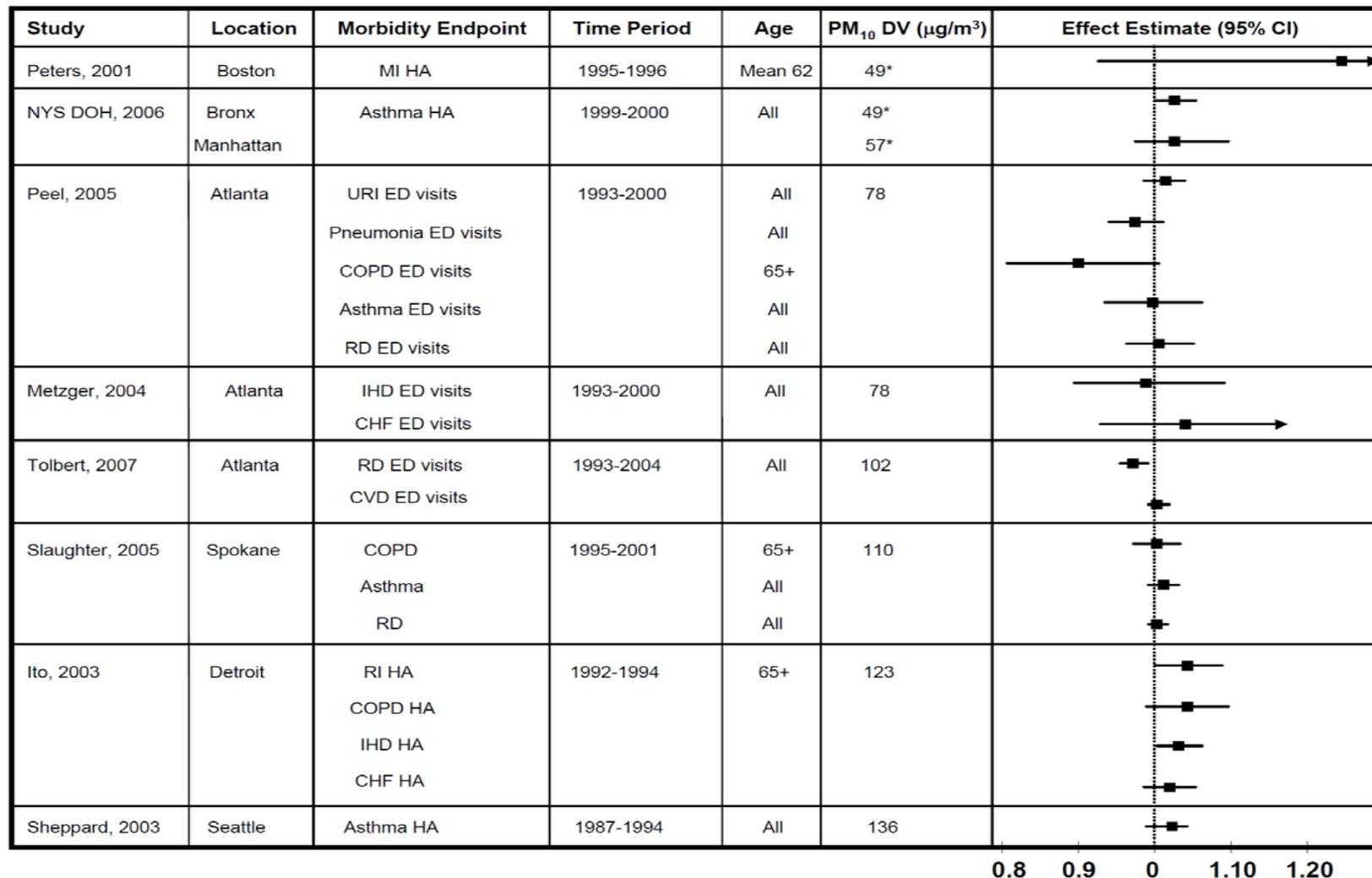
1 **Figure 3-2. PM₁₀ Air Quality and PM_{10-2.5} Effect Estimates in Locations of U.S. PM_{10-2.5} Mortality Studies⁺⁺**



2

3 ⁺⁺Studies in Figures 3-2 and 3-3 are a combination of those assessed in the last review and those assessed in the ISA in the current review. Studies in these
 4 figures are ordered by increasing PM₁₀ concentration-equivalent design values.

1 **Figure 3-3. PM₁₀ Air Quality PM_{10-2.5} Effect Estimates in Locations of U.S. PM_{10-2.5} Morbidity Studies**



2
3 * Concentration-equivalent design values were not identified for study periods less than three years. For study periods of less than three years, we averaged the
4 second highest 24-hour PM₁₀ concentrations for each year of the study (i.e., second highest concentration measured by the single monitor in the study area
5 recording the highest such concentration).

1 In addition to the single-city studies included in Figures 3-2 and 3-3 above, multi-city
2 averages of the 3-year expected exceedance concentration-equivalent design values for U.S.
3 multi-city studies were 114 $\mu\text{g}/\text{m}^3$ (for the locations evaluated by Zanobetti and Schwartz, 2009)
4 and 100 $\mu\text{g}/\text{m}^3$ (for the locations evaluated by Peng et al., 2008). As discussed above, the extent
5 to which overall $\text{PM}_{10-2.5}$ effect estimates reported in multi-city studies are associated with the air
6 quality in any particular location is uncertain. However, the ISA also presents Bayes-adjusted
7 single-city effect estimates for each of the cities evaluated by Zanobetti and Schwartz (US EPA,
8 2009a, Figure 6-29), providing the opportunity to consider associations between $\text{PM}_{10-2.5}$ and
9 mortality, and to consider the PM_{10} air quality, in each of the individual cities evaluated in this
10 study.

11 As discussed above, in single-city analyses in the locations evaluated by Zanobetti and
12 Schwartz, $\text{PM}_{10-2.5}$ effect estimates for mortality were generally positive but not statistically
13 significant, and most were similar in magnitude and precision, particularly for cardiovascular
14 mortality, across a wide range of estimated $\text{PM}_{10-2.5}$ concentrations (US EPA, 2009a, Figure 6-
15 29). Three-year PM_{10} expected exceedance concentration-equivalent design values in these
16 cities ranged from 50 $\mu\text{g}/\text{m}^3$ (Davie, FL) to 381 $\mu\text{g}/\text{m}^3$ (El Paso, TX). In most of the cities
17 evaluated (36 of 47), concentration-equivalent design values were below 150 $\mu\text{g}/\text{m}^3$ (Schmidt
18 and Jenkins, 2010). In the six cities where positive and statistically significant $\text{PM}_{10-2.5}$ mortality
19 effect estimates were reported (see above), concentration-equivalent design values were as
20 follows (Schmidt and Jenkins, 2010):

- 21 • Chicago: 113 $\mu\text{g}/\text{m}^3$
- 22 • Pittsburgh: 139 $\mu\text{g}/\text{m}^3$
- 23 • Birmingham: 154 $\mu\text{g}/\text{m}^3$
- 24 • Detroit: 165 $\mu\text{g}/\text{m}^3$
- 25 • St. Louis: 165 $\mu\text{g}/\text{m}^3$
- 26 • Salt Lake City: 283 $\mu\text{g}/\text{m}^3$

27 Therefore, while $\text{PM}_{10-2.5}$ effect estimates in single-city analyses were not statistically significant
28 for most locations evaluated by Zanobetti and Schwartz, including some locations with PM_{10}
29 concentrations well above those allowed by the current 24-hour PM_{10} standard, positive and
30 statistically significant $\text{PM}_{10-2.5}$ effect estimates were reported in two locations (Chicago,
31 Pittsburgh) with concentration-equivalent design values below 150 $\mu\text{g}/\text{m}^3$.

32 In considering $\text{PM}_{10-2.5}$ epidemiologic studies conducted in Canada and elsewhere outside
33 the U.S., we note that we generally do not have access to PM_{10} air quality information beyond
34 that published by the study authors. Many of these studies report PM concentrations averaged
35 across monitors, rather than from the highest monitor in the study area, and/or report only mean

1 or median concentrations. Lin et al. (2002) reported positive and statistically significant
2 associations between PM_{10-2.5} and asthma hospital admissions in children in Toronto (US EPA,
3 2009a; Figures 6-12, 6-15). The authors reported a maximum PM₁₀ concentration measured at a
4 single monitor in the study area of 116 µg/m³, suggesting that the PM₁₀ air quality in Toronto
5 during this study would have been permitted by the current 24-hour PM₁₀ standard. In contrast
6 Middleton et al. (2008), who reported that dust storms in Cyprus were associated with a
7 statistically significant increase in risk of hospitalization for all causes and a non-significant
8 increase in hospitalizations for cardiovascular diseases, reported a maximum 24-hour PM₁₀
9 concentration of 1,371 µg/m³. Thus, the dust storm-associated increases in hospitalizations
10 reported in this study occurred in an area with PM₁₀ concentrations that were likely well above
11 those allowed by the current standard. Other dust storm studies did not report maximum 24-hour
12 PM₁₀ concentrations from individual monitors, though the studies by Chan et al. (2008) and Bell
13 et al. (2008), who reported positive and statistically significant associations between dust storm
14 metrics and cardiovascular-related hospital admissions, reported that 24-hour PM₁₀
15 concentrations, averaged across monitors, exceeded 200 µg/m³. It is likely that peak
16 concentrations measured at individual monitors in these studies were much higher and, therefore,
17 24-hour PM₁₀ concentrations in these study areas were likely above those allowed by the current
18 standard.

19 Summary of Evidence-Based Considerations

20 New evidence supporting an association between PM_{10-2.5} and mortality and morbidity
21 has become available since the last review of the PM NAAQS. The available evidence was
22 judged in the ISA to be *suggestive of a causal relationship* between short-term PM_{10-2.5}
23 exposures and mortality, cardiovascular effects, and respiratory effects while the evidence was
24 judged *inadequate to infer a causal relationship* with long-term PM_{10-2.5} exposures (US EPA,
25 2009a, section 2.3.3; see Table 3-1 above). The evidence supporting a link between short-term
26 thoracic coarse particle exposures and adverse health effects comes primarily from
27 epidemiologic studies, with limited supporting evidence from controlled human exposure studies
28 and, to a lesser extent, animal instillation studies. This evidence includes several recent (i.e.,
29 published since the last review of the PM NAAQS) multi-city epidemiologic studies conducted
30 in the U.S., Canada, and Europe and a small number of recent studies of particles of non-urban
31 origin. In general, epidemiologic studies have reported positive, and in some cases statistically
32 significant, PM_{10-2.5} effect estimates. In the limited number of studies that have evaluated co-
33 pollutant models that include either gaseous pollutants or fine particles, PM_{10-2.5} effect estimates
34 generally remain positive, and in a few cases statistically significant.

35 Positive associations between PM_{10-2.5} and mortality and morbidity have been reported in
36 a number of locations across the U.S. with a wide range of PM_{10-2.5} and PM₁₀ concentrations.

1 Among single-city analyses, PM_{10-2.5} effect estimates were positive and statistically significant in
2 a few U.S. cities and at least one Canadian city with ambient PM₁₀ concentrations that would be
3 allowed by the current 24-hour PM₁₀ standard. However, most PM_{10-2.5} effect estimates, even
4 those reported in locations with PM₁₀ concentrations above the concentrations allowed by the
5 current standard, were not statistically significant.

6 **3.2.2 Staff Conclusions on Adequacy of Current PM₁₀ Standard**

7 In light of the available PM_{10-2.5} health evidence and the PM₁₀ air quality concentrations
8 in study locations, as discussed above, we revisit the overarching question: Does the currently
9 available scientific evidence, as reflected in the ISA, and air quality information support or call
10 into question the adequacy of the protection afforded by the current 24-hour PM₁₀ standard
11 against effects associated with exposures to thoracic coarse particles?

12 In considering the evidence and information as they relate to the adequacy of the current
13 24-hour PM₁₀ standard we note that, as discussed above, this standard is meant to protect the
14 public health against effects associated with short-term exposures to PM_{10-2.5}. In the last review,
15 it was judged appropriate to maintain such a standard given the “growing body of evidence
16 suggesting causal associations between short-term exposure to thoracic coarse particles and
17 morbidity effects, such as respiratory symptoms and hospital admissions for respiratory diseases,
18 and possibly mortality” (71 FR 61185, October 17, 2006). Given the expanded body of evidence
19 available in the current review, discussed in detail in the ISA (US EPA, 2009a, Chapter 6) and
20 summarized above, we conclude that the evidence continues to support the appropriateness of a
21 standard to protect the public health against effects associated with short-term exposures to
22 PM_{10-2.5}. In addition, when considering the evidence for associations with PM_{10-2.5} from different
23 types of sources and in different locations (e.g., urban/industrial as well as windblown dust of
24 non-urban origin), we conclude that it remains appropriate to provide some measure of
25 protection against exposures to all thoracic coarse particles.

26 In considering the evidence, we note that a decision on the adequacy of the public health
27 protection provided by the current PM₁₀ standard will be a public health policy judgment in
28 which the Administrator weighs that evidence and its inherent uncertainties. Therefore,
29 depending on the emphasis placed on different aspects of the evidence and uncertainties,
30 consideration of different conclusions on adequacy could be supported.

31 For example, one approach to considering the evidence and its associated uncertainties
32 would be to place emphasis on the following:

- 33 • Several multi-city epidemiologic studies conducted in the U.S., Canada, and Europe, as
34 well as a number of single-city studies, have reported generally positive, and in some
35 cases statistically significant, associations between short-term PM_{10-2.5} concentrations and

1 adverse health endpoints including mortality and cardiovascular- and respiratory-related
2 hospital admissions and emergency department visits.

- 3
- 4 • A number of single-city analyses (including the majority of the Bayes-adjusted single-
5 city analyses in locations evaluated by Zanobetti and Schwartz (2009)), using different
6 approaches to estimate ambient $PM_{10-2.5}$ concentrations, have reported positive $PM_{10-2.5}$
7 effect estimates in locations with 24-hour PM_{10} concentrations that are allowed by the
8 current PM_{10} standard. In a few cases, these $PM_{10-2.5}$ effect estimates were statistically
9 significant.
- 10
- 11 • While limited in number, studies that have evaluated co-pollutant models have generally
12 reported that $PM_{10-2.5}$ effect estimates remain positive, and in a few cases statistically
13 significant, when these models include gaseous pollutants or fine particles.
- 14
- 15 • Support for the plausibility of the associations reported in epidemiologic studies is
16 provided by a limited number of controlled human exposure studies reporting that short-
17 term (i.e., 2-hour) exposures to $PM_{10-2.5}$ decrease heart rate variability and increase
18 markers of pulmonary inflammation.
- 19

20 Such an approach to considering the evidence would place substantial weight on the generally
21 positive $PM_{10-2.5}$ effect estimates that have been reported for mortality and morbidity, even those
22 effect estimates that are not statistically significant. This could be judged appropriate given that
23 consistent results have been reported across multiple studies using different approaches to
24 estimate ambient $PM_{10-2.5}$ concentrations and that exposure measurement error, which is likely to
25 be larger for $PM_{10-2.5}$ than for $PM_{2.5}$, tends to bias the results of epidemiologic studies toward the
26 null hypothesis, making it less likely that associations will be detected. To the extent that a
27 decision on the adequacy of the current 24-hour PM_{10} standard were to place emphasis on the
28 considerations noted above, it could be judged that the current 24-hour PM_{10} standard does not
29 provide adequate public health protection and that it should be revised in order to increase
30 protection against effects associated with short-term exposures to thoracic coarse particles.

31 Another approach to considering the evidence and its uncertainties would be to place
32 emphasis on the following:

- 33 • While most of $PM_{10-2.5}$ effect estimates reported for mortality and morbidity were
34 positive, many were not statistically significant, even in single-pollutant models. This
35 includes effect estimates reported in study locations with PM_{10} concentrations well-
36 above those allowed by the current 24-hour PM_{10} standard.
- 37 • The number of epidemiologic studies that have employed co-pollutant models to
38 address the potential for confounding, particularly by $PM_{2.5}$, remains limited.
39 Therefore, the extent to which $PM_{10-2.5}$ itself, rather than one or more co-pollutants,
40 contributes to reported health effects remains uncertain.

- 1 • Only a limited number of experimental studies provide support for the associations
2 reported in epidemiologic studies, resulting in further uncertainty regarding the
3 plausibility of the associations between PM_{10-2.5} and mortality and morbidity reported
4 in epidemiologic studies.
- 5 • Limitations in PM_{10-2.5} monitoring and the different approaches used to estimate
6 PM_{10-2.5} concentrations across epidemiologic studies result in uncertainty in the
7 ambient PM_{10-2.5} concentrations at which the reported effects occur.
- 8 • The chemical and biological composition of PM_{10-2.5}, and the effects associated with
9 the various components, remains uncertain. Without more information on the chemical
10 speciation of PM_{10-2.5}, the apparent variability in associations across locations is
11 difficult to characterize.

12 To the extent that a decision on the adequacy of the current 24-hour PM₁₀ standard were to place
13 emphasis on the considerations noted above, it could be judged that, while it remains appropriate
14 to maintain a standard to protect against short-term exposures to thoracic coarse particles, the
15 available evidence does not provide a sound basis for concluding that the current 24-hour PM₁₀
16 standard fails to protect public health with an adequate margin of safety. Such an approach to
17 considering the evidence could also note the ISA conclusions that, when considered as a whole,
18 the available evidence is “suggestive” of a causal relationship between short-term PM_{10-2.5}
19 exposures and mortality, cardiovascular effects, and respiratory effects. These weight-of-
20 evidence conclusions contrast with those for the relationships between PM_{2.5} exposures and
21 adverse health effects, which were judged in the ISA to be either “causal” or “likely causal” for
22 mortality, cardiovascular effects, and respiratory effects. Thus, while this approach to
23 considering the evidence would recognize the positive, and in some cases statistically significant,
24 associations between PM_{10-2.5} and mortality and morbidity by maintaining a standard to protect
25 against exposures to thoracic coarse particles, it would place relatively greater emphasis on the
26 limitations and uncertainties noted above, which tend to complicate the interpretation of that
27 evidence.

28 Given all of the above, we conclude that consideration could be given to either retaining or
29 revising the current 24-hour PM₁₀ standard, depending on the approach taken to considering the
30 available evidence and information. Therefore, we judge that it is appropriate in this Policy
31 Assessment to consider what potential alternative standards, if any, could be supported by the
32 available scientific evidence in order to increase public health protection against exposures to
33 PM_{10-2.5}.

1 **3.3 CONSIDERATION OF POTENTIAL ALTERNATIVE STANDARDS**

2 Staff next considers the following overarching question:

3 **What potential alternative standard(s) could be supported by the currently available**
4 **scientific evidence and air quality information to increase public health protection against**
5 **exposures to PM_{10-2.5}?**

6 In addressing this overarching question, we consider how the currently available
7 scientific evidence and air quality information could inform decisions regarding the basic
8 elements of the NAAQS: indicator (section 3.3.1), averaging time (section 3.3.2), form (section
9 3.3.3), and level (section 3.3.4).

10 **3.3.1 Indicator**

11 As discussed above, PM₁₀ includes both PM_{10-2.5} and PM_{2.5}, with the relative contribution
12 of each to PM₁₀ mass varying across locations (see below). In the most recent review completed
13 in 2006, EPA concluded that the PM₁₀ indicator remained appropriate because a PM₁₀ standard
14 would be expected to provide appropriate protection against effects associated with exposures to
15 PM_{10-2.5}. In particular, a PM₁₀ indicator would be expected to target protection to urban areas,
16 where the evidence of effects from exposure to coarse PM is the strongest (71 FR at 61196). In
17 considering potential alternative standards in the current review, we have considered the
18 following question with regard to indicator:

- 19 • **To what extent does the available evidence and/or air quality information provide**
20 **support for retaining or revising the current PM₁₀ indicator?**

21 In addressing this question, we focus on the following considerations:

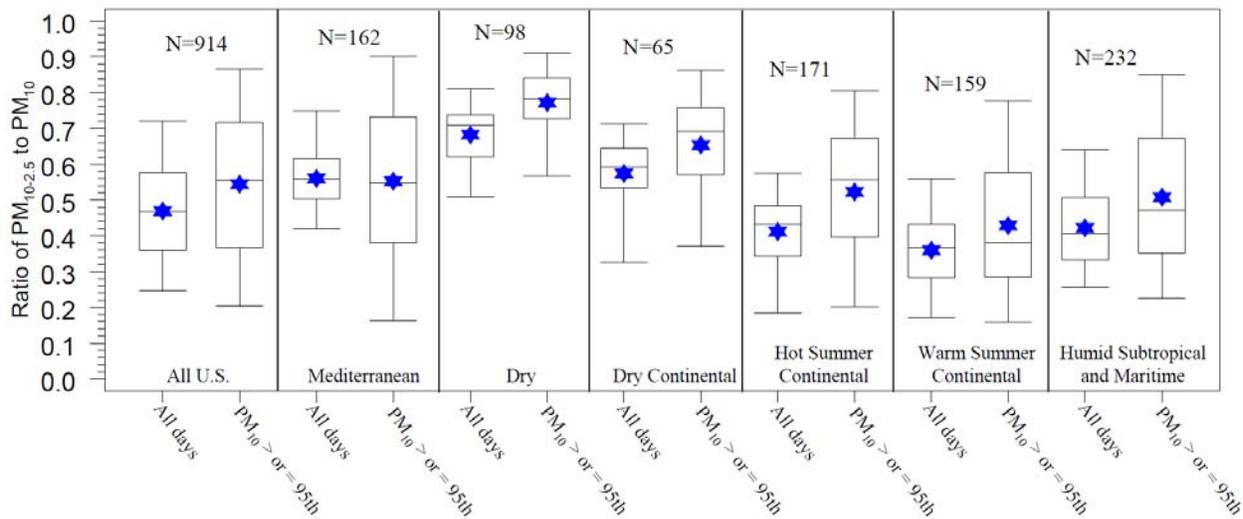
- 22 • The extent to which PM₁₀ is comprised of PM_{10-2.5}
23
24 • The appropriateness of a standard that allows lower PM_{10-2.5} concentrations in areas with
25 higher fine particle concentrations (i.e., urban areas) than areas with lower fine particle
26 concentrations (i.e., rural areas) (see Schmidt, 2005 for urban-rural comparison)
27

28 As an initial matter, we consider the proportion of PM₁₀ mass in different regions of the
29 U.S. that is PM_{10-2.5} (see US EPA, 2009a, section 3.5.1.1; Schmidt and Jenkins, 2010). Schmidt
30 and Jenkins (2010) divided the U.S. into climatic regions using the same approach as used in the
31 multi-city epidemiologic study by Zanobetti and Schwartz (2009) (see above).⁶ Consistent with

⁶The Mediterranean region includes CA, OR, WA. The Dry region includes NM, AZ, NV. The Dry Continental region includes MT, ID, WY, UT, CO. The Hot Summer Continental region includes SD, NE, IA, IL, IN, OH. The Warm Summer Continental region includes ND, MN, WI, MI, PA, NY, CT, RI, MA, VT, NH, ME. The Humid Subtropical and Maritime region includes FL, LA, TX, GA, AL, MS, AR, OK, KS, MO, TN, SC, NC, VA, WV, KY, NJ, DE, DC, MD.

1 the air quality analyses in the ISA (US EPA, 2009a, section 3.5.1.1) and the concentration
 2 estimates of Zanobetti and Schwartz (2009), $PM_{10-2.5}$ concentrations were higher in the western
 3 U.S. than the east, with the highest concentrations in the southwest (data not shown, from
 4 Schmidt and Jenkins, 2010). On average, ratios of $PM_{10-2.5}$ concentrations to PM_{10}
 5 concentrations were also higher in the western U.S. than in the eastern U.S., with the southwest
 6 (i.e., Dry region in Figure 3-4) having the highest ratios. Consistent results were reported in the
 7 ISA analyses of PM air quality (US EPA, 2009a, compare Tables 3-9 and 3-10). While the same
 8 general pattern persisted when the analysis was restricted to days with PM_{10} concentrations at the
 9 high end of the distribution of daily concentrations (i.e., at or above 95th percentile), ratios of
 10 $PM_{10-2.5}$ to PM_{10} on these days tended to be somewhat higher, on average, across most regions of
 11 the U.S. (see right-hand columns in Figure 3-4).

12 **Figure 3-4. Site-Level Ratio of Daily $PM_{10-2.5}$ to Daily PM_{10} ***



13 *Blue stars represent mean concentrations, horizontal lines represent median concentrations, boxes
 14 represent 75% confidence intervals, and error bars represent 95% confidence intervals. N values
 15 equal the number of site years of monitoring data for each region.
 16
 17

18 Thus, on average across the U.S., $PM_{10-2.5}$ comprises a larger portion of PM_{10} on days with
 19 relatively high PM_{10} concentrations than on days with more typical PM_{10} concentrations. This
 20 suggests that elevated PM_{10} concentrations across much of the U.S. are due in large part to
 21 elevations in $PM_{10-2.5}$ mass. Given this, we conclude that a PM_{10} standard would be most
 22 effective at limiting $PM_{10-2.5}$ concentrations if it focuses on the upper end of the distribution of
 23 daily PM_{10} concentrations. This issue is discussed further below in the context of the standard
 24 form (section 3.3.3).

25 Given the above conclusion, in further considering the issue of indicator we note that
 26 most of the evidence for positive associations between $PM_{10-2.5}$ and morbidity and mortality,

1 particularly evidence for these associations at relatively low concentrations of PM_{10-2.5}, continues
2 to come from studies conducted in locations where the PM_{10-2.5} is expected to be largely of urban
3 origin. While some studies have reported positive associations between relatively high
4 concentrations of particles of non-urban origin (i.e., crustal material from windblown dust in
5 non-urban areas, see above) and mortality and morbidity, we note that the extent to which these
6 associations would remain at the lower particle concentrations more typical of U.S. and
7 Canadian urban study locations remains uncertain.⁷ Given these considerations, and given the
8 increased potential for coarse particles in urban areas to become contaminated by toxic
9 components of fine particles from urban/industrial sources (US EPA, 2004), we conclude that it
10 remains appropriate to maintain a standard that allows lower ambient thoracic coarse particle
11 concentrations in urban areas than in non-urban areas.

12 Given this conclusion, we note that it would be reasonable to consider an indicator that
13 targets control on areas with the types of ambient mixes generally present in urban areas. Such
14 an indicator would focus control on areas with ambient mixes known with greater certainty to be
15 associated with adverse health effects and, therefore, would provide public health benefits with
16 the greatest degree of certainty. As noted in the last review of the PM NAAQS, a PM₁₀ standard
17 would allow lower concentrations of PM_{10-2.5} in areas with higher fine particle concentrations,
18 which tend to be urban locations, than areas with lower fine particle concentrations, which tend
19 to be rural locations (Schmidt et al., 2005). Therefore, we reach the conclusion that a PM₁₀
20 indicator would appropriately target protection to those locations where the evidence is strongest
21 for associations between adverse health effects and exposures to thoracic coarse particles. In
22 contrast, we note that a PM_{10-2.5} indicator, for a standard set at a single unvarying level, would
23 not achieve this targeting, given that allowable thoracic coarse particle concentrations would be
24 the same regardless of the location or the likely sources of PM. Therefore, given the currently
25 available evidence, one possible result of using a PM_{10-2.5} indicator would be a standard that is
26 overprotective in rural areas and/or underprotective in urban areas.

27 In addition, while we note that administrative feasibility is not an appropriate basis for
28 informing decisions on the NAAQS (see above), we also note that PM_{10-2.5} concentrations are not
29 routinely measured and reported at present (US EPA, 2009a, section 3.5.1.1). In the last review
30 of the PM NAAQS, EPA required monitoring of PM_{10-2.5} mass and expects approximately 80
31 stations to be reporting mass concentrations by January 1, 2011, as part of the National Core

⁷Other than the dust storm studies, we note that the study in Coachella Valley by Ostro et al. (2003) reported statistically significant associations in a location where coarse particles are expected to be largely due to windblown dust. Specifically, we note the CASAC conclusion in the last review that “studies from Ostro *et al.* showed significant adverse health effects, primarily involving exposures to coarse-mode particles arising from crustal sources” (Henderson, 2005). In considering this study, we also note the relatively high PM₁₀ concentrations in the study area (see Figure 3-2 above).

1 (NCore) network (<http://www.epa.gov/ttnamti1/ncore/index.html>). However, data from those
2 monitors are not yet available for locations and time periods of PM_{10-2.5} health studies.

3 Given all of the above considerations, we conclude that the available evidence supports
4 consideration in the current review of a PM₁₀ indicator for a standard that protects against
5 exposures to thoracic coarse particles. We further conclude that consideration of alternative
6 indicators (e.g., PM_{10-2.5}) in future reviews is desirable and could be informed by additional
7 research, as described below (section 3.4).

8 3.3.2 Averaging Time

9 Based primarily on epidemiologic studies that reported positive associations between
10 short-term (24-hour) PM_{10-2.5} concentrations and mortality and morbidity, the Administrator
11 concluded in the last review that the available evidence supported a 24-hour averaging time for a
12 standard intended to control thoracic coarse particles. In contrast, given the relative lack of
13 studies supporting a link between long-term exposures to thoracic coarse particles and morbidity
14 or mortality (US EPA, 2004a, Chapter 9), the Administrator further concluded that an annual
15 coarse particle standard was not warranted at that time (71 FR 61198-61199).

16 In the current review, we consider the extent to which the available evidence provides
17 information relevant for decisions on averaging time by considering the following question:

- 18 • **To what extent does the available evidence continue to support a 24-hour averaging**
19 **time for a standard meant to protect against effects associated with exposures to**
20 **PM_{10-2.5}?**

21 With regard to this question, we note the conclusions from the ISA regarding the weight
22 of evidence for short-term and long-term PM_{10-2.5} exposures as well as the studies on which those
23 conclusions are based. Specifically, as discussed above (see Table 3-1 above), the ISA
24 concludes that the existing evidence is *suggestive of a causal relationship* between short-term
25 PM_{10-2.5} exposures and mortality, cardiovascular effects, and respiratory effects (ISA, section
26 2.3.3). This conclusion is based largely on epidemiologic studies which have primarily
27 evaluated associations between 24-hour PM_{10-2.5} concentrations and morbidity and mortality
28 (e.g., see ISA, Figure 2-3), though a small number of controlled human exposure studies have
29 reported effects following shorter exposures (i.e., 2-hours) to PM_{10-2.5} (e.g., see ISA, sections
30 6.2.1.2, 6.3.3.2). In contrast, with respect to long-term exposures, the ISA concludes that
31 available evidence is *inadequate to infer a causal relationship* with all health outcomes
32 evaluated (US EPA, 2009a, section 2.3). Specifically, the ISA states, “To date, a sufficient
33 amount of evidence does not exist in order to draw conclusions regarding the health effects and
34 outcomes associated with long-term exposure to PM_{10-2.5}” (US EPA, 2009a, section 2.3.4; see
35 Table 3-1 above).

1 In considering these weight of evidence determinations in the ISA, we conclude that, at a
2 minimum, they suggest the importance of maintaining a standard that protects against short-term
3 exposures to thoracic coarse particles. Given that the majority of the evidence supporting the
4 link between short-term PM_{10-2.5} and morbidity and mortality is based on 24-hour average
5 thoracic coarse particle concentrations, we conclude that the evidence available in this review
6 continues to support consideration of a 24-hour averaging time for a PM₁₀ standard meant to
7 protect against effects associated with short-term exposures to PM_{10-2.5}. We further conclude that
8 the available evidence does not support consideration of an annual thoracic coarse particle
9 standard at this time. In reaching this conclusion, we also note that, to the extent a short-term
10 standard requires areas to reduce their 24-hour ambient particle concentrations, long-term
11 concentrations would also be expected to decrease. Therefore, a 24-hour standard meant to
12 protect against short-term exposures to thoracic coarse particles would also be expected to
13 provide some protection against any potential effects of long-term exposures at ambient
14 concentrations.

15 **3.3.3 Form**

16 The “form” of a standard defines the air quality statistic that is to be compared to the
17 level of the standard in determining whether an area attains that standard. In identifying a single
18 statistic for the form, we note that although future air quality improvement strategies in any
19 particular area are not yet defined, most such strategies are likely to affect a broad distribution of
20 PM air quality concentrations in an area. Therefore, although the form of the standard defines a
21 single statistic, any resulting reductions in health risks are likely to occur across a wide range of
22 concentrations.

23 As discussed above, in the last review the Administrator retained the one-expected
24 exceedance form of the primary 24-hour PM₁₀ standard. This decision was linked to the overall
25 conclusion that “the level of protection from coarse particles provided by the current 24-hour
26 PM₁₀ standard remains requisite to protect public health with an adequate margin of safety” (71
27 FR 61202). Because revising either the level or the form of the standard would have altered the
28 protection provided, it was concluded that such changes “would not be appropriate based on the
29 scientific evidence available at this time” (71 FR 21202). Therefore, the decision in the last
30 review to retain the one-expected-exceedance form was part of the broader decision that the
31 existing 24-hour standard provided requisite public health protection.

32 In the current review, we are also considering the form of the standard within the context
33 of the overall decision on whether, and if so how, to revise the current 24-hour PM₁₀ standard.
34 Given the conclusions above regarding the appropriate indicator and averaging time for

1 consideration for potential alternative standards, we consider potential alternative forms for a 24-
2 hour PM₁₀ standard. To frame our consideration of this issue, we pose the following question:

- 3 • **To what extent does available evidence and information support consideration of an**
4 **alternative form for a 24-hour PM₁₀ standard?**

5 Although the selection of a specific form must be made within the context of decisions on
6 the other elements of the standard, EPA generally favors concentration-based forms for short-
7 term standards. In 1997 EPA established a 98th percentile form for the 24-hour PM_{2.5} standard
8 and in 2010 EPA established a 98th percentile form for the 1-hour NO₂ standard (62 FR 38671;
9 75 FR 6474).⁸ In making these decisions, EPA noted that, as compared to an exceedance-based
10 form, such as the form of the current PM₁₀ standard, a concentration-based form is more
11 reflective of the health risks posed by elevated pollutant concentrations because it gives
12 proportionally greater weight to days when concentrations are well above the level of the
13 standard than to days when the concentrations are just above the level of the standard. In
14 addition, when averaged over three years, a concentration-based form can provide a stable
15 regulatory target, facilitating the development of stable implementation programs.

16 In further considering the issue of form in the current review of the 24-hour PM₁₀
17 standard, we note that a concentration-based form better compensates for missing data and less
18 than-daily monitoring than an exceedance-based form. This is a particularly important
19 consideration in the case of PM₁₀ because, depending on ambient PM₁₀ concentrations, the
20 frequency of PM₁₀ monitoring differs across locations (US EPA, 1987). With a concentration-
21 based form, such as a 98th percentile form, attainment status would be determined based on PM₁₀
22 concentrations from the same part of the annual distribution of 24-hour PM₁₀ concentrations,
23 despite the potential for differences in the frequency of PM₁₀ monitoring across locations. In
24 contrast, with the current one-expected-exceedance form, differences in monitoring frequency
25 can result in attainment status in some locations being based on estimates of the number of days
26 per year that the standard level is expected to be exceeded (i.e., in locations with less-than-daily
27 monitoring), rather than actual counts of the number of exceedances (see Appendix K to Part 50
28 of the CFR). The extent to which these estimates reflect actual exceedances becomes more
29 uncertain as the monitoring frequency decreases.

30 In considering specific concentration-based forms for a 24-hour PM₁₀ standard, we note
31 that, as discussed above, in past reviews (62 FR 38671; 75 FR 6474) EPA has judged it
32 appropriate to define a standard that provides a balance between adequately limiting the

⁸In the recently completed review of the SO₂ primary NAAQS, a 99th percentile form was adopted. In the case of SO₂, there was specific evidence that a 99th percentile form would be more effective than a 98th percentile form for controlling 5-minute peak SO₂ concentrations, which were of concern because controlled human exposure studies have linked 5-minute SO₂ exposures to respiratory effects (See 75 FR at 35541 (June 22, 2010)).

1 occurrence of peak pollutant concentrations and providing a stable regulatory target for
2 implementation programs. For short-term standards, this balance has been achieved by setting
3 the form of the standard at the upper end of the distribution of pollutant concentrations, but not
4 using the most extreme statistics, and by averaging pollutant concentrations over three years.
5 We conclude that similar considerations are relevant in the current review for a decision on the
6 form of a PM₁₀ standard. Specifically, given that PM_{10-2.5} is likely to make a larger contribution
7 to PM₁₀ mass on days with relatively high PM₁₀ concentrations than on days with more typical
8 PM₁₀ concentrations (see above), we conclude that it is appropriate to set the standard form at the
9 upper end of the annual distribution of 24-hour PM₁₀ concentrations. In addition, given the
10 potential for local sources to have important impacts on monitored PM₁₀ concentrations (US
11 EPA, 2009a, section 2.1.1.2), we also conclude that it is desirable to select a form that promotes
12 stability of local implementation programs.⁹

13 Based on all of the above considerations, we conclude that, to the extent it is judged
14 appropriate to revise the current 24-hour PM₁₀ standard, it would be appropriate to consider a
15 range of potential alternative standard levels based on 98th percentile PM₁₀ concentrations. In
16 drawing this conclusion we note that, because of the current expected-exceedance-based form
17 and the differences in monitoring frequency across locations, the stringency of a standard with a
18 98th percentile form, relative to the stringency of the current standard, will differ across locations.
19 One consequence of this is that, compared to the current standard, a standard with a 98th
20 percentile form (or any other concentration-based form) could be more stringent than the current
21 standard in some locations and less stringent in other locations, depending on the level of the
22 standard. This issue is discussed in more detail below within the context of specific potential
23 alternative standard levels.

24 **3.3.4 Level**

25 As noted above, to the extent it is judged in the current review that the 24-hour PM₁₀
26 standard does not provide adequate public health protection against exposures to thoracic coarse
27 particles, potential alternative standard levels could be considered. Given the conclusions
28 described above for indicator, averaging time, and form, we conclude that it would be
29 appropriate to consider potential alternative levels for a 24-hour PM₁₀ standard with a 98th

⁹We note that in the most recent review of the NO₂ primary NAAQS, a 98th percentile form was adopted, rather than a 99th percentile form, due to the potential for “instability in the higher percentile concentrations” near local sources (i.e., mobile sources on major roads) (75 FR 6493). See also, ATA III, 283 F. 3d at 374-75 (upholding 98th percentile form since “otherwise States would have to design their pollution control programs around single high exposure events that may be due to unusual meteorological conditions alone, rendering the programs less stable – and hence, we assume, less effective – than programs designed to address longer-term average conditions.”).

1 percentile form. To inform our consideration of this issue, we have considered the following
2 question:

- 3 • **To what extent does available evidence and air quality information support**
4 **consideration of alternative standard levels for a 24-hour PM₁₀ standard with a 98th**
5 **percentile form?**

6 Evidence-based Considerations

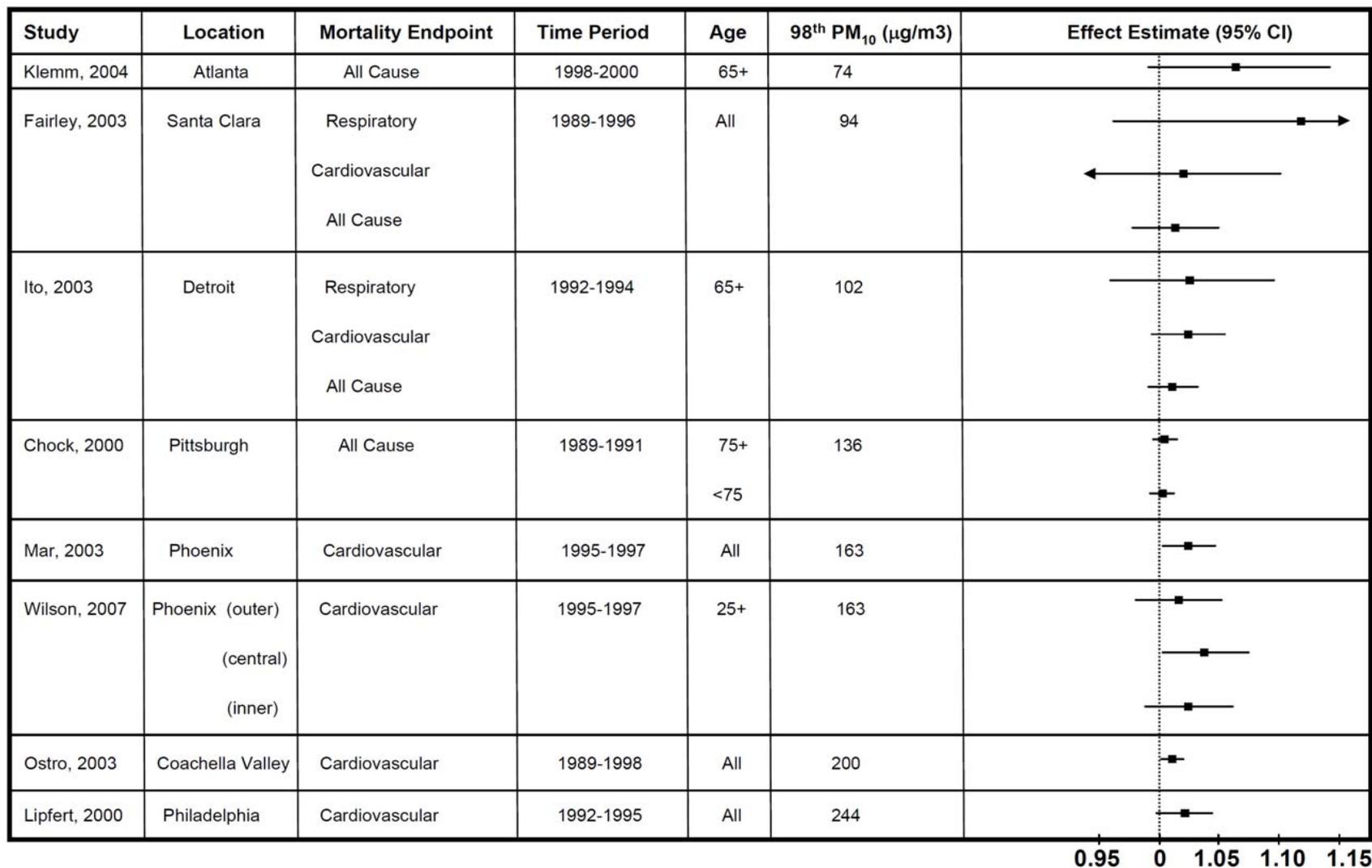
7 In considering the evidence as it relates to potential alternative standard levels for
8 consideration, we first consider the relative weight to place on specific epidemiologic studies,
9 including the weight to place on the uncertainties associated with those studies. We have
10 considered several factors in placing weight on specific epidemiologic studies including the
11 extent to which studies report statistically significant associations with PM_{10-2.5} and the extent to
12 which the reported associations are robust to co-pollutant confounding.

13 In addition, we consider the extent to which associations with PM_{10-2.5} can be linked to
14 the air quality in a specific location. With regard to this, we place greatest weight on information
15 from single-city analyses. Although, as discussed above, multi-city studies have advantages in
16 terms of power to detect associations and geographic coverage, the extent to which effects
17 reported in multi-city studies are associated with the short-term air quality in any particular
18 location is highly uncertain, especially when considering short-term concentrations at the upper
19 end of the distribution of daily concentrations for pollutants with relatively heterogeneous spatial
20 distributions such as PM_{10-2.5} and PM₁₀ (US EPA, 2009a, section 2.1.1.2). In contrast, single-city
21 studies are more limited in terms of power and geographic coverage but the link between
22 reported health effects and the air quality in a given city is more straightforward to establish (US
23 EPA, 2009a, section 2.1.1.2). Given this, in considering PM₁₀ concentrations in locations of
24 epidemiologic studies, we place the most weight on single-city studies (Figures 3-2 and 3-3) and
25 single-city analyses of the locations evaluated in the multi-city study by Zanobetti and Schwartz
26 (US EPA, 2009a, Figure 6-29).

27 In considering PM air quality in study locations, we also note that the available evidence
28 does not support the existence of thresholds, or lowest-observed-effects levels, in terms of 24-
29 hour average concentrations (US EPA, 2009a, section 2.4.3). In the absence of an apparent
30 threshold, for purposes of identifying a range of standard levels potentially supported by the
31 health evidence, we focus on the range of PM₁₀ concentrations that have been measured in
32 locations where U.S. epidemiologic studies have reported associations with PM_{10-2.5} (see U.S.
33 EPA, 2009a, Figures 6-1 to 6-30 for studies). In characterizing PM₁₀ air quality in PM_{10-2.5} study
34 locations, we have used EPA's AQS to identify the highest 98th percentile 24-hour PM₁₀
35 concentrations for each year in each study location (i.e., from the monitor in the study area
36 recording highest 98th percentile concentration), as described in Schmidt and Jenkins (2010).

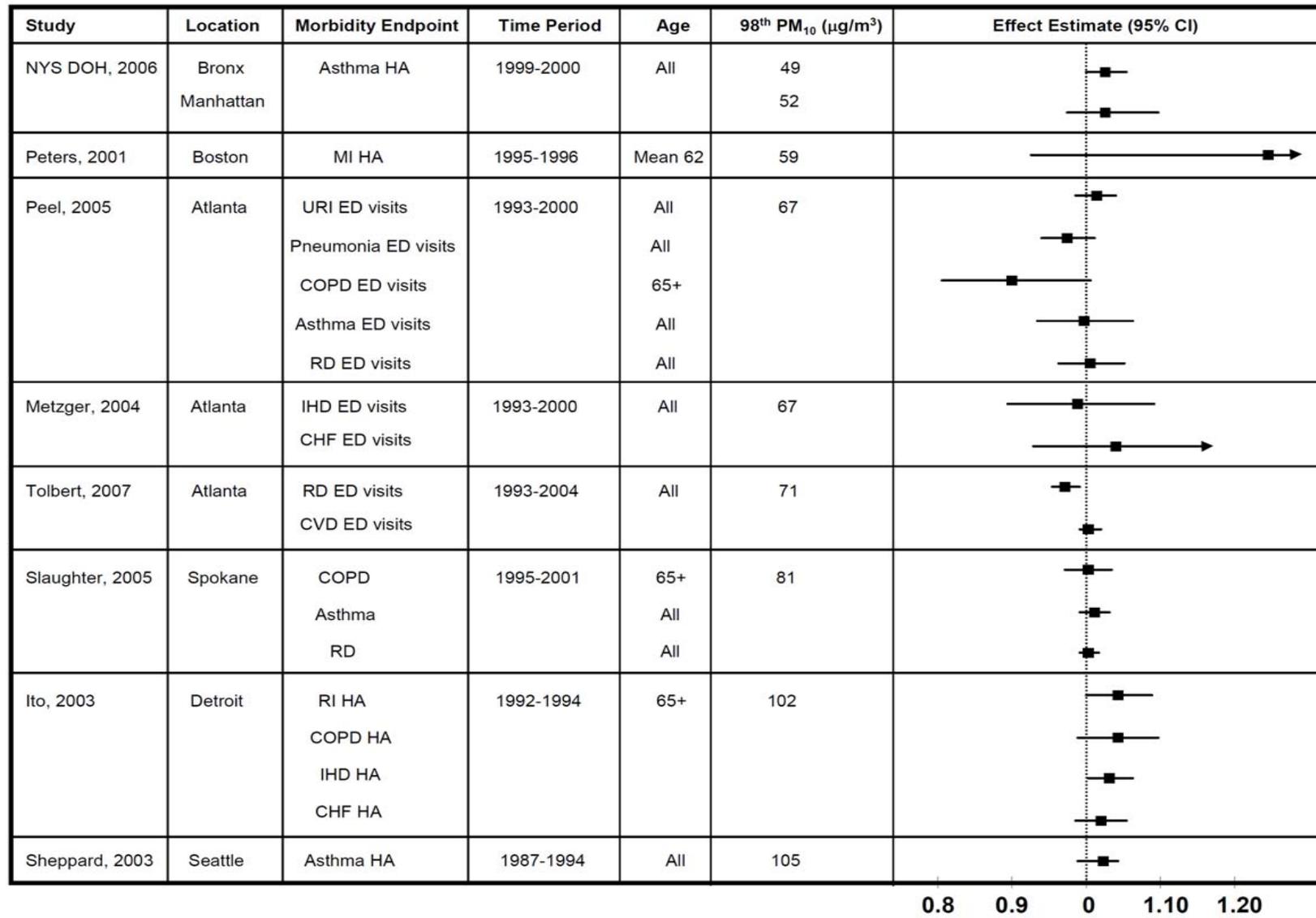
- 1 The 98th percentile concentrations from each study year were averaged together and these
- 2 averages are presented below in Figures 3-5 (PM_{10-2.5} mortality studies) and 3-6 (PM_{10-2.5}
- 3 morbidity studies) for locations of single-city studies.

1 **Figure 3-5. 98th Percentile PM₁₀ Concentrations in Locations of U.S. PM_{10-2.5} Mortality Studies***



2
3
4 *Studies in Figures 3-5 and 3-6 are a combination of those assessed in the last review and those assessed in the ISA in the current review. Studies in these figures are ordered by increasing 98th percentile PM₁₀ concentrations.

1 **Figure 3-6. 98th Percentile PM₁₀ Concentrations in Locations of U.S. PM_{10-2.5} Morbidity Studies**



2

1 In addition to the single-city study locations in Figures 3-5 and 3-6, 98th percentile PM₁₀
2 concentrations averaged across the study locations evaluated in the multi-city studies by
3 Zanobetti and Schwartz (2009) and Peng et al. (2008) were 78 µg/m³ and 68 µg/m³, respectively.
4 Multi-city effect estimates remained positive, and in some cases (i.e., Zanobetti and Schwartz for
5 all-cause and respiratory mortality) statistically significant, in co-pollutant models that included
6 fine particles.

7 Bayes-adjusted single-city effect estimates for the 47 cities evaluated by Zanobetti and
8 Schwartz (US EPA, 2009a, Figure 6-29), which were generally positive but not statistically
9 significant, can provide some additional insight into the PM₁₀ concentrations in specific locations
10 where associations between PM_{10-2.5} and mortality have been reported. The 98th percentile PM₁₀
11 concentrations in these cities ranged from 39 µg/m³ (Davie, FL) to 195 µg/m³ (El Paso, TX), and
12 in the 6 cities where positive and statistically significant PM_{10-2.5} mortality effect estimates were
13 reported, 98th percentile PM₁₀ concentrations were as follows (Schmidt and Jenkins, 2010):

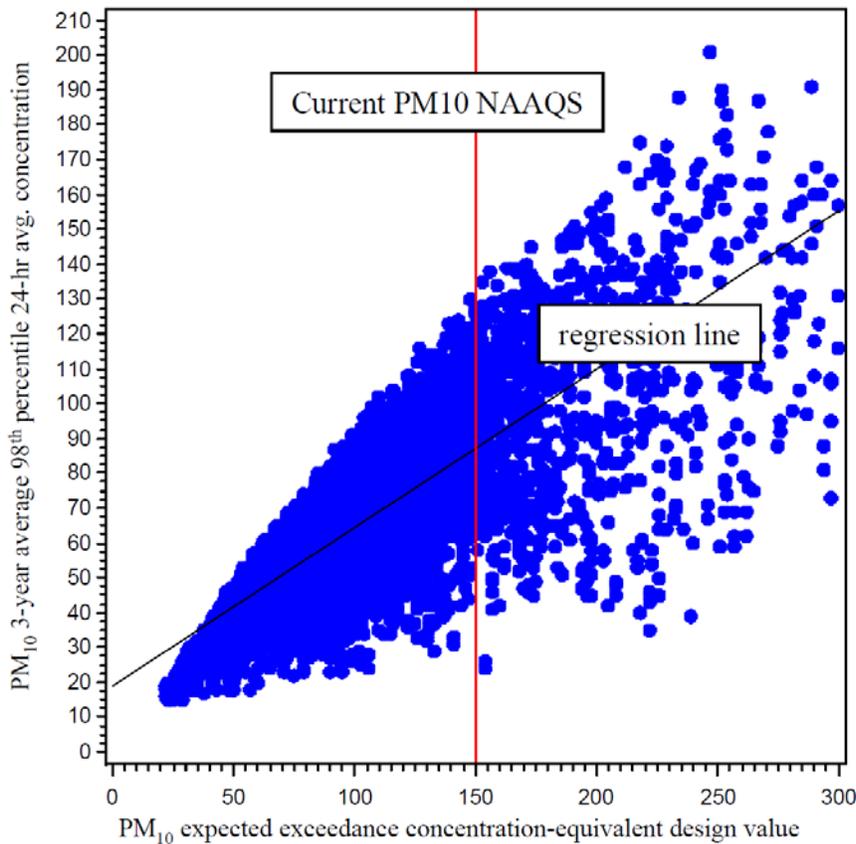
- 14 • Chicago: 92 µg/m³
- 15 • Salt Lake City: 98 µg/m³
- 16 • Detroit: 105 µg/m³
- 17 • Pittsburgh: 110 µg/m³
- 18 • Birmingham: 122 µg/m³
- 19 • St. Louis: 138 µg/m³

20 Thus, in the single-city mortality studies in Figure 3-5 above, as well as the Bayes-adjusted
21 single-city analyses of the locations evaluated by Zanobetti and Schwartz, positive and
22 statistically significant PM_{10-2.5} effect estimates were reported in some locations with 98th
23 percentile PM₁₀ concentrations as low as 92 µg/m³ (i.e., locations evaluated by Mar et al., 2003;
24 Ostro et al., 2003; Wilson et al., 2007; and the cities listed above, US EPA 2009a, Figure 6-29).
25 Similarly, among the U.S. morbidity studies, Ito (2003) reported a positive and statistically
26 significant PM_{10-2.5} effect estimate for hospital admissions for ischemic heart disease in Detroit,
27 where the 98th percentile PM₁₀ concentration was well above 92 µg/m³. PM_{10-2.5} effect estimates
28 in this study remained positive, and in some cases statistically significant, in co-pollutant models
29 with gaseous pollutants (US EPA, 2009a, Figures 6-5 and 6-15). Other morbidity studies
30 generally did not report statistically significant PM_{10-2.5} effect estimates, though effect estimates
31 were marginally significant (i.e., the lower confidence limit was 0) for asthma hospital
32 admissions in the Bronx (NYSDOH, 2006) and for respiratory infection-related admissions in
33 Detroit (Ito et al., 2003).

1 Air Quality-based Considerations

2 In addition to the evidence-based considerations described above, we have used EPA's
3 AQS to estimate the level of a 24-hour PM₁₀ standard with a 98th percentile form that would
4 approximate the degree of protection, on average across the country, provided by the current 24-
5 hour PM₁₀ standard with its one-expected-exceedance form. Based on regression of 98th
6 percentile PM₁₀ concentrations onto one-expected-exceedance concentration equivalent design
7 values, a 98th percentile PM₁₀ concentration of 87 µg/m³ is, on average, equivalent to a one-
8 expected-exceedance design value of 150 µg/m³ (see Figure 3-7 below and Schmidt and Jenkins,
9 2010). However, as indicated in Figure 3-7, the range of equivalent concentrations varies
10 considerably across monitoring sites (95% confidence interval ranges from 63 to 111 µg/m³) (see
11 Schmidt and Jenkins, 2010). As a consequence, while we note that a 98th percentile 24-hour
12 PM₁₀ standard with a level of 87 µg/m³ would be expected to provide public health protection
13 that is, on average across the U.S., equivalent to the protection provided by the current standard,
14 in some locations such a 98th percentile standard could be more protective than the current
15 standard while in other locations it could be less protective than the current standard.

1 **Figure 3-7. Composite 3-year PM₁₀ 98th percentile 24-hour average concentration versus**
2 **the PM₁₀ expected exceedance concentration-equivalent design value**



3
4

5 In considering this issue in the specific locations where we have evidence for associations
6 between PM_{10-2.5} and mortality or morbidity, we note that, as described above, positive and
7 statistically significant PM_{10-2.5} effect estimates have been reported in the following U.S.
8 locations:

- 9
- Coachella Valley (Ostro et al., 2003)
 - 10 • Detroit (Ito et al., 2003)
 - 11 • Phoenix (Mar et al., 2003; Wilson et al., 2007)
 - 12 • Chicago, Salt Lake City, Detroit, Pittsburgh, Birmingham, St. Louis (US EPA, 2009a,
13 Figure 6-29)

14 In considering the PM₁₀ air quality, we note that 98th percentile PM₁₀ concentrations were above
15 87 µg/m³ in all of these cities, even those with PM₁₀ concentration-equivalent design values
16 below 150 µg/m³ (Detroit, Chicago, Pittsburgh) (see section 3.2.1 above for PM₁₀ concentration-
17 equivalent design values).

1 To provide a broader perspective, we also used EPA's AQS to compare the number of
2 U.S. counties with PM₁₀ one-expected-exceedance concentration-equivalent design values above
3 150 µg/m³ to the number with 98th percentile PM₁₀ concentrations above 87 µg/m³ (for the years
4 2006 to 2008) (see Schmidt and Jenkins, 2010). The results in Table 3-2 below indicate the
5 following:

- 6 • In the eastern U.S., the number of counties with PM₁₀ one-expected-exceedance
7 concentration-equivalent design values above 150 µg/m³ is the same as that with 98th
8 percentile PM₁₀ concentrations above 87 µg/m³ (i.e., none in the northeast and three in the
9 southeast).
- 10
- 11 • Some counties in the western U.S. (i.e., ten counties total in the southwest, northwest, and
12 Southern California) had PM₁₀ one-expected-exceedance concentration-equivalent design
13 values above 150 µg/m³ but had 98th percentile PM₁₀ concentrations below 87 µg/m³.
- 14
- 15 • A small number of counties in the midwest (i.e., four counties total in the industrial midwest
16 and upper midwest) had one-expected-exceedance concentration-equivalent design values
17 below 150 µg/m³ but had 98th percentile PM₁₀ concentrations above 87 µg/m³.

1

2 **Table 3-2. Counties with PM₁₀ Concentration-Equivalent Design Values > 150 µg/m³ and 98th percentile PM₁₀ concentrations**
 3 **> 87 µg/m³**

Region >	All U.S.	Northeast	Southeast	Industrial Midwest	Upper Midwest	Southwest	Northwest	Southern California	
Total # of counties >	318	40	60	57	35	24	79	18	
Total population >	122,582	15,753	27,009	23,646	5,143	11,037	16,188	22,695	
	Statistic	Numbers of counties, populations, and percentages of total							
3-year average 98 th percentile > 87 µg/m ³	# counties	38	0	3	3	3	12	8	10
	population	22,450	0	4,159	1,789	710	6,106	1,366	8,220
	% # counties	12%	0%	5%	5%	9%	50%	10%	56%
	% population	18%	0%	15%	8%	14%	55%	8%	36%
3-Year Expected Exceedance Equivalent Design Value > 150 µg/m ³	# counties	45	0	3	1	1	14	15	11
	population	32,169	0	4,159	348	28	7,675	2,075	17,724
	% # counties	14%	0%	5%	2%	3%	58%	19%	61%
	% population	26%	0%	15%	1%	1%	70%	13%	78%

4

1 Integration of evidence- and air quality-based considerations

2 In considering the evidence and air quality information within the context of identifying
3 potential alternative standard levels for consideration, we note the following:

- 4 • Although a 98th percentile 24-hour PM₁₀ concentration of 87 µg/m³ is equivalent, on
5 average across the country, to a PM₁₀ concentration-equivalent design value of 150
6 µg/m³, the 98th percentile standard could be more stringent than the current standard in
7 some locations (e.g., in the industrial and upper midwest) and less stringent than the
8 current standard in other locations (e.g., northwest, southwest, and southern California).
- 9 • A 98th percentile 24-hour PM₁₀ standard with a level at or below 87 µg/m³ would be
10 expected to maintain PM₁₀ and PM_{10-2.5} concentrations below those present in U.S. study
11 locations where PM_{10-2.5} effect estimates have been reported to be positive and
12 statistically significant.
- 13 • While studies generally reported positive PM_{10-2.5} effect estimates in locations with 98th
14 percentile PM₁₀ concentrations below 87 µg/m³, these effect estimates were not
15 statistically significant.

16 To the extent the above considerations are emphasized, the approach to considering
17 potential alternative standard levels would place more weight on potential alternative standard
18 levels for which the evidence is strongest (e.g., multiple single-city studies conducted in different
19 locations, using different approaches, and reporting positive and statistically significant
20 associations with PM_{10-2.5}). In particular, such an approach would place more weight on positive
21 and statistically significant PM_{10-2.5} effect estimates that have been reported in single-city
22 analyses and less weight on PM_{10-2.5} effect estimates that are not statistically significant and/or
23 that reflect estimates across multiple cities. It could be judged appropriate to place less weight
24 on PM_{10-2.5} effect estimates that are not statistically significant given the relatively large amount
25 of uncertainty that is associated with the broader body of PM_{10-2.5} health evidence, including
26 uncertainty in the extent to which health effects evaluated in epidemiologic studies result from
27 exposures to PM_{10-2.5} itself, rather than one or more co-occurring pollutants. In addition, it could
28 be judged appropriate to place less weight on 98th percentile PM₁₀ concentrations averaged
29 across multiple cities, given the uncertainty in linking multi-city effect estimates with the air
30 quality in any particular location. To the extent this approach to considering the evidence and
31 uncertainties is applied, we conclude that a standard level around 85 µg/m³ would be appropriate
32 to consider.

33 In contrast, alternative approaches to considering the evidence could lead to consideration
34 of standard levels below 85 µg/m³. For example, given that exposure error is particularly
35 important for PM_{10-2.5} epidemiologic studies and can bias the results of these studies toward the
36 null hypothesis, it could be judged appropriate to place more weight on positive associations
37 reported in epidemiologic studies, even when those associations are not statistically significant.

1 In addition, despite the uncertainties associated with interpreting multi-city average air quality
2 concentrations, it could be judged appropriate to place weight on PM_{10-2.5} multi-city effect
3 estimates and, therefore, on the multi-city averages of 98th percentile PM₁₀ concentrations in
4 study locations. To the extent that these considerations are emphasized, we conclude that a
5 standard level as low as 65 µg/m³ could potentially be considered. A level of 65 µg/m³ would be
6 expected to maintain PM₁₀ and PM_{10-2.5} concentrations below those in all of the single-city
7 mortality study locations in Figure 3-5 above, including those where PM_{10-2.5} effect estimates
8 were not statistically significant. In addition, a standard level of 65 µg/m³ would be expected to
9 maintain PM₁₀ and PM_{10-2.5} concentrations below the multi-city averages of 98th percentile PM₁₀
10 concentrations in the locations evaluated by Zanobetti and Schwartz (2009) and Peng et al.
11 (2008), below concentrations in most locations of single-city morbidity studies (including studies
12 conducted in Atlanta reporting both positive and negative effect estimates), and below
13 concentrations in many locations where Bayes-adjusted single-city PM_{10-2.5} effect estimates were
14 positive, though not statistically significant.

15 In considering potential alternative standard levels below 65 µg/m³, we note that, as
16 discussed above, the overall body of PM_{10-2.5} health evidence is relatively uncertain, with
17 somewhat stronger support in U.S. studies for associations with PM_{10-2.5} in locations with 98th
18 percentile PM₁₀ concentrations above 87 µg/m³ than in locations with 98th percentile PM₁₀
19 concentrations below 65 µg/m³. Specifically, we note the following:

- 20 • Epidemiologic studies, either single-city or multi-city, have not reported positive and
21 statistically significant PM_{10-2.5} effect estimates in locations with 98th percentile PM₁₀
22 concentrations (multi-city average 98th percentile concentrations in the case of multi-city
23 studies) at or below 65 µg/m³.
24
- 25 • Although single-city morbidity studies have reported positive, but not statistically
26 significant, associations with PM_{10-2.5} in locations with 98th percentile PM₁₀
27 concentrations below 65 µg/m³, the results of U.S. morbidity studies were generally less
28 consistent than those of mortality studies, with some PM_{10-2.5} effect estimates being
29 positive while others were negative (i.e., negative effect estimates were reported in
30 several studies conducted in Atlanta, where the 98th percentile PM₁₀ concentrations
31 ranged from 67 µg/m³ to 71 µg/m³).
32
- 33 • Although Bayes-adjusted single-city PM_{10-2.5} effect estimates were positive, but not
34 statistically significant, in some locations with PM₁₀ concentrations below 65 µg/m³,
35 these effect estimates were based on the difference between community-wide PM₁₀ and
36 PM_{2.5} concentrations. As discussed above, it is not clear how these estimates of PM_{10-2.5}
37 concentrations compare to those more typically used in other studies to calculate PM_{10-2.5}
38 effect estimates. At present, corroborating studies are lacking that use other approaches
39 (i.e., co-located monitors, dichotomous samplers) to estimate/measure PM_{10-2.5} in
40 locations with the lowest PM₁₀ concentrations.

1 In light of these limitations in the evidence for a relationship between PM_{10-2.5} and
2 adverse health effects in locations with relatively low PM₁₀ concentrations, along with the
3 overall uncertainties in the body of PM_{10-2.5} health evidence as described above and in the ISA,
4 we conclude that while it could be judged appropriate to consider standard levels as low as 65
5 µg/m³, it is not appropriate, based on the currently available body of evidence, to consider
6 standard levels below 65 µg/m³.

7 To provide some perspective on the potential implications of standard levels from different
8 parts of the range of 65 to 85 µg/m³, staff assessed (based on 2006 to 2008 air quality data) the
9 percentage of counties, and the population in those counties, that would likely not attain different
10 standard levels within this range. The results of this analysis are presented in appendix 3A
11 (Table 3A-1), but these results were not considered as a basis for the above staff conclusions.

12 **3.4 STAFF CONCLUSIONS ON A PRIMARY THORACIC COARSE PARTICLE** 13 **STANDARD**

14 In reaching conclusions on the adequacy of the current PM₁₀ standard and potential
15 alternative standards to provide requisite protection against health effects associated with short-
16 term exposures to thoracic coarse particles, staff has considered the basic elements of the
17 NAAQS: indicator, averaging time, form, and level (section 3.3.1 to 3.3.4 above). In
18 considering available scientific evidence and air quality information, we reflect upon the
19 evidence and information available in the last review integrated with evidence and information
20 that has become available since that review as assessed and presented in the ISA (US EPA,
21 2009a) and summarized above in sections 3.2 and 3.3.

22 We recognize that selecting from among potential alternative standards will necessarily
23 reflect consideration of the evidence as well as the uncertainties inherent in that evidence. In
24 considering the current PM₁₀ standard and identifying potential alternative primary standards for
25 consideration, we are mindful that the Clean Air Act requires standards to be set that are
26 requisite to protect public health with an adequate margin of safety, such that the standards are to
27 be neither more nor less stringent than necessary. Thus, the Act does not require that the
28 NAAQS be set at zero-risk levels, but rather at levels that avoid unacceptable risks to public
29 health.

- 30 (1) Consideration should be given to retaining or revising the current 24-hour PM₁₀ primary
31 standard, depending on the relative weight placed on the evidence supporting associations
32 with PM_{10-2.5} and the uncertainties associated with this evidence.
- 33 (2) The indicator for thoracic coarse particles should continue to be PM₁₀. This conclusion is
34 based on our assessment of the evidence for effects related to particles of urban and non-
35 urban origins. We also conclude that future research should be targeted so as to inform
36 consideration of different indicators in future reviews.

1 (3) The averaging time should be 24-hours for a PM₁₀ standard meant to protect against short-
2 term exposures to thoracic coarse particles. This conclusion reflects the body of
3 epidemiologic studies, which are most often based on 24-hour average PM_{10-2.5}
4 concentrations.

5 (4) To the extent consideration is given to revising the current standard:

6 (a) Consideration should be given to a 98th percentile form for a 24-hour PM₁₀
7 standard. This conclusion is based on consideration of providing a balance between
8 limiting peak concentrations and providing a stable regulatory target, compensating
9 for differences in monitoring frequency across locations, and focusing the standard
10 on days when PM_{10-2.5} is likely to make a relatively larger contribution to PM₁₀
11 mass.

12 (b) Consideration should be given to PM₁₀ standard levels from 85 µg/m³ down to
13 about 65 µg/m³. This range of levels is based on consideration of 98th percentile
14 PM₁₀ concentrations in U.S. study locations where PM_{10-2.5} epidemiologic studies
15 have been conducted. We conclude that standard levels around the upper end of
16 this range are most strongly supported by the evidence given the availability of
17 multiple single-city analyses in different locations and using different approaches to
18 estimate PM_{10-2.5}, with some reporting positive and statistically significant PM_{10-2.5}
19 effect estimates. Standard levels as low as 65 µg/m³ could be considered to the
20 extent more weight is placed on positive, but not statistically significant,
21 associations and on multi-city PM_{10-2.5} effect estimates.

22 **3.5 KEY UNCERTAINTIES AND AREAS FOR FUTURE RESEARCH AND** 23 **DATA COLLECTION**

24 In this section, we highlight key uncertainties and areas for future health-related research,
25 model development, and data collection activities. These efforts, if undertaken, could provide
26 important evidence for informing future PM NAAQS reviews and, in particular, consideration of
27 possible alternative indicators, averaging times, forms, and/or levels. In some cases, research in
28 these areas can go beyond aiding standard setting to informing the development of more efficient
29 and effective control strategies.

30 As an initial matter, we note that many of the research needs identified for fine particles
31 (see above, section 2.5) are also relevant for thoracic coarse particles. This includes research in
32 the following areas:

- 33 • Sources and components of PM_{10-2.5}, including source apportionment modeling;
34 monitoring of components; linking specific sources/components to health outcomes;
35 linking sources/components to intra- and inter-city differences in health effects; and
36 linking sources/components to population exposures
- 37 • Understanding the extent to which an association between thoracic coarse particles and
38 specific health effects can be modified by co-pollutants

- 1 • Understanding associations with a broad range of cardiovascular and respiratory
2 endpoints as well as adverse effects in the nervous system, on reproduction, and/or on
3 development
- 4 • Understanding C-R relationships and the confidence bounds around these relationships,
5 especially at lower ambient thoracic coarse particle concentrations
- 6 • Understanding air quality distributions in locations of epidemiologic studies
- 7 • Identifying populations susceptible to PM_{10-2.5}-related health effects
- 8 • Modeling to estimate PM_{10-2.5} mass and composition in areas with less-than-daily
9 monitoring

10 These uncertainties and areas for future research are discussed above in section 2.5 and
11 that discussion will not be repeated here. In addition to the above, there are several areas for
12 future research that are particularly relevant for thoracic coarse particles. These include the
13 following:

- 14 • The body of experimental inhalation studies (e.g., controlled human exposure and animal
15 toxicology studies) is currently relatively sparse. Additional well-conducted experimental
16 studies could play an important role in weight of evidence judgments in future ISAs.
17 Therefore, experimental evaluation of effects (e.g., vasomotor function, airways
18 responsiveness, pulmonary function/inflammation) of concentrated ambient PM_{10-2.5} from
19 specific sources (e.g., traffic, industrial, non-industrial) would be useful, particularly if
20 exposure-response relationships are evaluated.
- 21 • Exposure error is of particular concern for thoracic coarse particles, given the relative lack of
22 monitoring and its less homogeneous atmospheric distribution compared to fine particles (US
23 EPA, 2009a, section 2.1.1.2). Therefore, short-term studies with well-characterized personal
24 exposures (e.g., panel studies) would be useful. Such studies could examine indicators of
25 cardiovascular and respiratory morbidity (e.g., arrhythmia, ischemia, vasomotor function,
26 respiratory symptoms, pulmonary inflammation, pulmonary function, pulmonary injury) and
27 would be particularly useful if they evaluated concentration-response relationships and/or
28 effects of repeated peak exposures.
- 29 • Epidemiologic studies currently use a variety of approaches to measure/estimate PM_{10-2.5}
30 concentrations. It is important that we better understand the relationship between results
31 from studies that estimate PM_{10-2.5} concentrations using either (1) difference method of co-
32 located monitors, (2) difference method of county-wide averages of PM₁₀ and PM_{2.5}, or (3)
33 direct measurement of PM_{10-2.5} using a dichotomous sampler. In addition, as described
34 above, PM_{10-2.5} monitoring will be required at NCORE sites by 2011. It would be useful for
35 future epidemiologic studies to make use of these new PM_{10-2.5} monitoring sites.
- 36 • Very little information is available to inform weight of evidence conclusions for endpoints
37 associated with long-term PM_{10-2.5} exposures. Epidemiologic and animal toxicological
38 studies of long-term exposures (i.e., months to years) to PM_{10-2.5} would be helpful. Such
39 studies could evaluate links with cardiovascular and respiratory morbidity, reproductive and
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1 **4 REVIEW OF THE SECONDARY STANDARDS FOR VISIBILITY-**
2 **RELATED EFFECTS**

3 This chapter presents staff conclusions with regard to the adequacy of the current suite of
4 secondary PM_{2.5} standards to protect against PM-related visibility impairment as well as
5 alternative secondary PM standards that are appropriate for consideration in this review. Our
6 assessment of these issues is framed by a series of key policy-relevant questions, which expand
7 upon those presented at the outset of this review in the IRP. The answers to these questions will
8 inform decisions on whether, and if so how, to revise the current suite of secondary PM_{2.5}
9 standards for the purpose of providing appropriate protection from PM-related visibility
10 impairment.

11 In presenting staff conclusions on a range of alternative secondary standards that are
12 appropriate for consideration, we note that the final decision is largely a public welfare policy
13 judgment. A final decision must draw upon scientific information and analyses about PM-
14 related visibility impairment and related impacts on public welfare, as well as judgments about
15 how to deal with the range of uncertainties that are inherent in the scientific evidence and
16 analyses. Our approach to informing these judgments is discussed more fully below.

17 Information on the approaches used to set the secondary PM_{2.5} standards in past reviews
18 as well as our current approach for this review are presented in section 4.1. Our conclusions
19 regarding the adequacy of the current suite of secondary PM_{2.5} standards to protect against PM-
20 related visibility impairment are presented in section 4.2. Section 4.3 presents our conclusions
21 with respect to alternative PM standards by focusing on each of the basic elements of the
22 standards: pollutant indicator (section 4.3.1), averaging time (section 4.3.2), and level and form
23 (section 4.3.3). The performance of alternative standards, with a focus on the uniformity of
24 protection from visibility impairment afforded by the alternative standards, is evaluated in
25 section 4.3.4. Section 4.4 summarizes all staff conclusions on the secondary PM standards for
26 visibility protection. This chapter concludes with an overview of areas of key uncertainties and
27 suggested future research areas and data collection efforts (section 4.5)

28 **4.1 APPROACH**

29 Staff’s approach for reviewing the current suite of secondary PM_{2.5} standards builds upon
30 and broadens the approaches used in previous PM NAAQS reviews. We first present a brief
31 summary of the approaches used to review and establish secondary PM standards in the last two
32 reviews of the PM NAAQS (section 4.1.1). Recent litigation on the 2006 standards has resulted
33 in the remand of the secondary annual and 24-hour PM_{2.5} NAAQS to EPA as discussed in section

1 4.1.2. Our current approach for evaluating the secondary PM_{2.5} standards using both evidence-
2 and impact assessment-based considerations is outlined in section 4.1.3.

3 **4.1.1 Approaches Used in Previous Reviews**

4 The original suite of secondary PM_{2.5} standards was established in 1997 and revisions to
5 those standards were made in 2006. The approaches used in making final decisions on
6 secondary standards in those reviews, as well as the current review, utilize different ways to
7 consider the underlying body of scientific evidence. They also reflect an evolution in our
8 understanding of the nature of the effect on public welfare from visibility impairment, from that
9 focused only on Class I area visibility impacts to a more multifaceted approach that also
10 considers PM-related impacts on non-Class I area visibility, such as in urban areas. This
11 evolution has occurred in conjunction with the expansion of available PM data and information
12 from associated studies of public perception, valuation and personal comfort and well being.

13 **4.1.1.1 Review Completed in 1997**

14 In 1997, EPA revised the identical primary and secondary PM₁₀ NAAQS in part by
15 establishing new identical primary and secondary PM_{2.5} standards. In revising the secondary
16 standards, EPA recognized that PM produces adverse effects on visibility and that impairment of
17 visibility was experienced throughout the U.S., in multi-state regions, urban areas, and remote
18 mandatory Class I Federal areas alike. However, in considering an appropriate level for a
19 secondary standard to address adverse effects of PM_{2.5} on visibility, EPA concluded that the
20 determination of a single national level was complicated by regional differences in several
21 factors that influence visibility such as background and current levels of PM_{2.5}, composition of
22 PM_{2.5}, and average relative humidity. Variations in these factors across regions could thus result
23 in situations where attaining an appropriately protective concentration of fine particles in one
24 region might or might not provide adequate protection in a different region. The EPA also
25 determined that there was insufficient information at that time to establish a level for a national
26 secondary standard that would represent a threshold above which visibility conditions would
27 always be adverse and below which visibility conditions would always be acceptable.

28 Based on these considerations, EPA assessed potential visibility improvements in urban
29 areas and on a regional scale that would result from attainment of the new primary standards for
30 PM_{2.5}. The agency concluded that the spatially averaged form of the annual PM_{2.5} standard was
31 well suited to the protection of visibility, which involves effects of PM_{2.5} throughout an extended
32 viewing distance across an urban area. Based on air quality data available at that time, many
33 urban areas in the Northeast, Midwest, and Southeast, as well as Los Angeles, were expected to
34 see perceptible improvement in visibility if the annual PM_{2.5} primary standard was attained. The
35 EPA also concluded that in some areas attainment of the 24-hour PM_{2.5} standard would be

1 expected to reduce, to some degree, the number and intensity of “bad visibility” days, i.e., the
2 20% of days having the greatest impairment over the course of a year.

3 Having concluded that attainment of the annual and 24-hour PM_{2.5} primary standards
4 would lead to visibility improvements in many eastern and some western urban areas, EPA also
5 considered whether these standards could provide potential improvements to visibility on a
6 regional scale. Based on information available at the time, EPA concluded that attainment of
7 PM_{2.5} secondary standards set identical to the primary standards would be expected to result in
8 visibility improvements in the eastern U.S. at both urban and regional scales, but little or no
9 change in the western U.S., except in and near certain urban areas.

10 The EPA then considered the potential effectiveness of a regional haze program, required
11 by sections 169A and 169B of the Act¹ to address those effects of PM on visibility that would
12 not be addressed through attainment of the primary PM_{2.5} standards. The regional haze program
13 would be designed to address the widespread, regionally uniform type of haze caused by a
14 multitude of sources. The structure and requirements of sections 169A and 169B of the Act
15 provide for visibility protection programs that can be more responsive to the factors contributing
16 to regional differences in visibility than can programs addressing a nationally applicable
17 secondary NAAQS. The regional haze visibility goal is more protective than a secondary
18 NAAQS since the goal addresses any man-made impairment rather than just impairment at levels
19 determined to be adverse. Thus, an important factor considered in the 1997 review was whether
20 a regional haze program, in conjunction with secondary standards set identical to the suite of
21 PM_{2.5} primary standards, would provide appropriate protection for visibility in non-Class I areas.
22 The EPA concluded that the two programs and associated control strategies should provide such
23 protection due to the regional approaches needed to manage emissions of pollutants that impair
24 visibility in many of these areas.

25 For these reasons, EPA concluded that a national regional haze program, combined with
26 a nationally applicable level of protection achieved through secondary PM_{2.5} standards set
27 identical to the primary PM_{2.5} standards, would be an appropriate approach for addressing
28 regional variations in the adverse effects of PM_{2.5} on visibility than establishing national
29 secondary standards for PM at lower levels than the primary PM_{2.5} standards. The EPA further
30 recognized that people living in certain urban areas may place a high value on unique scenic
31 resources in or near these areas, and as a result might experience visibility problems attributable
32 to sources that would not necessarily be addressed by the combined effects of a regional haze

¹ In 1977, Congress established as a national goal “the prevention of any future, and the remedying of any existing, impairment of visibility in mandatory Class I Federal areas which impairment results from manmade air pollution”, section 169A(a)(1) of the Act. The EPA is required by section 169A(a)(4) of the Act to promulgate regulations to ensure that “reasonable progress” is achieved toward meeting the national goal.

1 program and PM_{2.5} secondary standards. The EPA concluded that in such cases, state or local
2 regulatory approaches, such as past action in Colorado to establish a local visibility standard for
3 the City of Denver, would be more appropriate and effective in addressing these special
4 situations because of the localized and unique characteristics of the problems involved.
5 Visibility in an urban area located near a mandatory Class I Federal area could also be improved
6 through state implementation of the current visibility regulations, by which emission limitations
7 can be imposed on a source or group of sources found to be contributing to “reasonably
8 attributable” impairment in the mandatory Class I Federal area.

9 Based on these considerations, EPA set secondary PM_{2.5} standards identical to the
10 primary PM_{2.5} standards, in conjunction with a regional haze program under sections 169A and
11 169B of the Act, as the most appropriate and effective means of addressing the welfare effects
12 associated with visibility impairment. Together, the two programs and associated control
13 strategies were expected to provide appropriate protection against the effects of PM on visibility
14 and enable all regions of the country to make reasonable progress toward the national visibility
15 goal.

16 **4.1.1.2 Review Completed in 2006**

17 In 2006, EPA revised the secondary PM_{2.5} standards to address visibility impairment by
18 making them identical to the revised primary standards. The EPA’s decision regarding the need
19 to revise the secondary PM_{2.5} standards reflected a number of new developments and sources of
20 information that had occurred and/or become available following the 1997 review. First, EPA
21 promulgated a regional haze program in 1999 (65 FR 35713) which required states to establish
22 goals for improving visibility in Class I areas and to adopt control strategies to achieve these
23 goals. Second, extensive new information from visibility and fine particle monitoring networks
24 had become available, allowing for updated characterizations of visibility trends and PM levels
25 in urban areas, as well as Class I areas. These new data allowed EPA to better characterize
26 visibility impairment in urban areas and the relationship between visibility and PM_{2.5}
27 concentrations. Finally, additional studies in the U.S. and abroad provided the basis for the
28 establishment of standards and programs to address specific visibility concerns in a number of
29 local areas. These studies (e.g., in Denver, Phoenix, British Columbia) utilized photographic
30 representations of visibility impairment and produced reasonably consistent results in terms of
31 the visual ranges found to be generally acceptable by study participants. EPA considered the
32 information generated by these studies useful in characterizing the nature of particle-induced
33 haze and for informing judgments about the acceptability of various levels of visual air quality in
34 urban areas across the U.S. Based largely on this information, the Administrator concluded that
35 it was appropriate to revise the secondary PM standards to provide increased protection from

1 visibility impairment principally in urban areas, in conjunction with the regional haze program
2 for protection of visual air quality in Class I areas.

3 In so doing, the Administrator recognized that PM-related visibility impairment is
4 principally related to fine particle levels, such that it was appropriate to focus the review on
5 whether the current secondary PM_{2.5} standards should be revised. The Administrator also
6 recognized that perception of visibility impairment is most directly related to instantaneous
7 levels of visual air quality, such that in considering whether the current suite of secondary
8 standards would provide the appropriate degree of protection, he concluded that it was
9 appropriate to focus on just the 24-hour secondary PM_{2.5} standard to provide requisite protection.

10 The EPA then considered whether PM_{2.5} remained the appropriate indicator for a
11 secondary standard to protect visibility, primarily in urban areas. EPA noted that PM-related
12 visibility impairment is principally related to fine particle levels. Hygroscopic components of
13 fine particles, in particular sulfates and nitrates, contribute disproportionately to visibility
14 impairment under high humidity conditions. Particles in the coarse mode generally contribute
15 only marginally to visibility impairment in urban areas. With the substantial addition to the air
16 quality and visibility data made possible by the national urban PM_{2.5} monitoring networks, an
17 analysis conducted for the 2006 review found that, in urban areas, visibility levels showed far
18 less difference between eastern and western regions on a 24-hour or shorter time basis than
19 implied by the largely non-urban data available in the 1997 review. In analyzing how well PM_{2.5}
20 concentrations correlated with visibility in urban locations across the U.S., the 2005 Staff Paper
21 (US EPA, 2005) concluded that clear correlations existed between 24-hour average PM_{2.5}
22 concentrations and reconstructed light extinction, which is directly related to visual range. These
23 correlations were similar in the eastern and western regions of the U.S. These correlations were
24 less influenced by relative humidity and more consistent across regions when PM_{2.5}
25 concentrations are averaged over shorter, daylight time periods (e.g., 4 to 8 hours) when relative
26 humidity was generally lower and less variable. The 2005 Staff Paper noted that a standard set at
27 any specific PM_{2.5} concentration would necessarily result in visual ranges that vary somewhat in
28 urban areas across the country, reflecting the variability in the correlations between PM_{2.5}
29 concentrations and light extinction. The 2005 Staff Paper concluded that it was appropriate to
30 use PM_{2.5} as an indicator for standards to address visibility impairment in urban areas, especially
31 when the indicator is defined for a relatively short period (e.g., 4 to 8 hours) of daylight hours.
32 Based on their review of the Staff Paper, most CASAC Panel members also endorsed such a
33 PM_{2.5} indicator for a secondary standard to address visibility impairment (Henderson, 2005a).
34 Based on the above considerations, the Administrator concluded that PM_{2.5} should be retained as
35 the indicator for fine particles as part of a secondary standard to address visibility protection, in
36 conjunction with averaging times from 4 to 24 hours.

1 In considering what level of protection against PM-related visibility impairment would be
2 appropriate, the Administrator took into account the results of the public perception and attitude
3 surveys in the U.S. and Canada, state and local visibility standards within the U.S., and visual
4 inspection of photographic representations of several urban areas across the U.S. In the
5 Administrator's judgment, these sources provided useful but still quite limited information on the
6 range of levels appropriate for consideration in setting a national visibility standard primarily for
7 urban areas, given the generally subjective nature of the public welfare effect involved. Based
8 on photographic representations of varying levels of visual air quality, public perception studies,
9 and local and state visibility standards, the 2005 Staff Paper concluded that 30 to 20 $\mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$
10 represented a reasonable range for a national visibility standard primarily for urban areas, based
11 on a sub-daily averaging time. The upper end of this range was below the levels at which
12 illustrative scenic views are significantly obscured, and the lower end was around the level at
13 which visual air quality generally appeared to be good based on observation of the illustrative
14 views. This concentration range generally corresponded to median visual ranges in urban areas
15 within regions across the U.S. of approximately 25 to 35 km, a range which was bounded above
16 by the visual range targets selected in specific areas where state or local agencies placed
17 particular emphasis on protecting visual air quality. In considering a reasonable range of forms
18 for a $\text{PM}_{2.5}$ standard within this range of levels, the 2005 Staff Paper concluded that a
19 concentration-based percentile form was appropriate, and that the upper end of the range of
20 concentration percentiles should be consistent with the 98th percentile used for the primary
21 standard and that the lower end of the range should be the 92nd percentile, which represented the
22 mean of the distribution of the 20 percent most impaired days, as targeted in the regional haze
23 program. While recognizing that it was difficult to select any specific level and form based on
24 then currently available information (Henderson, 2005a), the CASAC Panel was generally in
25 agreement with the ranges of levels and forms presented in the 2005 Staff Paper.

26 The Administrator also considered the level of protection that would be afforded by the
27 proposed suite of primary $\text{PM}_{2.5}$ standards (71 FR 2681), on the basis that although significantly
28 more information was available than in the 1997 review concerning the relationship between fine
29 PM levels and visibility across the country, there was still little available information for use in
30 making the relatively subjective value judgment needed in selecting the appropriate degree of
31 protection to be afforded by such a standard. In so doing, the Administrator compared the extent
32 to which the proposed suite of primary standards would require areas across the country to
33 improve visual air quality with the extent of increased protection likely to be afforded by a
34 standard based on a sub-daily averaging time. Based on such an analysis, the Administrator
35 observed that the predicted percent of counties with monitors not likely to meet the proposed
36 suite of primary $\text{PM}_{2.5}$ standards was actually somewhat greater than the predicted percent of

1 counties with monitors not likely to meet a sub-daily secondary standard with an averaging time
2 of 4 daylight hours, a level toward the upper end of the range recommended in the 2005 Staff
3 Paper, and a form within the recommended range. Based on this comparison, the Administrator
4 concluded that revising the secondary 24-hour PM_{2.5} standard to be identical to the proposed
5 revised primary PM_{2.5} standard (and retaining the current annual secondary PM_{2.5} standard) was a
6 reasonable policy approach to addressing visibility protection primarily in urban areas. In
7 proposing this approach, the Administrator also solicited comment on a sub-daily (4- to 8-hour
8 averaging time) secondary PM_{2.5} standard (71 FR 2675-2781).

9 In commenting on EPA's proposal, the CASAC requested that a sub-daily standard to
10 protect visibility be favorably reconsidered (Henderson, 2006). The CASAC noted three
11 cautions regarding the Agency's proposed reliance on a secondary PM_{2.5} standard identical to the
12 proposed 24-hour primary PM_{2.5} standard: (1) PM_{2.5} mass measurement is a better indicator of
13 visibility impairment during daylight hours, when relative humidity is generally low; the sub-
14 daily standard more clearly matches the nature of visibility impairment, whose adverse effects
15 are most evident during the daylight hours; using a 24- hour standard as a proxy introduces error
16 and uncertainty in protecting visibility; and sub-daily standards are used for other NAAQS and
17 should be the focus for visibility; (2) CASAC and its monitoring subcommittees have repeatedly
18 commended EPA's initiatives promoting the introduction of continuous and near-continuous PM
19 monitoring, and expanded deployment of continuous PM_{2.5} monitors is consistent with setting a
20 sub-daily standard to protect visibility; (3) The analysis showing a similarity between
21 percentages of counties not likely to meet what they considered to be a lenient 4- to 8-hour
22 secondary standard and a secondary standard identical to the proposed 24-hour primary standard
23 was a numerical coincidence that was not indicative of any fundamental relationship between
24 visibility and health. The CASAC Panel further stated that "visual air quality is substantially
25 impaired at PM_{2.5} concentrations of 35 µg/m³" and that "it is not reasonable to have the visibility
26 standard tied to the health standard, which may change in ways that make it even less appropriate
27 for visibility concerns."

28 In reaching a final decision, the Administrator focused on the relative protection provided
29 by the proposed primary standards and the limitations in the information available concerning
30 studies of public perception and attitudes regarding the acceptability of various degrees of
31 visibility impairment in urban areas, as well as on the subjective nature of the judgment required.
32 In so doing, the Administrator concluded that caution was warranted in establishing a distinct
33 secondary standard for visibility impairment and that the available information did not warrant
34 adopting a secondary standard that would provide either more or less protection against visibility
35 impairment in urban areas than would be provided by secondary standards set equal to the
36 proposed primary PM_{2.5} standards.

1 **4.1.2 Remand of Secondary PM_{2.5} Standards**

2 Several parties filed petitions for review following promulgation of the revised PM
3 NAAQS in 2006. These petitions challenged several aspects of the final rule including EPA’s
4 decision to set the secondary NAAQS for fine PM, which protect the public welfare from
5 adverse visibility effects, at the same level as the primary NAAQS, which protect public health.

6 On judicial review, the D.C. Circuit remanded to EPA for reconsideration the secondary
7 NAAQS for fine PM because the Agency’s decision was unreasonable and contrary to the
8 requirements of section 109(b)(2). The petitioners argued that EPA’s decision lacked a reasoned
9 basis. First, they asserted that EPA never determined what level of visibility was “requisite to
10 protect the public welfare”. They argued that EPA unreasonably rejected the target level of
11 protection provided by its staff, while failing to provide a target level of its own. The court
12 stated “the EPA’s failure to identify such a level when deciding where to set the level of air
13 quality required by the revised secondary fine PM NAAQS is contrary to the statute and
14 therefore unlawful. Furthermore, the failure to set any target level of visibility protection
15 deprived the EPA’s decision-making of a reasoned basis.” *American Farm Bureau Federation v.*
16 *EPA*, 559 F. 3d 512 (D.C. Cir. 2009).

17 Second, the petitioners challenged EPA’s method of comparing the protection expected
18 from potential standards. They contended that the EPA relied on a meaningless numerical
19 comparison, ignored the effect of humidity on the usefulness of a standard using a daily
20 averaging time, and unreasonably concluded that the primary standards would achieve a level of
21 visibility roughly equivalent to the level the EPA staff and CASAC deemed “requisite to protect
22 the public welfare.” Again, the court found that EPA’s equivalency analysis failed on its own
23 terms. The same table showing the alternative secondary standard used for comparison to the
24 alternative primary to show equivalency also included six other standards within the
25 recommended CASAC range that would be more “protective” under EPA’s definition than the
26 primary standards. Two-thirds of the potential standards within the CASAC’s recommended
27 range would be substantially more protective than the primary standards. The court found that
28 EPA failed to explain why it looked only at one of the few potential standards that would be less
29 protective and only slightly so than the primary standards. More fundamentally, however, the
30 court found that EPA’s equivalency analysis demonstrated nothing about the relative protection
31 offered by the different standards, and that the tables offered no valid information about the
32 relative visibility protection provided by the standards.

33 Finally, the Staff Paper made clear that a visibility standard using a daily averaging time
34 would be confounded by regional differences in humidity. The EPA acknowledged this
35 problem, yet did not address this issue in concluding that the primary standards would be
36 sufficiently protective of visibility. Therefore, the court granted the petition for review and

1 remanded for reconsideration the secondary fine PM NAAQS. American Farm Bureau
2 Federation, 559 F. 3d at 520-24.

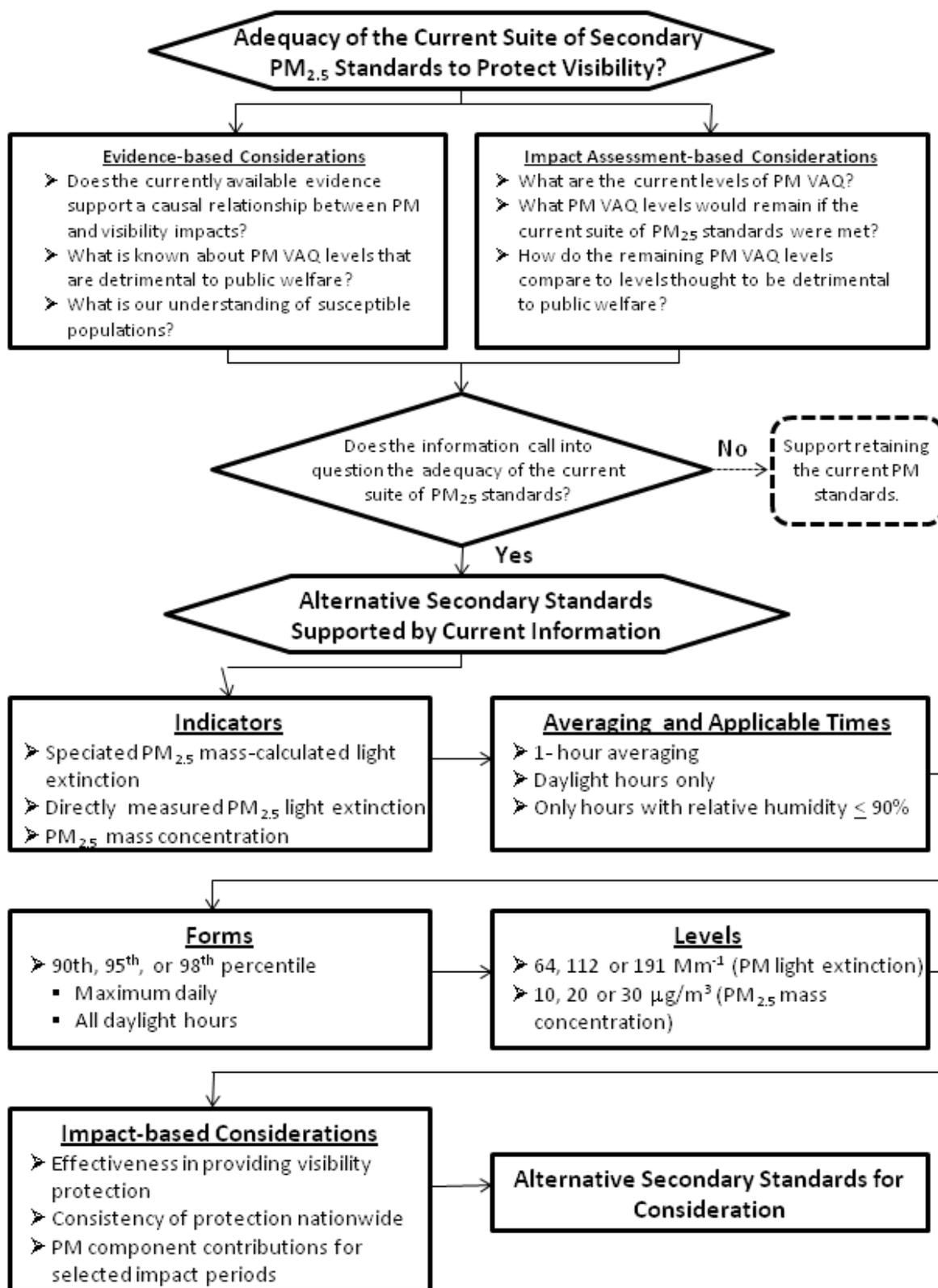
3 **4.1.3 Current Approach**

4 The staff's approach in this review broadens the general approaches used in the last two
5 PM NAAQS reviews by utilizing, to the extent available, enhanced tools, methods, and data to
6 more comprehensively characterize visibility impacts. As such, staff is taking into account both
7 evidence-based and impact-based considerations to inform staff conclusions related to the
8 adequacy of the current secondary PM_{2.5} standards and alternative standards that are appropriate
9 for consideration in this review. In so doing, we are seeking to provide as broad an array of
10 options as is supportable by the available information, recognizing that the selection of a specific
11 approach to reaching final decisions on the secondary PM_{2.5} standards will reflect the judgments
12 of the Administrator. Figure 4-1 below provides a diagram of our approach.

13 For the purposes of this second draft Policy Assessment (PA), staff has drawn from the
14 qualitative evaluation of all studies discussed in the *Integrated Science Assessment for*
15 *Particulate Matter (Final Report)* (ISA, USEPA, 2009a). The discussions presented in this
16 chapter consider the extensive new air quality and source apportionment information available
17 from the regional planning organizations, long-standing evidence of PM effects on visibility, and
18 public preference studies from four urban areas, as discussed in chapter 9 of the ISA, as well as
19 the integration of evidence across disciplines presented in chapter 2 of the ISA. In addition,
20 limited information that has become available regarding the characterization of public
21 preferences in urban areas has provided some new perspectives on the usefulness of this
22 information in informing the selection of target levels of urban visibility protection. On these
23 bases, we are again focusing our assessments in this review on visibility conditions in urban
24 areas.

25 Our conclusions reflect our understanding of both evidence-based and impact-based
26 considerations to inform two overarching questions related to: (1) the adequacy of the current
27 suite of PM_{2.5} standards and (2) what potential alternative standards, if any, should be considered
28 in this review to provide appropriate protection from PM-related visibility impairment. In
29 addressing these broad questions, we have organized the discussions below around a series of
30 more specific questions reflecting different aspects of each overarching question. When
31 evaluating the visibility protection afforded by the current or any alternative standards
32 considered, we have taken into account the four basic elements of the NAAQS (e.g., indicator,
33 averaging time, level, and form).

Figure 4-1. Overview of Approach for Review of Secondary PM_{2.5} Standards



1 **4.2 ADEQUACY OF CURRENT STANDARDS**

2 In considering the adequacy of the current suite of PM_{2.5} standards, staff addresses the
3 following overarching question:

4 **Does the currently available scientific evidence and visibility impact information, as**
5 **reflected in the ISA and UFVA, support or call into question the adequacy of the visibility**
6 **protection afforded by the current suite of fine particle standards?**

7 To inform the answer to this overarching question, we have posed a series of more
8 specific questions to aid in considering the currently available scientific evidence and the results
9 of recent quantitative visibility impact analyses in a policy-relevant context, as discussed below.
10 In considering the scientific and technical information, we reflect upon both the information
11 available in the last review and information that is newly available since the last review as
12 assessed and presented in the final ISA and final UFVA (US EPA, 2009a; US EPA, 2010b).

13 **4.2.1 Evidence-based Considerations**

14 In reviewing the adequacy of the current suite of PM_{2.5} standards, we have taken into
15 account evidence-based considerations, primarily as presented in the ISA, by considering causal
16 inference, impacts on susceptible populations, and the kind of visibility effects that have been
17 observed in urban areas that would likely meet the current standards.

- 18 • **To what extent does the newly available scientific evidence strengthen or call into**
19 **question evidence of associations between ambient fine particle exposures and visibility**
20 **effects?**

21 New research conducted by regional planning organizations in support of the Regional
22 Haze Rule, as discussed in chapter 9 of the ISA, continues to support and refine our
23 understanding of the nature of the PM visibility effect and the source contributions to that effect
24 in rural and remote locations. Additional byproducts of this research include new insights
25 regarding the regional source contributions to urban visibility and better characterization of the
26 urban excess that occurs in many cities above regional background. Ongoing urban PM_{2.5}
27 speciated and mass monitoring has produced new information that has allowed for updated
28 characterization of visibility trends and current levels in urban areas. Information from both of
29 these sources of PM data, while useful, has not however changed the fundamental and long
30 understood science characterizing the contribution of PM, especially fine particles, to visibility
31 impairment from the last review. This science, briefly summarized below, provides the basis for
32 the ISA designation of the relationship between PM and visibility impairment as causal.

33 Visibility impairment is caused by the scattering and absorption of light by suspended
34 particles and gases in the atmosphere. The combined effect of light scattering and absorption by

1 both particles and gases is characterized as light extinction, (i.e. the fraction of light that is
2 scattered or absorbed per unit of distance in the atmosphere). Light extinction is measured in
3 units of 1/distance, which is often expressed in the technical literature as 1/(million meters) or
4 inverse megameters (abbreviated Mm^{-1}). When PM is present in the air, its contribution to light
5 extinction typically greatly exceeds that of gases.

6 The amount of light extinction contributed by PM depends on the particle size
7 distribution and composition, as well as its concentration. If details of the ambient particle size
8 distribution and composition (including the mixing of components) are known, Mie theory can
9 be used to accurately calculate PM light extinction (ISA chapter 9). However, routine
10 monitoring rarely includes measurements of particle size and composition information with
11 sufficient detail for such calculations. A much simpler algorithm can be used to estimate light
12 extinction using routinely monitored fine particle ($PM_{2.5}$) speciation and coarse particle mass
13 ($PM_{10-2.5}$) data, plus relative humidity information needed to estimate the contribution by liquid
14 water in solution with hygroscopic PM components (ISA section 9.2.2.2 and UFVA chapter 3).

15 The concentration of each of the major aerosol components is multiplied by a dry
16 extinction efficiency value and for the hygroscopic components (e.g., ammoniated sulfate and
17 ammonium nitrate) an additional multiplicative term to account for the water growth to estimate
18 these components' contribution to light extinction. Both the dry extinction efficiency and water
19 growth terms are developed by some combination of empirical assessment and theoretical
20 calculation using typical particle size distributions associated with each of the major aerosol
21 components, and they are evaluated by comparing the algorithm estimates of light extinction
22 with coincident optical measurements. Summing the contribution of each component gives the
23 estimate of total light extinction. The most commonly used algorithm is referred to as the
24 IMPROVE algorithm because it was developed specifically to use the aerosol monitoring data
25 generated at network sites and with equipment specifically designed to support the Interagency
26 Monitoring of Protected Visual Environments (IMPROVE) program and was evaluated using
27 IMPROVE optical measurements at the subset of sites that make those measurements (Malm et
28 al., 1994). The formula for the traditional IMPROVE algorithm is shown below.

$$\begin{aligned} b_{ext} \approx & 3 \times f(RH) \times [\text{Sulfate}] \\ & + 3 \times f(RH) \times [\text{Nitrate}] \\ & + 4 \times [\text{Organic Mass}] \\ & + 10 \times [\text{Elemental Carbon}] \\ & + 1 \times [\text{Fine Soil}] \\ & + 0.6 \times [\text{Coarse Mass}] \\ & + 10 \end{aligned}$$

1 Light extinction (b_{ext}) is in units of Mm^{-1} , the mass concentrations of the components
2 indicated in brackets are in $\mu\text{g}/\text{m}^3$, and $f(RH)$ is the unitless water growth term that depends on
3 relative humidity. The dry extinction efficiency for particulate organic mass is larger than those
4 for particulate sulfate and nitrate principally because the density of the dry inorganic compounds
5 is higher than that assumed for the PM organic mass components. Since IMPROVE does not
6 include ammonium ion monitoring, the assumption is made that all sulfate is fully neutralized
7 ammonium sulfate and all nitrate is assumed to be ammonium nitrate. Though often reasonable,
8 neither assumption is always true (see Section 9.2.3.1). In the eastern U.S. during the summer
9 there is insufficient ammonia in the atmosphere to neutralize the sulfate fully. Fine particle
10 nitrates can include sodium or calcium nitrate, which are the fine particle fraction of generally
11 much coarser particles due to nitric acid interactions with sea salt at near-coastal areas (sodium
12 nitrate) or nitric acid interactions with calcium carbonate in crustal aerosol (calcium nitrate).
13 Despite the simplicity of the algorithm, it performs reasonably well and permits the contributions
14 to light extinction from each of the major components (including the water associated with the
15 sulfate and nitrate compounds) to be separately approximated.

16 The $f(RH)$ terms inflate the particulate sulfate and nitrate light scattering for high relative
17 humidity conditions. For relative humidity below 40% the $f(RH)$ value is 1, but it increases to 2
18 at ~66%, 3 at ~83%, 4 at ~90%, 5 at ~93% and 6 at ~95% relative humidity. The result is that
19 both particulate sulfate and nitrate are more efficient per unit mass than any other aerosol
20 component for relative humidity above ~85% where its total light extinction efficiency exceeds
21 the $10 \text{ m}^2/\text{g}$ associated with elemental carbon (EC). Based on this algorithm, particulate sulfate
22 and nitrate are estimated to have comparable light extinction efficiencies (i.e., the same dry
23 extinction efficiency and $f(RH)$ water growth terms), so on a per unit mass concentration basis at
24 any specific relative humidity they are treated as equally effective contributors to visibility
25 effects.

26 Inspection of the PM component-specific terms in the simple algorithm shows that most
27 of the $\text{PM}_{2.5}$ components contribute 5 times or more light extinction than a similar concentration
28 of $\text{PM}_{10-2.5}$. We also know that particles with hygroscopic components (e.g. particulate sulfate
29 and nitrate) contribute more light extinction at higher relative humidity than at lower relative
30 humidity because they change size in the atmosphere in response to ambient relative humidity
31 conditions. PM containing elemental or black carbon absorbs light as well as scattering it,
32 making it the component with the greatest light extinction contributions per unit of mass
33 concentration, except for the hygroscopic components under high relative humidity conditions.

34 As a result of better characterization of PM in rural and remote locations, refinements in
35 the original IMPROVE algorithm have been made to better account for the aging of the organic
36 aerosols that occurs during transport from urban to more distant areas, and to add a component

1 for sea salt. However, the focus on urban areas in this review made these refinements
2 unnecessary due to the fact that urban areas contain a higher proportion of newly emitted
3 aerosols, and because particulate sea salt, while expected to contribute to visibility impairment in
4 coastal areas, is natural in origin and not controllable through regulation. Therefore, the
5 assessments done in support of this review have relied on the original IMPROVE algorithm,

6 As mentioned above, particles are not the only contributor to ambient visibility
7 conditions. Light scattering by gases also occurs in ambient air. Under pristine atmospheric
8 conditions, naturally occurring gases such as N₂ and O₂ cause what is known as Rayleigh
9 scattering. Rayleigh scattering, which depends on the density of air as a function primarily of the
10 elevation above sea level, can be treated as a site-dependent constant. Rayleigh contribution to
11 light extinction is only significant under pristine conditions. The only other commonly occurring
12 atmospheric gas to appreciably absorb light in the visible spectrum is NO₂. NO₂ forms in the
13 atmosphere from NO emissions associated with combustion processes. These combustion
14 processes also emit PM at levels that generally contribute much higher light extinction than the
15 NO₂ (i.e. NO₂ absorption is generally less than approximately 5% of the light extinction, except
16 where emission controls remove most of the PM prior to releasing the remaining gases to the
17 atmosphere). The remainder of this document focuses on the contribution of PM, which is
18 typically much greater than that of gases, to ambient light extinction, unless otherwise specified.

- 19 • **To what extent does the available evidence inform our understanding of the temporal**
20 **nature of the PM visibility effect, including relevant exposure periods, associated**
21 **atmospheric conditions, and diurnal patterns of exposure?**

22 Diurnal Periods of Interest

23 Typically, we think of visibility associated with daytime periods. We recognize,
24 however, that PM light extinction behaves the same at night as during the day, enhancing the
25 scattering of anthropogenic light, contributing to the “skyglow” within and over populated areas,
26 adding to the total sky brightness, and contributing to the reduction in contrast of stars against
27 the background. These effects produce the visual result of a reduction of the number of visible
28 stars and the disappearance of diffuse or subtle phenomena such as the Milky Way. The
29 extinction of starlight is a secondary and minor effect also caused by increased PM scattering
30 and absorption.

31 However, there are significant and important differences between daytime and nighttime
32 visual environments that potentially make the nighttime period inappropriate to address at this
33 time. First, daytime visibility has dominated the attention of those who have studied the
34 visibility effects of air pollution. As a result, little research has been conducted on nighttime
35 visibility and the state of the science is not yet comparable to that associated with daytime

1 visibility impairment. Second, in addition to air pollution, nighttime visibility is affected by the
2 addition of light into the sight path from numerous sources, including anthropogenic light sources
3 such as artificial outdoor lighting, which varies dramatically across space, and natural sources
4 including the moon, planets, and stars. Light sources and ambient conditions are typically five to
5 seven orders of magnitude dimmer at night than in sunlight. Moonlight, like sunlight, introduces
6 light throughout an observer's sight path at a constant angle. On the other hand, dim starlight
7 emanates from all over the celestial hemisphere while artificial lights are concentrated in cities
8 and illuminate the atmosphere from below. These different light sources will yield variable
9 changes in visibility as compared to what has been established for the daytime scenario, in which
10 a single source, the sun, is by far the brightest source of light. Third, the human psychophysical
11 response (e.g., how the human eye sees and processes visual stimuli) at night is expected to
12 differ (ISA, section 9.2.2).

13 Given the above, we do not believe that the science is available at this time to support
14 adequate characterization of nighttime PM light extinction effects. In addition, we are not aware
15 of preference or valuation studies providing information on public preferences for nighttime
16 visual air quality (VAQ). Thus, we limit our consideration of PM visibility impacts to daylight
17 hours only.

18 Exposure Durations of Interest

19 Very little is known about the role exposure duration plays in determining the
20 acceptability or unacceptability of a given level of VAQ on the public welfare. We do know
21 from preference and/or valuation studies that atmospheric visibility conditions can be quickly
22 assessed and preferences determined. These studies show that a momentary glance at an image
23 of a scene (i.e. less than a minute) is enough for study participants to consistently judge the
24 acceptability or unacceptability of the viewed visual air quality conditions. Outside these
25 controlled settings, we are unaware of any studies that characterize the extent to which different
26 frequencies and durations of exposure to visibility conditions contribute to the degree of public
27 welfare impact that occurs.

28 Recognizing the need for further research in this area, it is useful to consider a variety of
29 circumstances that are commonly expected to occur and to evaluate the potential impact of
30 visibility impairment on the public welfare based on the information we do have. In some
31 circumstances, such as infrequent visits to scenic vistas in natural or urban environments, people
32 are motivated specifically to take the opportunity to view a valued scene and are likely to do so
33 for many minutes to hours to appreciate various aspects of the vista they choose to view. In such
34 circumstances, the viewer may consciously evaluate how the VAQ at that time either enhances
35 or diminishes the experience or view. However, the public also has many more opportunities to

1 notice visibility conditions on a daily basis in settings associated with performing daily routines
2 (e.g. during commutes, while walking the dog, or when taking out the recyclables). These
3 scenes, whether iconic or generic, may not be consciously viewed for their scenic value, but their
4 VAQ may still affect a person’s sense of wellbeing. Research has demonstrated that people are
5 emotionally affected by low VAQ, that perception of pollution is correlated with stress,
6 annoyance and symptoms of depression, and that VAQ is deeply intertwined with a “sense of
7 place,” affecting peoples sense of the desirability of a neighborhood (ISA section 9.2.4). Though
8 we do not know the extent to which these emotional effects are linked to different periods of
9 exposure to poor VAQ, providing additional protection against short-term exposures to levels of
10 VAQ considered unacceptable in the context of the preference studies would be expected to
11 provide some degree of protection against the risk of loss in the public’s “sense of wellbeing”.

12 Some people have mostly intermittent opportunities on a daily basis (e.g. during morning
13 and/or afternoon commutes) to experience ambient visibility conditions as they spend much of
14 their time indoors without access to windows. For such people a view of poor VAQ during their
15 morning commute may provide their perception of the day’s visibility conditions until the next
16 time they venture outside during daylight hours later or perhaps the next day. Other people have
17 exposure to visibility conditions throughout the day, so that a day with multiple hours of
18 visibility impairment would likely be judged as having a greater impact on their wellbeing than a
19 day with just one such hour.

20 We have no information or studies on the fraction of the public that has only one or a few
21 opportunities to experience visibility during the day, or information or studies on the duration of
22 the effect on wellbeing from exposure to different durations of poor VAQ conditions. However,
23 it is logical to conclude that people with limited opportunities to experience visibility conditions
24 on a daily basis would receive the entire impact of the day’s VAQ based on the visibility
25 conditions that occur during the short time period when they can see it. Since this group could
26 be affected on the basis of observing VAQ conditions for periods as short as one hour or less,
27 and because in some areas the worst PM VAQ of the day occurs during the morning (see UFVA,
28 figure 3-12) when many people are commuting, we believe it is appropriate to consider
29 characterizing PM visibility conditions in terms of the worst or maximum hourly value during
30 daylight hours for each day for purposes of evaluating the adequacy of the current suite of
31 secondary standards.

32 For another group of observers, those who have access to visibility conditions often or
33 continuously throughout the day, the entire impact of the day’s visibility conditions may be
34 based on the varying visibility conditions they observe throughout the day. For this group, it
35 might be more appropriate to evaluate the adequacy of the current suite of secondary standards in
36 terms of all daylight hours in the day.

1 Temporal Variations of Visibility Impacts

2 While visibility conditions can change quickly (i.e., less than a minute), atmospheric
3 sight path averaged light extinction which is pertinent to visibility impacts generally changes
4 more slowly (i.e. tens of minutes generally). Sub-hourly variations in light extinction
5 determined at any point in the atmosphere are likely the result of small-scale spatial pollution
6 features (i.e. plumes) being transported by the wind across that point. At typical wind speeds
7 found in U.S. cities, an hour corresponds to a few tens of kilometers of air flowing past a point,
8 which is similar to visibility sight path lengths of interest in urban areas.

9 PM concentrations and light extinction in urban environments vary from hour to hour
10 throughout the day due to a combination of diurnal meteorological conditions and systematic
11 changes in emissions activity (e.g. rush hour traffic). Generally low mixing heights at night and
12 during the early morning hour tend to trap locally produced emissions, which are diluted as the
13 mixing height increases due to heating during the day. Low temperatures and high relative
14 humidity at night are conducive to the presence of ammonium nitrate particles and water growth
15 by hygroscopic particles compared with the generally higher temperatures and lower relative
16 humidity later in the day. These combine to make early morning the most likely time for peak
17 urban visibility impacts. Superimposed on such systematic time of day variations are the effects
18 of synoptic meteorology (i.e., those associated with changing weather) and regional-scale air
19 quality that can generate peak visibility impacts any time of day. The net effects of the
20 systematic urban and larger scale variations are that peak daytime PM light extinction can occur
21 any time of day though more often in early morning hours (UFVA section 3.4.2 and 3.4.3 and
22 shown in Figures 3-9, 3-10 and 3-12). Use of multi-hour averaging of PM light extinction would
23 suppress peak hourly conditions that are expected to represent peak visibility impacts.

24 Meteorological Causes of Visibility Impacts

25 Visibility is also reduced directly by the presence of precipitation and fog regardless of
26 the presence or absence of PM. A secondary PM NAAQS is not meant to protect against such
27 sources of visibility impairment that occur irrespective of PM levels. Therefore, one
28 consideration in the development of alternative standard forms (discussed below in section 4.3)
29 was the inclusion of a relative humidity screen in an effort to avoid misinterpretation of the direct
30 effect of meteorological conditions on visibility and those caused by PM air quality.

- 31 • **Based on currently available information, what range of levels of visibility impairment**
32 **is reasonable to consider in reaching judgments about the adequacy of the current**
33 **NAAQS?**

34 In order to identify levels of visibility impairment appropriate for consideration in setting
35 secondary PM NAAQS to protect the public welfare, we comprehensively examined information

1 that was available in this review regarding people’s stated preferences for acceptable and
2 unacceptable visual air quality.

3 Light extinction is an atmospheric property that by itself does not directly translate into a
4 public welfare effect. Instead, light extinction becomes meaningful in the context of the impact
5 of visibility on the human observer. This has been studied in terms of the acceptability or
6 unacceptability expressed for it by a human observer. The perception of the visibility impact of
7 a given level of light extinction occurs in conjunction with the associated characteristics and
8 lighting conditions of the viewed scene. Thus, a given level of light extinction may be perceived
9 differently by an observer looking at a different scene or the same scene with different lighting
10 characteristics. Likewise, different observers looking at the same scene with the same lighting
11 may have different preferences regarding the associated VAQ. When scene and lighting
12 characteristics are held constant, the perceived appearance of a scene (i.e., how well the scenic
13 features can be seen and the amount of visible haze) depends only on changes in light extinction.
14 This has been demonstrated using the WinHaze model that uses image processing technology to
15 apply user-specified changes in light extinction values to the same base photograph with set
16 scene and lighting characteristics.

17 Much of what we know about the acceptability of levels of visibility comes from survey
18 studies in which participants were asked questions about their preference or the value they place
19 on various visibility levels as displayed to them in scenic photographs and/or WinHaze images
20 with a range of known light extinction levels. Urban visibility preference studies for four urban
21 areas were reviewed in the UFVA (chapter 2) to assess the light extinction levels judged by the
22 participant to have acceptable visibility. While the results differed among the four urban areas,
23 results from a rating exercise showed that within each preference study, survey participants
24 consistently distinguish between different levels of light extinction and prefer and value visibility
25 associated with lower light extinction levels among the scenic images they are shown.

26 The reanalysis of urban preference studies included three completed urban visibility
27 preference survey studies plus a pair of smaller focus studies designed to explore and further
28 develop urban visibility survey instruments. The three western studies included one in Denver,
29 Colorado (Ely et al., 1991), one in the lower Fraser River valley near Vancouver, British
30 Columbia (BC), Canada (Pryor, 1996), and one in Phoenix, Arizona (BBC Research &
31 Consulting, 2003). A pilot focus group study was also conducted for Washington, DC (Abt
32 Associates Inc., 2001). In response to an EPA request for public comment on the Scope and
33 Methods Plan (74 FR 11580, March 18, 2009), we received comments (Smith, 2009) about the
34 results of a new Washington, DC focus group study that had been conducted using methods and
35 approaches similar to the method and approach employed in the EPA pilot study (Smith and
36 Howell, 2009). When taken together, these studies from the four different urban areas included a

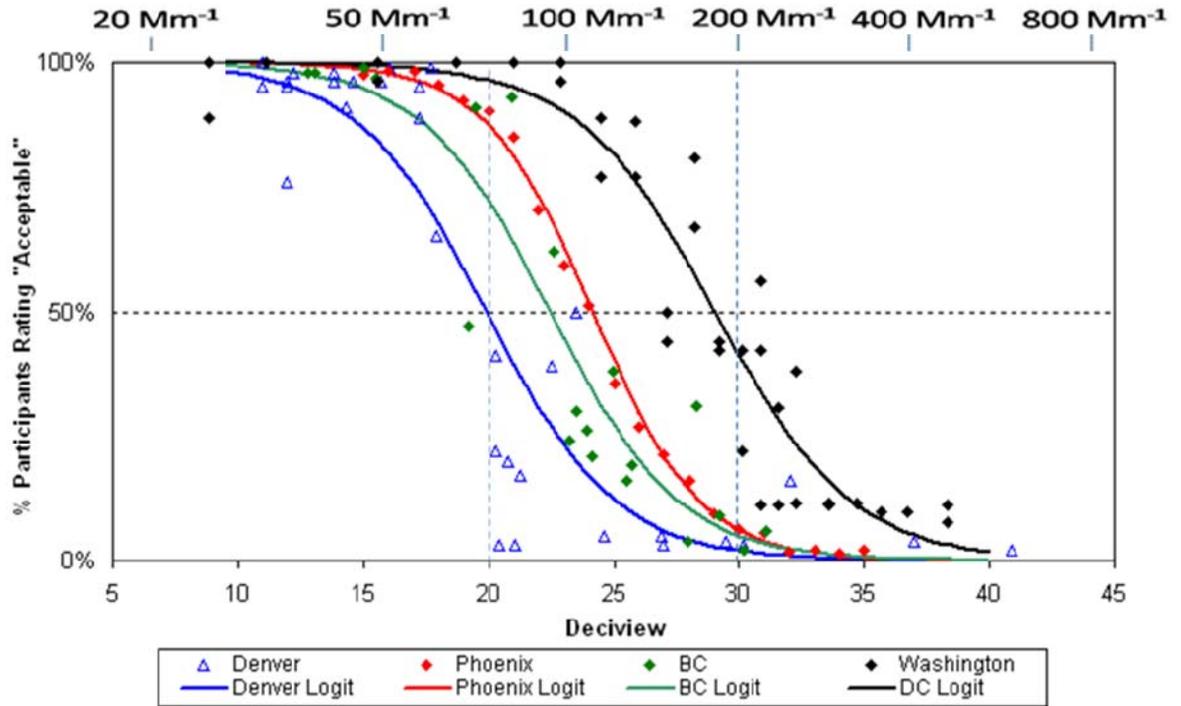
1 total of 852 individuals, with each individual responding to a series of questions answered while
2 viewing a set of images of various urban VAQ conditions.

3 The approaches used in the four studies are similar and are all derived from the method
4 first developed for the Denver urban visibility study. In particular, the studies all used a similar
5 group interview type of survey to investigate the level of visibility impairment that participants
6 described as “acceptable.” While each study asked the basic question, “What level of visibility
7 degradation is acceptable?”, the term “acceptable” was not defined, so that each person’s
8 response was based on his/her own values and preferences for VAQ. Given the similarities in
9 the approaches used, we concluded that it is reasonable to compare the results to identify overall
10 trends in the study findings and that this comparison can usefully inform the selection of a range
11 of levels for use in further analyses. However, variations in the specific materials and methods
12 used in each study introduce uncertainties that should also be considered when interpreting the
13 results of these comparisons. Key differences between the studies include: 1) image presentation
14 methods (e.g., projected slides of actual photos, projected images generated using WinHaze (a
15 significant technical advance in the method of presenting VAQ conditions), or use of computer
16 monitor screen; 2) number of participants in each study; 3) participant representativeness of the
17 general population of the relevant metropolitan area; and 4) specific wording used to frame the
18 questions used in the group interview process.

19 In the UFVA, each study was evaluated separately and figures developed to display the
20 percentage of participants that rated each photograph as “acceptable”. The horizontal axis was in
21 terms of light extinction (deciview or dv) and the vertical axis in terms of percent of participants
22 rating “acceptable”. Ely et al. (1991) introduced a “50% acceptability” criteria analysis of the
23 Denver preference study results. The 50% acceptability criteria is designed to identify the VAQ
24 level that best divides the photographs into two groups: those with a VAQ rated as acceptable by
25 the majority of the participants, and those rated not acceptable by the majority of participants.
26 We adopted the criteria as a useful index for comparison between studies. The results of each
27 individual analysis were then combined graphically to allow for visual comparison. Figure 4-2
28 (Figure 2-16 in UFVA) presents the graphical summary of the results of the studies in the four
29 cities and draws on results previously presented in Figures 2-3, 2-5, 2-7 and 2-11 of chapter 2 in
30 the UFVA. Figure 4-2 also contains lines at 20 dv and 30 dv that effectively and pragmatically
31 identify a range where the 50% acceptance criteria occur across all four of the urban preference
32 studies. Out of the 114 data points shown in Figure 4-2, only one photograph (or image) with a
33 VAQ below 20 dv was rated as acceptable by less than 50% of the participants who rated that

1 photograph.² Similarly, only one image with a VAQ above 30 dv was rated acceptable by more
 2 than 50% of the participants who viewed it.³
 3

Figure 4-2. Summary of results of urban visibility studies in four cities, showing the identified range of the 50% acceptance criteria .⁴



4
 5 As can be seen in the figure, each urban area has a separate and unique response curve
 6 that appears to indicate that it is distinct from the others. These curves are the result of a
 7 logistical regression analysis using a logit model of the greater than 19,000 ratings of haze
 8 images as acceptable or unacceptable. The model results can be used to estimate the VAQ
 9 deciview values where the estimated response functions cross the 50% acceptability level, as
 10 well as any alternative criteria levels. Selected examples of these are shown in Table 4-1 (Table
 11 2-4 in UFVA, Chapter 2). These results show that the logit model data also support the upper
 12 and lower ends of the range of 50th percentile acceptability values (e.g. near 20 dv for Denver
 13 and near 30 dv for Washington, DC) already identified in Figure 4-2 (see Table 2-4 UFVA).

² Only 47% of the BC participants rated a 19.2 dv photograph as acceptable.

³ In the 2001 Washington, D.C. study, a 30.9 dv image was used as a repeated slide. The first time it was shown 56% of the participants rated it as acceptable, and 11% rated it as acceptable the second time it was shown. The same VAQ level was rated as acceptable by 42% of the participants in the 2009 study (Test 1).

⁴ Top scale shows light extinction in inverse megameter units; bottom scale in deciviews. Logit analysis estimated response functions are shown as the color-coded curved lines for each of the four urban areas

Table 4-1. Logit model estimated VAQ values corresponding to various percent acceptability values for the four cities.

	Denver	British Columbia	Phoenix	Washington, DC
90% Acceptability criteria	14.21	16.80	24.15	23.03
75% Acceptability criteria	17.05	19.63	21.80	26.03
50% Acceptability Criteria	19.90	22.45	24.15	29.03
25% Acceptability criteria	22.74	25.28	26.51	32.03
10% Acceptability criteria	25.59	28.10	28.87	35.03

Based on the composite results and the effective range of 50th percentile acceptability across the four urban preference studies shown in Figure 4-2 and Table 4-1, benchmark levels have been selected in a range from 20 dv to 30 dv (74 Mm⁻¹ to 201 Mm⁻¹) for the purpose of provisionally assessing whether visibility conditions would be considered acceptable (i.e., less than the low end of the range, unacceptable (i.e., greater than the high end of the range, or potentially acceptable (within the range). A midpoint of 25 dv (122 Mm⁻¹) was also selected for use in the assessment. This level is also very near to the 50th percentile criteria value from the Phoenix study (i.e. 24.3 dv), which is by far the best of the four studies in terms of least noisy preference results and the most representative selection of participants. Based on the currently available information, we conclude that the use of 25 dv to represent the middle of the distribution of results seems well supported.

These three benchmark values provide a low, middle, and high set of light extinction conditions that are used to provisionally define daylight hours with urban haze conditions that have been judged unacceptable by the participants of these preference studies. As discussed above, PM light extinction is taken to be light extinction minus the Rayleigh scatter (i.e. light scattering by atmospheric gases which is on average about 10 Mm⁻¹), so the low, middle and high levels correspond to PM light extinction levels of about 64 Mm⁻¹, 112 Mm⁻¹ and 191 Mm⁻¹. In the UFVA, these three levels were called Candidate Protection Levels (CPLs). We continue to use this term in this document. However, it is important to note that the degree of protection provided by a secondary NAAQS is not determined solely by the level of the standard but by all the components (e.g., indicator, form, level, averaging time) being applied together. Therefore, the reader should keep in mind that the term CPL is meant only to indicate levels within a range that we feel are appropriate for consideration that could, in conjunction with other aspects of the standard, provide an appropriate degree of visibility protection.

1 In characterizing our degree of confidence in each CPL and across the range, a number of
2 issues were considered. Looking first at the two studies that define the upper and lower bounds
3 of the range, we considered whether they represent a true regional distinction in preferences for
4 urban visibility conditions between western and eastern U.S. There is little information available
5 to help evaluate this, especially given that we have preference studies in only one eastern urban
6 area. Smith and Howell (2009) found little difference in preference response to Washington, DC
7 haze photographs between the study participants from Washington, DC and those from Houston,
8 TX. This provides some limited evidence that the value judgment of the public in different areas
9 of the country may not be an important factor in explaining the differences in these study results.

10 In further considering what factors could explain the observed differences in preferences
11 across the four urban areas, we noted that the urban scenes used in each study had different
12 characteristics. For example, each of the western urban visibility preference study scenes
13 included mountains in the background while the single eastern urban study did not. It is also true
14 that each of the western scenes included objects at greater distances from the camera location
15 than in the Washington, DC study. There is no question that objects at a greater distance have a
16 greater sensitivity to perceived visibility changes as light extinction is changed compared to
17 otherwise similar scenes with objects at a shorter range. This alone might explain the difference
18 between the results of the Washington DC, study and those from the Western urban studies.
19 Also, it seems likely that people value the views of mountains in the background more than
20 generic distant buildings in the foreground of the western scenes; just as it seems likely that the
21 Capital Mall and Washington Monument were the likely objects of greatest interest for the
22 Washington, DC study base photograph. Having scenes with the object of greatest intrinsic
23 value nearer and hence less sensitive for Washington compared with more distant objects of
24 greatest intrinsic value in the western urban areas could further explain the difference in
25 preference results.

26 Another question that we considered was whether the high CPL value that is based on the
27 Washington DC preference results is likely to be generally representative of urban areas that do
28 not have associated mountains or other valued objects visible in the distant background. Such
29 areas would include the middle of the country and many areas in the eastern US. In order to
30 examine this issue, an effort would have to be made to see if scenes in such areas could be found
31 that would be generally comparable to the western scenes (e.g., contain valued scenic elements at
32 more sensitive distances than that used in the Washington, DC study). This is only one of a
33 family of issues concerning how exposure to urban scenes of varying sensitivity affects public
34 perception, for which no information is currently available. Additional urban visibility
35 preference studies employing images selected of potentially more sensitive scenes could help
36 evaluate whether a lower value for the high CPL is supportable. Other investigations to

1 determine how common such scenes are in various regions of the country would also be
2 informative. Until such information becomes available, the high end of the CPL range (30 dv)
3 seems to be an appropriate level to consider.

4 With respect to the low end of the range, we considered factors that might further refine
5 our understanding of the robustness of this level. We concluded that additional urban preference
6 studies, especially with a greater variety in types of scenes, including potentially more sensitive
7 western urban scenes, could help evaluate whether a lower CPL value than the currently selected
8 20 dv is supportable. Further, the reason for the noisiness in data points around the curves
9 apparent in both the Denver and British Columbia results compared to the smoother curve fit of
10 Phoenix study results could be explored. One possible explanation that we identified is that
11 these older studies used photographs taken at different times of day and on different days to
12 capture the range of light extinction levels needed for the preference studies. In contrast, the use
13 of WinHaze in the Phoenix (and Washington, DC) study reduced variations in scene appearance
14 that affects preference rating and avoided the uncertainty inherent in using ambient
15 measurements to represent sight path averaged light extinction values. Reducing these sources
16 of noisiness and uncertainty in the results of future studies of sensitive urban scenes could
17 provide more certainty in the selection of a low CPL value.

- 18 • **To what extent does the available information demonstrate or suggest that PM-related**
19 **visibility impairment (within the range of CPLs) is occurring at current ambient**
20 **conditions or at levels that would meet the current standards?**

21 Current Visibility Levels

22 Chapter 3 of the UFVA characterized current visibility conditions in terms of both PM_{2.5}
23 and light extinction levels for the 14 urban areas⁵ selected and compared them to the CPLs
24 identified above.

25 As an initial matter, we note that visibility impairment during periods with fog or
26 precipitation occurs irrespective of the presence or absence of PM. In order to avoid
27 precipitation and fog confounding estimates of PM visibility impairment, and as advised by
28 CASAC as part of its comments on the first public review draft of the UFVA, we restricted our
29 assessment of visibility conditions to daylight hours with relative humidity less than or equal to
30 90% (UFVA section 3.3.5 and Appendix G). However, not all periods with relative humidity
31 above 90% have fog or precipitation. Removing those hours from application of a secondary PM

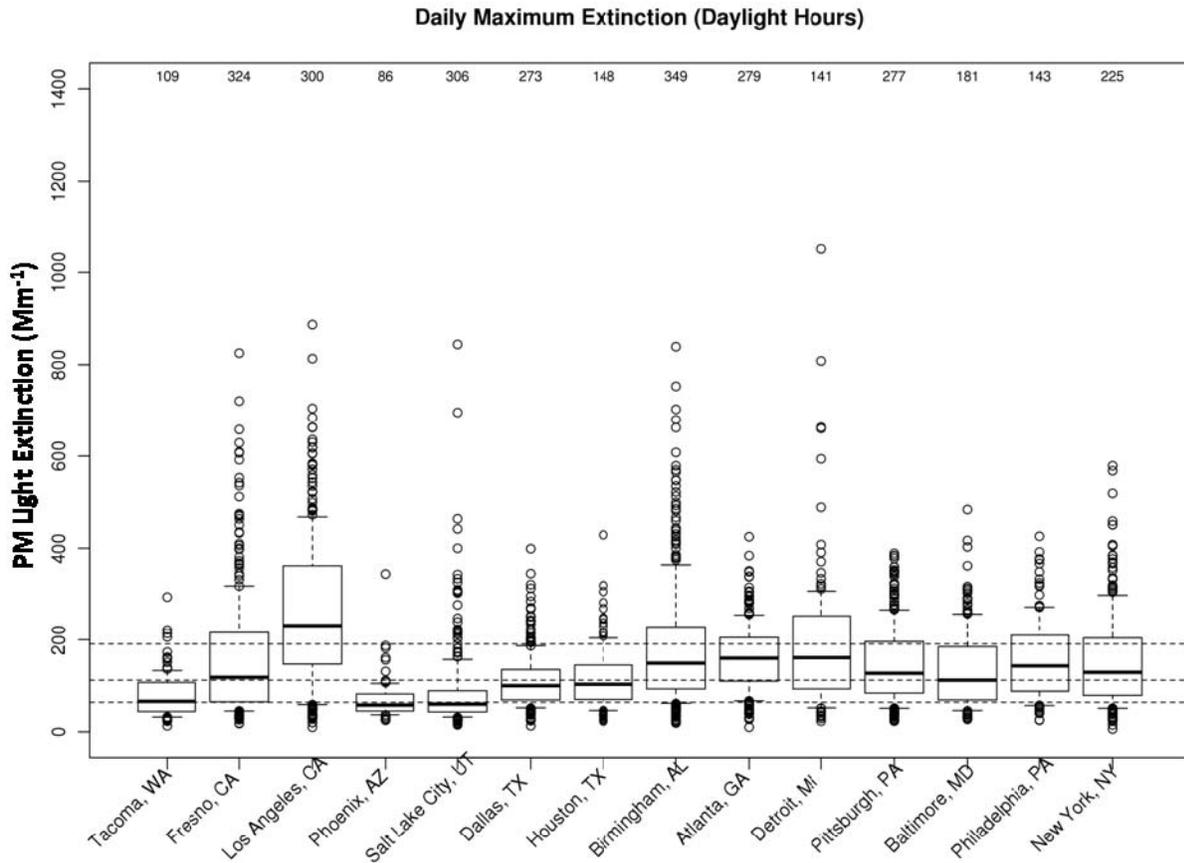
⁵ Comments on the second review draft of the UFVA from those familiar with the monitoring sites in St. Louis indicate that the site selected to provide continuous PM₁₀ monitoring, though less than a mile from the site of the PM_{2.5} data is not representative of the urban area and resulted in unrealistically large PM_{10-2.5} values. The EPA staff considers these comments credible and removed the St. Louis assessment results, leaving 14 of the 15 urban areas from the UFVA to be included in the Policy Assessment.

1 standard involves a tradeoff between the benefits of avoiding many of the hours with
2 meteorological causes of visibility impacts and the cost of not counting some hours with high
3 relative humidity without fog or precipitation, where the growth of hygroscopic PM into large
4 solution droplets results in enhanced PM visibility impacts. For the 14 urban areas included in
5 the assessment for which updated meteorological data were obtained, a 90% relative humidity
6 cutoff criterion is effective in that on average less than 6% of the hours are removed from
7 consideration, yet those hours have on average ten times the likelihood of rain, six time the
8 likelihood of snow/sleet, and 34 times the likelihood of fog compared with hours with 90% or
9 lower relative humidity.

10 Figure 4-3 (Figure 3-8 in UFVA) presents box-and-whisker plots to illustrate the
11 distributions of the estimates of daylight 1-hour reconstructed PM₁₀ light extinction levels in
12 each area during the 2005-2007 time period. The distribution of the daily maximum 1-hour
13 values is shown. The horizontal dashed lines in the plots represent the low, middle, and high
14 CPLs for PM₁₀ light extinction of 64, 112, and 191 Mm⁻¹, corresponding to the benchmark VAQ
15 values of 20 dv, 25 dv and 30 dv as discussed above. Table 4-2 (Table 3-7 in UFVA) provides
16 the percentages of days (across all of 2005-2007, unweighted) in which the daily maximum
17 daylight 1-hour PM₁₀ light extinction level was greater than each of the three CPLs (excluding
18 hours with relative humidity greater than 90 percent).

19 From these displays it can be seen that among the 14 urban areas, those in the East and in
20 California tend to have a higher frequency of visibility conditions above the high CPL compared
21 with those in the western US. Both Figure 4-3 and Table 4-2 indicate that all 14 urban areas
22 have daily maximum hourly PM₁₀ light extinctions that exceed even the highest CPL some of the
23 time. Again, the non-California western urban locations have the lowest frequency of maximum
24 hourly PM₁₀ light extinction with values in excess of the high CPL for 8 percent or fewer of the
25 days. Except for the two Texas and the non-California western urban areas, all of the other
26 urban areas exceed that high CPL from about 20 percent to over 60 percent of the days. Based
27 on these estimated maximum hourly PM₁₀ light extinction estimates, all 14 of the urban areas
28 exceed the low CPL for about 40 percent to over 90 percent of the days. Based on all of the
29 above, we conclude that current levels of PM₁₀ light extinction associated with recent PM air
30 quality exceed levels that could reasonably be considered as protective of the public welfare.

1 **Figure 4-3. Distribution of estimated maximum daily daylight 1-hour PM₁₀ light extinction**
 2 **across the 2005-2007 period, by study area (excluding hours with relative humidity greater**
 3 **than 90 percent). (Adapted from Figure 3-8 in UFVA)***



4
 5 **In the box-and-whisker plot, the box represents the 25th to 75th percentile range and the whiskers*
 6 *represent the 10th and 90th percentile points of the data; individual data points below the 10th percentile*
 7 *and above the 90th percentile are graphed as small circles. The three dashed horizontal lines represent*
 8 *the three CPL levels of 64, 112, and 191 Mm⁻¹.*

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Table 4-2. Percentage of daily maximum hourly values of daylight PM₁₀ light extinction exceeding CPLs (excluding hours with relative humidity greater than 90 percent). (adapted from Table 3-7 in UFVA)

Study Area	Number of Days with Estimates	Candidate Protection Level		
		64 Mm ⁻¹	112 Mm ⁻¹	191 Mm ⁻¹
		(a) Percentage of Daily Maximum Hourly Values Exceeding CPL		
Tacoma	109	52	22	4
Fresno	324	75	52	30
Los Angeles	300	90	83	62
Phoenix	86	42	7	1
Salt Lake City	306	44	17	8
Dallas	273	80	41	10
Houston	148	79	45	11
Birmingham	349	89	65	34
Atlanta	279	91	75	31
Detroit	141	87	68	43
Pittsburgh	277	85	57	26
Baltimore	181	80	50	23
Philadelphia	143	86	64	31
New York	225	83	59	28
<i>Average</i>	<i>224</i>	<i>75</i>	<i>50</i>	<i>25</i>

6 Visibility Levels That Just Meet Current Standards

7 In the UFVA, we modeled the “what if” scenario based on simulating just meeting the
 8 current suite of PM_{2.5} secondary standards: 15 µg/m³ annual average PM_{2.5} concentration and 35
 9 µg/m³ 24-hour average PM_{2.5} concentration with a 98th percentile form, averaged over three
 10 years. The steps needed to model the “what if” conditions involve explicit consideration of
 11 changes in PM_{2.5} components and are described here. First, we applied proportional rollback to
 12 all the PM_{2.5} monitoring sites in each study area, taking into account policy relevant background
 13 (PRB) PM_{2.5} mass, to “just meet” the NAAQS scenario for the area as a whole, not just at the
 14 visibility assessment study site. The health risk assessment document (EPA 2010a) describes
 15 this procedure in detail. The degree of rollback is controlled by the highest annual or 24-hour
 16 design value, which in most study areas is from a site other than the site used in this visibility
 17 assessment. The relevant result from this analysis is the percentage reduction in non-PRB PM_{2.5}
 18 mass need to “just meet” the NAAQS scenario, for each study area. These percentage reductions
 19 are shown in Table 4-4 of the UFVA. Note that Phoenix and Salt Lake City meet the 15/35

1 NAAQS scenario under current conditions and require no reduction. PM_{2.5} levels in these two
2 cities were not “rolled up.” Second, for each day and hour for each PM_{2.5} component, we
3 subtracted the PRB concentration from the current conditions concentration, to determine the
4 non-PRB portion of the current conditions concentration. Third, we applied the percentage
5 reduction from step 1 to the non-PRB portion of each of the five PM_{2.5} components and added
6 back the PRB portion of the component. Finally, we applied the IMPROVE algorithm, using the
7 reduced PM_{2.5} component concentrations, the current conditions PM_{10-2.5} concentration for the
8 day and hour, and relative humidity for the day and hour to calculate the PM₁₀ light extinction.
9 We then included the term for Rayleigh scattering.

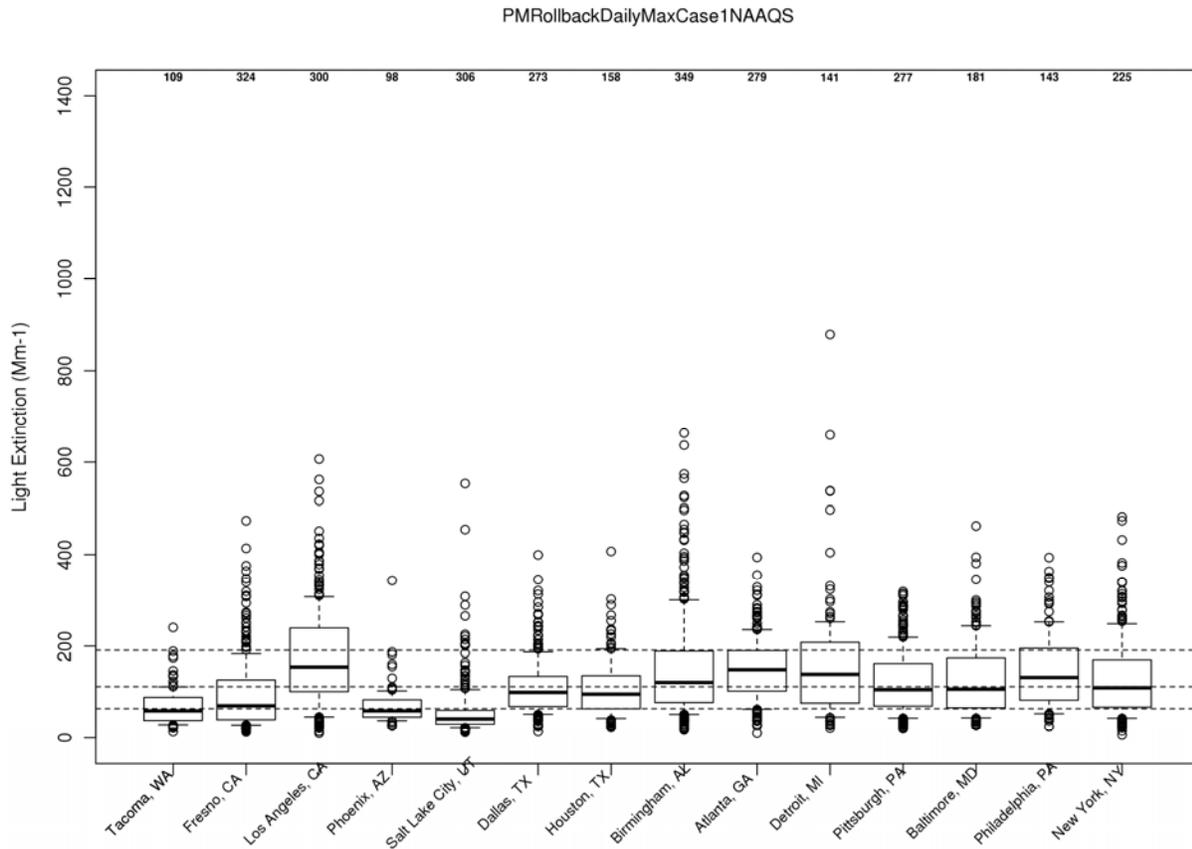
10 Figure 4-4 and Table 4-3 display the results of the rollback procedure as a box and
11 whisker plot of daily maximum daylight 1-hour PM₁₀ light extinction and the percentage of daily
12 maximum hourly PM₁₀ light extinction values exceeding the CPLs when just meeting the current
13 PM_{2.5} NAAQS scenario of 15/35 µg/m³ (excluding hours with relative humidity greater than 90
14 percent. These displays show that at the current PM NAAQS level (i.e., 15/35) all of the eastern
15 urban areas and Los Angeles exceed the least restrictive CPL more than 10% of the time and that
16 only Tacoma would not exceed the least restrictive CPL more than 2% of the time.

1

2 **Figure 4-4. Distribution of daylight 1-hour PM₁₀ light extinction when rolled back to just**
3 **meet current PM fine NAAQS across the 2005-2007 period, by study area (excluding hours**
4 **with relative humidity greater than 90 percent). ***

5 **NAAQS Scenario: 15 µg/m³ annual; 35 µg/m³ 24-hour**

6 **Displayed: Daily Max Daylight Light Extinction (excluding hours >90% RH)**



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9 ** In the box-and-whisker plot, the box represents the 25th to 75th percentile range and the*
10 *whiskers represent the 10th and 90th percentile points of the data; individual data points below the 10th*
11 *percentile and above the 90th percentile are graphed as small circles. The three dashed horizontal lines*
12 *represent the three CPL levels of 64, 112, and 191 Mm⁻¹*

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Table 4-3 Percentage of daily maximum hourly values of daylight PM₁₀ light extinction exceeding CPLs when “just meeting” the current PM_{2.5} NAAQS (15/35 µg/m³) (excluding hours with relative humidity greater than 90 percent). (Adapted from Table 4-7 in UFVA).

Study Area	Number of Days with Estimates	Candidate Protection Level		
		64 Mm ⁻¹	112 Mm ⁻¹	191 Mm ⁻¹
		(a) Percentage of Daily Maximum Hourly Values Exceeding CPL		
Tacoma	109	43	10	1
Fresno	324	54	30	10
Los Angeles	300	85	69	39
Phoenix	86	44	6	1
Salt Lake City	306	24	9	4
Dallas	273	81	41	10
Houston	148	75	41	11
Birmingham	349	84	55	24
Atlanta	279	90	71	25
Detroit	141	80	61	33
Pittsburgh	277	78	48	16
Baltimore	181	78	48	19
Philadelphia	143	85	61	28
New York	225	76	45	19
Average	224	70	43	17

6 **4.2.2 Summary**

7 In summary, we conclude that the available information in this review, as described
8 above and in the UFVA and ISA, clearly calls into question the adequacy of the current suite of
9 PM_{2.5} standards in the context of public welfare protection from visibility impairment, primarily
10 in urban areas, and supports consideration of alternative standards to provide appropriate
11 protection.

12 This conclusion is based first on the large percentage of days that exceed the range of
13 CPLs identified for consideration under both current and just meet PM air quality conditions. In
14 particular, under just meet conditions for the suite of secondary PM NAAQS (i.e. 15/35 µg/m³)
15 greater than 10% of the days exceed the highest, least protective CPL of 191 Mm⁻¹ for 9 of the
16 14 urban areas. When the middle CPL of 112 Mm⁻¹ is considered, 11 of the 14 cities have
17 greater than 10% of their days exceeding with a range of 30 to 74% of days exceeding this level.

1 At the lowest CPL of 64 Mm^{-1} , the percentage of days exceeding this level range from 24 to 90%
2 across the 14 urban areas.

3 Second, we conclude that the averaging times associated with the current suite of $\text{PM}_{2.5}$
4 NAAQS are not well suited to protect against PM-related visibility impairment on the basis that,
5 in the context of the studies, short-term exposure (e.g., 1 hour or less) is sufficient for an
6 unacceptable level of visual air quality to be observed and the associated impacts of that
7 observation to be registered by the observer. Since some portion of the population may only
8 have the opportunity to observe one hour or less of ambient daylight visibility conditions, relying
9 on an averaging time as long as that of the current 24-hour and annual $\text{PM}_{2.5}$ standards would
10 make it difficult to identify a requisite level of protection that would translate into appropriate
11 protection against the maximum daily value. In addition, these longer averaging times also result
12 in the inclusion of nighttime conditions, for which the science needed to support identification of
13 appropriate levels of visibility protection is not well developed or understood.

14 In reaching the conclusion that the current suite of $\text{PM}_{2.5}$ standards is inadequate to
15 provide the appropriate protection of the public welfare from known and/or anticipated adverse
16 effects by calling into question the adequacy of the current levels and averaging times, we also
17 considered whether the current indicator of $\text{PM}_{2.5}$ remains useful in relating ambient PM to the
18 public welfare effect of visibility impairment. Section 4.3 below discusses these and other
19 considerations in its discussion of alternative standards for consideration.

20 **4.3 CONSIDERATION OF ALTERNATIVE STANDARDS**

21 Having reached the conclusion that just meeting the current suite of $\text{PM}_{2.5}$ standards
22 continues to allow levels of visual air quality impairment that, based on the scientific evidence
23 and information available in this review, can reasonably be considered adverse to the public
24 welfare, this section will discuss alternative standards that could potentially provide requisite
25 public welfare protection from known and/or anticipated adverse effects. This section discusses
26 indicator, averaging time and form, including how they relate to the characteristics of the
27 visibility effect of interest.

28 **4.3.1 Nature of the Indicator**

- 29 • **To what extent does currently available information provide support for considering a**
30 **different indicator(s) for PM to replace or supplement the $\text{PM}_{2.5}$ mass-based indicator?**

31 As described below, EPA staff has considered three alternative indicators: $\text{PM}_{2.5}$ mass,
32 directly measured $\text{PM}_{2.5}$ light extinction, and speciated $\text{PM}_{2.5}$ mass-calculated light extinction.
33 $\text{PM}_{2.5}$ mass is the same indicator as is used by the current suite of $\text{PM}_{2.5}$ NAAQS. Directly
34 measured $\text{PM}_{2.5}$ light extinction is a measurement of the light extinction caused by $\text{PM}_{2.5}$ under
35 ambient conditions. Speciated $\text{PM}_{2.5}$ mass-calculated light extinction uses the IMPROVE

1 algorithm to calculate $PM_{2.5}$ light extinction using measured dry $PM_{2.5}$ speciation and relative
2 humidity data. We believe that consideration of the use of either directly measured $PM_{2.5}$ light
3 extinction or speciated $PM_{2.5}$ mass-calculated light extinction as an indicator is justified because
4 light extinction is a physically meaningful measure of the ambient $PM_{2.5}$ characteristic that is
5 most relevant and directly related to visibility effects. Further, $PM_{2.5}$ is the component of PM
6 responsible for most of the visibility impairment in most urban areas. In these areas, the
7 contribution of $PM_{10-2.5}$ is a minor contributor to visibility impairment most of the time,
8 although at some locations (see UFVA Figure 3-13 for Phoenix) $PM_{10-2.5}$ can be a major
9 contributor to urban visibility effects. In the absence of PM air quality information from a much
10 larger number of urban areas across the country, it is not possible at this time to know how many
11 urban areas fall into this category, though it is reasonable to presume that other urban areas in the
12 desert southwestern region of the country may have conditions similar to the conditions shown
13 for Phoenix. A possible solution to this issue would be to add a companion $PM_{10-2.5}$ standard to
14 provide control under those circumstances. Insufficient information is available at this time to
15 consider this latter refinement.

16 The basis for considering each of these three indicators is discussed below. The
17 discussion also addresses monitoring data requirements for the directly measured $PM_{2.5}$ light
18 extinction and for speciated $PM_{2.5}$ mass-calculated light extinction.

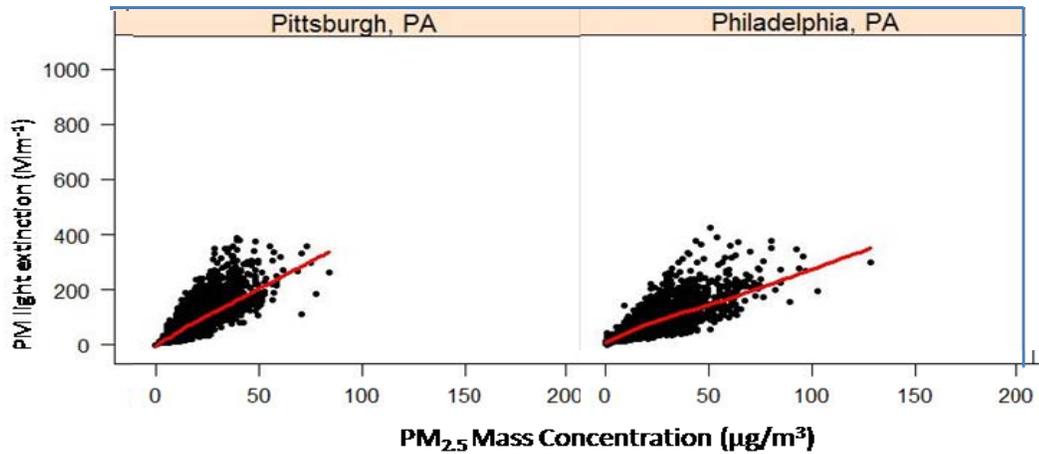
19 $PM_{2.5}$ Mass Indicator

20 $PM_{2.5}$ mass monitoring methods are in wide spread use, including the Federal Reference
21 Method (FRM) involving the collection of periodic (1 day in 6 and 1 day in 3) 24-hour filter
22 samples. These samples are then subsequently analyzed to determine 24-hour PM fine mass.
23 The Chemical Speciation Network (CSN) and continuous $PM_{2.5}$ monitoring produce hourly
24 average mass concentrations and are conducted at many locations. At a few locations,
25 continuous speciation sampling produces measures of some of the major $PM_{2.5}$ chemical
26 components. These routine monitoring activities do not include measurement of the water
27 content of the ambient PM that contributes, often significantly, to visibility impacts. Further, the
28 PM mass concentration monitoring does not provide information on the composition of the
29 ambient PM which also contributes to the variability in the amount of visibility impact associated
30 with any ambient PM mass concentration.

31 The overall performance of $PM_{2.5}$ mass as a predictor of visibility effects as indicated by
32 PM_{10} light extinction can be seen in scatter plots shown in Figure 4-5 for Pittsburgh and
33 Philadelphia, PA (Similar plots for all 14 urban areas are in Appendix D, Figure D-2 of the
34 UFVA). These demonstrate the variation in hourly PM_{10} light extinction corresponding to any

1 specific level of PM_{2.5} mass concentration as well as statistical differences of the average
2 relationships (depicted as the best fit lines) between cities.

3 **Figure 4-5 Scatter plots of PM₁₀ light extinction versus PM_{2.5} mass concentration of two**
4 **cities (from UFVA Appendix D, Figure D2).**



5
6 Directly Measured PM_{2.5} Light Extinction Indicator

7 PM light extinction⁶ is the major component of or contributor to light extinction, which is
8 the property of the atmosphere that is most directly related to visibility effects. It differs from
9 light extinction by the nearly constant contributions for Rayleigh (or clean air) light scattering
10 and the minor contributions by NO₂ light absorption. The net result is that PM light extinction
11 has a one-to-one relationship to light extinction, unlike PM_{2.5} mass concentration which, as
12 shown above, does not have this characteristic. PM_{2.5} light extinction⁷ can be directly measured.
13 Direct measurement of PM_{2.5} light extinction can be accomplished using several instrumental
14 methods, some of which have been used for decades to routinely monitor the two components of
15 PM_{2.5} light extinction (light scattering and absorption) or to jointly measure both as total light
16 extinction (from which Rayleigh scattering is subtracted to get PM_{2.5} light extinction). There are
17 a number of advantages to direct measurements for use in a secondary standard relative to
18 calculated PM_{2.5} light extinction that was used to generate hourly values for the UFVA. These
19 include the greater accuracy of direct measurements with shorter averaging times and the overall

⁶ PM light extinction is used here to refer to the light extinction caused by PM regardless of particle size, so it includes the contributions from particle larger than 10µm that are additive to the PM₁₀ light extinction.

⁷ PM_{2.5} light extinction is the contribution to PM light extinction by particles sampled through an inlet with a particle size 50% cutpoint of 2.5µm diameter.

1 greater simplicity when compared to the need for measurements of multiple parameters needed
2 to generate the calculated PM light extinction level for a standard.

3 In evaluating whether direct measurement of PM₁₀ light extinction is appropriate to
4 consider in the context of this PM NAAQS review, EPA produced a White Paper on Particulate
5 Matter (PM) Light Extinction Measurements (US EPA, 2010f), and solicited comment on the
6 white paper from the Ambient Air Monitoring and Methods Subcommittee (AAMMS) of
7 CASAC. In its review (Russell and Samet, 2010) of the white paper, the AAMMS made the
8 recommendation to EPA that direct measurement be limited to PM_{2.5} light extinction as this can
9 be accomplished by a number of commercially available instruments and because PM_{2.5} is
10 generally responsible for most of the PM visibility impairment in urban areas. They indicated
11 that it is technically more challenging at this time to accurately measure the PM_{10-2.5} component
12 of light extinction.

13 Speciated PM_{2.5} Mass-Calculated Light Extinction Indicator

14 PM_{2.5} light extinction can be estimated from PM_{2.5} mass and speciation data plus relative
15 humidity data, as is presently routinely done on a 24-hour average basis under the Regional Haze
16 Program using data from the rural IMPROVE monitoring network. Applying a conceptually
17 similar approach on an hourly basis using data from the urban CSN monitoring network would
18 permit the use of routinely collected data from the current CSN network, and possibly similar
19 new monitoring sites, as input to a simple algorithm to calculate PM_{2.5} light extinction for use in
20 a secondary PM NAAQS.

21 The approach used to generate hourly PM₁₀ light extinction for the UFVA was one
22 particular implementation of such a conceptual approach. It involved the use of the original
23 IMPROVE algorithm with estimates of hourly PM_{2.5} components derived from day-specific 24-
24 hour and hourly measurements of PM_{2.5} mass, 24-hour measurements of PM_{2.5} composition, and
25 (for some but not all study sites) hourly PM_{10-2.5} mass, along with daily temperature data and
26 hourly relative humidity information (UFVA, Section 3.3).⁸ The UFVA approach entailed
27 numerous and complex data processing steps to generate hourly PM composition information
28 from these less time-resolved data, including application of the SANDWICH approach to adjust
29 for nitrate losses and to estimate organic carbonaceous material via mass balance. EPA staff
30 employed complex custom software to do these data processing steps. The complexity of the
31 approach used was reasonable for assessment purposes at 15 urban areas. While the UFVA
32 approach conceivably could also be the basis for defining a national visibility standard, a simpler
33 approach would have greater transparency, and simplicity, and thus should be considered for that

⁸ Daily temperature data enters the UFVA approach via the SANDWICH method. This fact was not highlighted in the UFVA.

1 purpose. Therefore, we evaluated the degree to which simpler approaches would work as well as
2 the more complex method used in the UFVA. This evaluation (described briefly below and in
3 more detail in Appendix 4B, especially Table 4B-2) demonstrated that the PM₁₀ light extinction
4 values developed for the UFVA can be well approximated using the same IMPROVE algorithm
5 applied to hourly PM_{2.5} composition values that were much more simply generated in locations
6 where PM_{10-2.5} does not contribute significantly to PM VAQ.

7 The simplified approach favored by EPA staff (i.e., approach F in Appendix 4B), is
8 aimed at calculating hourly PM_{2.5} light extinction using the original IMPROVE equation (see
9 section 4.2.1 above) excluding the Rayleigh term for light scattering by atmospheric gases (i.e.
10 10 Mm⁻¹) and the term for light scattering by PM_{10-2.5} (i.e., 0.6 x [Coarse Mass]). This leaves
11 five PM_{2.5} species concentration terms (i.e., sulfate, nitrate, organic carbonaceous mass,
12 elemental carbon, and fine soil/crustal), plus relative humidity that need to be determined on an
13 hourly basis from monitoring data. Table 4B-2 in Appendix 4B describes ten steps in this
14 simplified approach (denoted approach F in Appendix 4B) in detail, which are summarized here:

- 15 i. Estimate 24-hour organic carbonaceous mass from the CSN organic carbon
16 measurement, via a multiplier, rather than by the SANDWICH method.
- 17 ii. Calculate 24-hour fine soil/crustal from the CSN measurements of the related PM_{2.5}
18 crustal elements.
- 19 iii. Calculate 24-hour neutralized, dry sulfate and nitrate in the same manner as in the
20 IMPROVE network and Regional Haze program, by multiplication by factors of
21 1.375 and 1.29 respectively.
- 22 iv. Sum the five components, for each day.
- 23 v. Calculate the five component fractions (i.e., percentages) of the sum-of-five, for
24 each day.
- 25 vi. Looking across all CSN sampling days in a given calendar month, average the daily
26 component fractions for sulfate, for nitrate, etc. These monthly-average component
27 fractions will be applied on every day in that month.
- 28 vii. For each day, determine a scaling factor to reconcile hourly PM_{2.5} mass
29 measurements from a continuous instrument with the 24-hour PM_{2.5} mass
30 determined by FRM sampling.⁹
- 31 viii. For each day, apply this scaling factor to the 1-hour measurements of PM_{2.5} mass
32 from the continuous instrument.
- 33 ix. For each day and daylight hour in the month, multiply the scaled value of hourly
34 PM_{2.5} mass by the component fractions from step (v).

⁹ Steps (vii) and (viii) were performed in the analysis that is presented in Appendix 4B, but perhaps would not need to part of the approach for actual implementation. The Comments column of Table 4B-2 discusses this aspect in more detail.

- 1 x. These hourly speciated PM_{2.5} concentrations are then combined with collocated
2 hourly averaged relative humidity as input to the IMPROVE algorithm to calculate
3 the speciated PM_{2.5} mass-calculated light extinction.

4 A more detailed description of the sources of the data and steps required to determine
5 speciated PM_{2.5} mass-calculated light extinction by this approach is contained in Appendix 4B
6 (see Table 4B-2). Also, Table 4B-1 of Appendix 4B compares/contrasts this approach with the
7 UFVA approach and with another approach of intermediate complexity. Use of this approach
8 would require establishing Federal Reference Methods for sampling and laboratory analysis for
9 each of the PM_{2.5} components that is measured, which could be based on existing CSN
10 procedures, including the IMPROVE-like procedures for elemental carbon and organic carbon
11 that have recently been fully implemented in the CSN network.

12 The PM_{2.5} light extinction values generated by using this simplified approach are
13 comparable to those developed for use in the UFVA as indicated by the regression statistics for
14 scatter plots of the paired data (i.e., slopes of the regression equation and R² values are near 1 as
15 show in Tables 4B-3 and 4B-4 in Appendix 4B). We believe that this simplified approach
16 provides reasonably good estimates of PM_{2.5} light extinction.

17 Comparison of PM Components: PM_{2.5} Mass and PM₁₀ Light Extinction Indicators

18 Given our understanding of the causal relationship between ambient PM and PM visual
19 air quality and the contribution of various PM components to PM light extinction under ambient
20 conditions, an environmentally-relevant indicator for PM VAQ should be able to appropriately
21 order the hours/days by the magnitude of PM VAQ impairment. One benefit of selecting such
22 an indicator would be to identify the components of PM that are most important for improving
23 PM VAQ in a given area.

24 Because the PM component contributions to PM₁₀ light extinction are weighted
25 according to their light extinction efficiency while their contributions to PM_{2.5} mass are equally
26 weighted (i.e., the sum of the component concentrations is the total PM_{2.5} mass concentration),
27 our initial hypothesis was that the choice of indicator could significantly influence the relative
28 contributions by PM components for the percent of hours selected above the level of a standard.

29 In order to evaluate this hypothesis, EPA staff conducted an assessment which explored
30 the PM₁₀ components contributing to the top 10% of the maximum daily 1-hour and top 2% of
31 all hours for PM_{2.5} mass and PM₁₀ light extinction (see Appendix 4C, Indicator Comparisons and
32 Component Apportionment Comparisons). While this comparison used calculated PM₁₀ light
33 extinction instead of the directly measured PM_{2.5} light extinction or speciated PM_{2.5} mass-
34 calculated light extinction indicators described above, we believe that comparisons done with
35 these latter indicators would produce sufficiently similar results for those areas in which PM_{10-2.5}

1 is not a concern. These findings, described below and in Appendix 4C, are considered useful in
2 informing judgments regarding the implications of indicator selection.

3 A careful examination of stacked bar charts for each hour included in the top 10% of the
4 maximum daily 1-hour and top 2% of all hours forms for both the PM_{2.5} mass and PM₁₀ light
5 extinction indicators shows that the two indicators often select the same hour or another hour
6 during the same day as top contributors (compare plot *a* to plot *d* and plot *e* to plot *h* in Figures
7 4C-1 to 4C-14 and see a tally of the numbers of common and unique days in Table 4C-1). When
8 the two indicators select unique hours among the top contributors, the PM_{2.5} mass selected hours
9 necessarily have lower PM₁₀ light extinction values than those selected by PM₁₀ light extinction
10 as the indicator, generating a bias. However, the average results for each of the 14 urban areas
11 show a remarkably similar identification of the relative amounts of the PM₁₀ components
12 responsible for the highest values regardless of which indicator is used to select the hours
13 (compare plot *a* to *d* and plot *e* to plot *h* in Figure 4C-15).

14 For any selected hour, the relative contributions of components to PM mass is not the
15 same as the relative contributions of the components to PM light extinction because the differing
16 light extinction efficiencies of the components. These differences can be seen by comparing
17 paired plots in Figure 4C-1 to 4C-15 (i.e. compare *a* and *b*, *c* and *d*, *e* and *f*, etc.). The result is
18 that components that contribute little to PM light extinction are more prominent contributors to
19 PM mass concentration (e.g., fine soil). Similarly the relative importance of the hygroscopic PM
20 components (i.e., PM sulfates and nitrates) compared to that of the non-hygroscopic components
21 (i.e. primarily organic and elemental carbon) to PM mass is generally different than it is to PM
22 light extinction and that difference changes as a function of relative humidity.

23 In summary, this comparison demonstrates that the selection of indicator has little effect
24 on which hours/days contribute to the top 10% of the maximum daily 1-hour and top 2% of all
25 hours for PM_{2.5} mass and PM₁₀ light extinction. However, in terms of properly identifying the
26 PM_{2.5} species that are the largest contributors to the visibility impairment in those selected
27 hours/days for the purpose of targeting controls, a calculated PM_{2.5} light extinction indicator
28 would correctly weight the different components in terms of their differing contributions to
29 visibility impairment, while the PM_{2.5} mass indicator would not. Likewise, a direct measurement
30 of PM_{2.5} light extinction could not, by itself, provide speciated component information useful for
31 targeting controls. Therefore, we conclude that the speciated PM_{2.5} mass-calculated light
32 extinction indicator has advantages over the PM_{2.5} mass indicator in the context of this PM
33 NAAQS review.

1 4.3.2 Averaging and Applicable Times

2 Consideration of appropriate averaging times, diurnal periods, and ambient conditions
3 over which the indicator would be measured were informed by consideration of the nature of PM
4 visibility effects.

5 With respect to consideration of appropriate averaging times, as discussed above (section
6 4.2.1), we took into account what we know from the studies concerning how quickly people
7 experience and judge visibility conditions, the possibility that some fraction of the public
8 experience infrequent or short periods of exposure to ambient visibility conditions, and the
9 typical rate of change of the path averaged PM light extinction over urban areas. While
10 perception of change in visibility can occur in less than a minute, meaningful changes to path-
11 averaged light extinction occur more slowly and can be well represented by hourly averaging.
12 Multi-hour averaging times, on the other hand, would have the effect of reducing and masking
13 the magnitude of hourly peak visibility values which can change significantly from one hour to
14 the next (see UFVA Figure 3-12). Reduction of peak values through multi-hour averaging
15 reduces the ability of the indicator to accurately characterize the visibility effects experienced by
16 the segment of the population that experiences infrequent short-term exposures during peak
17 periods. Therefore, we conclude that a 1-hr averaging time is appropriately suited to characterize
18 meaningful short-term variations in visibility conditions.

19 With respect to selection of an appropriate diurnal period, staff recognized that nighttime
20 visibility impacts, described in the ISA (section 9.2.2) are significantly different from daytime
21 impacts and not sufficiently well understood to be included at this time. As a result, we conclude
22 that a secondary standard to protect visibility should only apply to daylight hours at this time. In
23 the UFVA, daylight hours were defined to be those morning hours having no minutes prior to
24 local sunrise and afternoon hours having no minutes after local sunset. This definition ensures
25 the exclusion of periods of time where the sun is not the primary outdoor source of light to
26 illuminate scenic features.

27 In considering the well-known interaction of PM with ambient relative humidity
28 conditions, staff acknowledges that PM is not necessarily the primary source of visibility
29 impairment during periods with fog or precipitation. In order to reduce the probability that hours
30 identified as having a high degree of visibility impairment for purposes of determining
31 compliance with a standard were caused by fog or precipitation, staff determined that a relative
32 humidity screen that excludes daylight hours with average relative humidity above 90% is
33 appropriate (UFVA section 3.3.5 and Appendix G). For example, for the 15 urban areas¹⁰

¹⁰ The 90% relative humidity cap assessment was conducted as part of the UFVA on all 15 of the urban areas, including St. Louis that is not displayed in the Policy Assessment.

1 included in the final UFVA, a 90% relative humidity cutoff criterion proved effective in that on
2 average less than 6% of the daylight hours were removed from consideration, yet those same
3 hours had on average 10 times the likelihood of rain, 6 times the likelihood of snow/sleet, and 34
4 times the likelihood of fog compared with hours with 90% or lower relative humidity. However,
5 not all periods with relative humidity above 90% have fog or precipitation. We recognize that
6 removing those hours from consideration involves a tradeoff between the benefits of avoiding
7 many of the hours with meteorological causes of visibility impacts and not counting some hours
8 without fog or precipitation in which high humidity levels (> 90%) lead to the growth of
9 hygroscopic PM to large solution droplets resulting in enhanced PM visibility impacts.

10 **4.3.3 Alternative Levels/Forms**

11 Candidate Protection Levels

12 The results from the visibility preferences studies conducted in four urban areas define a
13 range of low, middle and high CPLs of 20 dv, 25 dv and 30 dv which are equivalent to PM₁₀
14 light extinction of values of 64 Mm⁻¹, 112 Mm⁻¹, and 191 Mm⁻¹(see section 4.2 above). With
15 only the four preference study results, the individual low and high CPL are in fact reflective of
16 the results from the Denver and Washington, DC studies in particular, and the middle CPL is
17 very near to the 50th percentile criteria result from Phoenix.

18 Determining PM_{2.5} mass concentration values that correspond to the low, middle and
19 high CPL is complicated by the lack of a one-to-one relationship between PM_{2.5} light extinction
20 and PM_{2.5} mass. By considering the extinction efficiency values (i.e., the ratio of PM_{2.5} light
21 extinction to PM_{2.5} mass) used in the original IMPROVE algorithm that range from 3 m²/g to 10
22 m²/g, a range of PM_{2.5} concentration levels that roughly corresponds to a low estimate of the low
23 CPL and high estimate of the high CPL values can be determined. The full range is from ~6
24 µg/m³ to ~60 µg/m³. Keeping in mind that there is no exact one-to-one correspondence between
25 PM_{2.5} mass concentration and PM_{2.5} light extinction, no individual value in this range of
26 conditions corresponds to the CPL values expressed in terms of PM_{2.5} light extinction, so a
27 number of alternate values within the range were selected for assessing their visibility protection
28 performance when combined with alternative forms (as described below). Five alternative PM_{2.5}
29 mass concentration values were selected for this purpose: 10 µg/m³, 20 µg/m³, 30 µg/m³, 40
30 µg/m³ and 60 µg/m³.

31 Alternative Forms

32 The form of the current 24-hour PM_{2.5} NAAQS is set so that the level of the standard is
33 compared to the three -year average of the annual 98th percentile of the measured indicator. The
34 purpose in averaging for three years is to provide stability from the occasional effects of inter-

1 annual meteorological variability that can result in unusual high pollution levels for a particular
2 year that is otherwise typical. The use of a percentile form, among other things, makes the
3 standard less subject to the possibility of inappropriate violations caused by statistical outlier
4 indicator values. A percentile form can also be used to take into account the number of times an
5 exposure might occur as part of the judgment on protectiveness in setting a NAAQS. For all of
6 these reasons, we believe it is appropriate to consider incorporating the use of a three-year
7 average of a specified percentile for the PM secondary standard.

8 The urban visibility preference studies that provided results leading to the range of CPLs
9 being considered in this document, offer no information that addresses the frequency of time that
10 visibility levels should be below those values. Given this lack of information and recognizing
11 that the nature of the public welfare effect is one of aesthetics and/or feelings of wellbeing, we
12 believe that it is not necessary to consider eliminating all exposures above the level of the
13 standard and that allowing some number of hours/days with reduced visibility can reasonably be
14 considered. In the UFVA, 90th, 95th and 98th percentile forms are assessed in the PM light
15 extinction NAAQS scenarios (Chapter 4). The hourly PM_{2.5} mass concentration scenarios that
16 are described and assessed below and in Appendix 4A include only the 90th and 95th percentile
17 forms.¹¹

18 Another aspect of the form that needs to be considered is whether to include all daylight
19 hours or only the maximum daily daylight 1-hour. The maximum daily daylight 1-hour form is
20 more appropriate for protecting the welfare of people who have limited, infrequent or
21 intermittent exposure to visibility during the day (e.g. during commutes), but spend most of their
22 time without an outdoor view. For such people a view of poor visibility during their morning
23 commute may represent their perception of the day's visibility conditions until the next time they
24 venture outside during daylight, hours later or perhaps the next day. Other people have exposure
25 to visibility conditions throughout the day. For those people it might be more appropriate to
26 include every daylight hour in assessing compliance with a standard, since a day with multiple
27 hours with visibility impairment is likely to be judged to have a greater impact on their wellbeing
28 than a day with just one such hour.

29 We do not have information regarding the fraction of the public that has only one or a
30 few opportunities to experience visibility during the day, nor do we have information on the role
31 the duration of the observed visibility conditions has on wellbeing effects associated with those
32 visibility conditions. However, it is logical to conclude that people with limited opportunities to
33 experience visibility conditions on a daily basis would experience the entire impact associated
34 with visibility based on their short term exposure. The impact of visibility for those who have

¹¹ Assessment of the 98th percentile with PM_{2.5} mass was thought to be unnecessarily restrictive so it was not conducted to simply and speed the assessment process.

1 access to visibility conditions often or continuously during the day may be based on varying
2 conditions throughout the day. Based on these considerations, the segment of the population
3 with infrequent access to visibility could be characterized as a susceptible population relative to
4 peak visibility impairment, while those with longer exposures are a susceptible population for
5 somewhat longer-term visibility impairment.

6 In light of these considerations, the UFVA assessment of the various PM₁₀ light
7 extinction scenarios included both the maximum daily and all daylight hours forms. We noticed
8 a close correspondence between the level of protection afforded for all 15 urban areas in the
9 assessment by the maximum daily daylight 1-hour using the 90th percentile form and the all
10 daylight 1-hour combined with the 98th percentile form (UFVA section 4.1.4). On this basis,
11 staff notes that the reductions in visibility impairment required to meet either form of the
12 standard would provide protection to both fractions of the public (i.e., those with limited
13 opportunities and those with greater opportunities to view PM-related visibility conditions).

14 Another way to assess the two forms (i.e., maximum daily and all daylight hours) is to
15 compare the PM components for hours that would be targeted for reduction by the use of each.
16 This was done for both indicators (i.e. PM_{2.5} mass concentration and PM₁₀ light extinction) by
17 examining and comparing the component contributions for current conditions (i.e., 2005 – 2007)
18 for the hours with the highest levels for each indicator. Stacked bar plots showing the PM₁₀
19 component composition of the top 10% of hours for the maximum daily 1-hour and of the top
20 2% of all daylight hours for both indicators and each of the 14 urban areas are described in
21 Appendix 4C.

22 For Figures 4C-1 to 4C-14 composition displayed in plots labeled *a* and *e* for the light
23 extinction indicator and *c* and *g* for the PM_{2.5} mass indicator can be compared to assess the effect
24 of using the maximum daily compared with the all daylight hours forms. Some of the same
25 hours are selected by both forms. By definition the top 10% maximum daily hour form never
26 has more than 1 hour selected per day. There are very nearly twice as many hours in the top 2%
27 of all hours compared to the top 10% of maximum daily hours. However, it is not unusual to
28 have multiple hours selected in the same day by the top 2% of all hours form. These hours
29 selected in the same day tend to have nearly identical composition. The top 2% of all hours may
30 select hours on a greater number of days compared with the top 10% of maximum daily hours, or
31 if there are enough instances of multiple hours per day, it could select hours on a smaller number
32 of days compared with the top 10% of maximum daily hours. A list of the numbers of common
33 and unique days selected comparing the two forms for both indicators and the number of days
34 selected by each combination of indicator and form is shown in Table 4C-2.

35 For most of the urban areas the number of common days is much larger than the number
36 of unique days. Generally, the PM component mix is less similar for sites with a lower number

1 of common and a greater number of unique days. For most of the 14 urban areas, the differences
2 in components contributions for the two forms are minor. The greatest difference between the
3 two forms is for Salt Lake City. Average effects of the choice of form for each urban areas is
4 displayed in Figure 4C-15 (compare plot *a* to plot *e* and plot *c* to plot *g*). This shows that the
5 differences in PM component contributions differ more depending on the choice of indicator
6 than the choice of form. Thus, these two specific percentile levels correspond to the nearly
7 equivalent levels of protection afforded by the maximum daily 90th percentile and the all daylight
8 hours 98th percentile forms.

9 **4.3.4 Performance of Alternative Standards**

10 We conducted assessments of alternative standards based both on hourly PM₁₀ light
11 extinction as the indicator (see Chapter 4 of the UFVA) and on the hourly PM_{2.5} mass
12 concentration indicator (see Appendix 4A in this document). In the context of these assessments
13 of the 14 urban areas, PM₁₀ light extinction is dominated by PM_{2.5} light extinction except for
14 Phoenix, and therefore we consider these assessments reasonably comparable to either the
15 directly measured or speciated PM_{2.5} mass-calculated light extinction indicators.¹² There are
16 nine PM₁₀ light extinction alternative standards discussed here (i.e., daily maximum daylight 1-
17 hour PM₁₀ light extinction at the low, middle and high CPL for each of the 90th, 95th and 98th
18 percentile forms) and there are ten PM_{2.5} mass concentration alternative standards (i.e., five
19 concentration levels for each of the 90th and 95th percentile forms). In both cases the assessment
20 involves rolling back non-policy relevant background (PRB) PM_{2.5} light extinction or PM_{2.5} mass
21 concentration values until these specific alternative standards are just met.

22 In considering the performance of alternative standards, we focused on the consistency
23 across the 14 urban areas of the resulting visibility conditions, as measured in terms of light
24 extinction, when the alternative scenarios are just met. Because of the one-to-one
25 correspondence between light extinction and PM₁₀ light extinction,¹³ the PM₁₀ light extinction
26 indicator based standards should produce visibility conditions exactly as prescribed. The ability
27 to structure a PM₁₀ light extinction based alternative standard to the level of visibility protection
28 desired is an advantage because it could afford the most uniform degree of visibility protection
29 nationwide.

30 Figures similar to Figure 4-5 for the other PM₁₀ light extinction scenarios are shown in
31 Appendix F of the UFVA. Table 4-4 shows the design values for the 9 scenarios based on
32 maximum daily 1-hour PM₁₀ light extinction. When an area just meets a scenario, its design

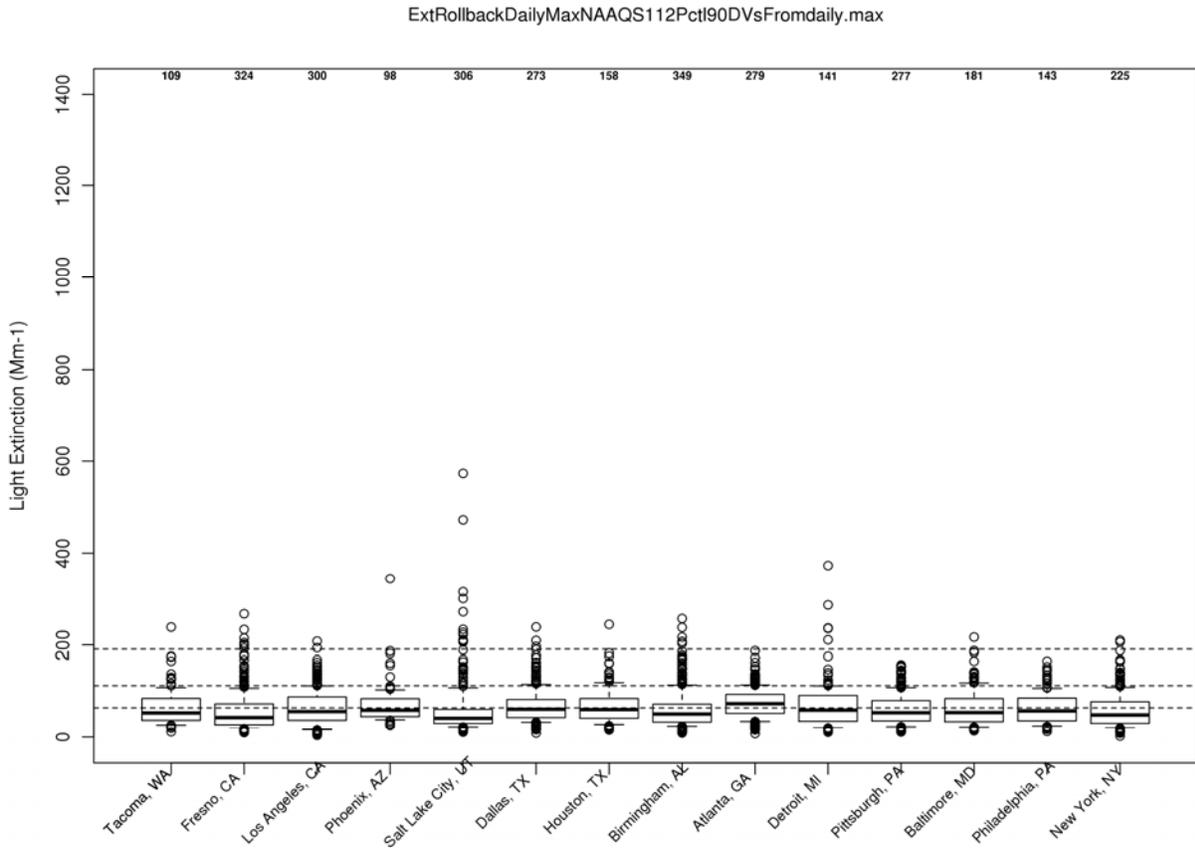
¹² These assessments were conducted prior to the decision to focus on PM_{2.5} light extinction indicators.

¹³ In this assessment light extinction = PM₁₀ light extinction + 10Mm⁻¹, where the last term is Rayleigh or clean air light scattering that in fact can range from about 8Mm⁻¹ to 12Mm⁻¹ depending on average sight path elevation above sea level.

1 value in principle should exactly equal the NAAQS level, so preparation of this table serves as a
2 check against calculation errors. Note that the design values in Table 4-4, resulting from the
3 rollback steps described in section 4.1.4 of the UFVA, in some cases do not exactly equal the
4 assumed level of the NAAQS, although all are quite close. In some cases (e.g. Phoenix for 191
5 $\text{Mm}^{-1}/90^{\text{th}}$ and 95th percentile), current conditions already meet the scenario specifications so no
6 rollback was necessary and current design values are shown in Table 4-4 and reflected in box
7 and whisker plot figures. The minor differences between prescribed and assessed design values
8 seen for some applications of the rollback assessment are due to hours switching in the PM_{10}
9 light extinction frequency distribution that is purely an artifact of the rollback methodology as
10 described in the UFVA (section 4.3). These discrepancies were judged too small to justify
11 iterative rollback that could have been used to eliminate them.

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Figure 4-6 Daily maximum daylight 1-hour PM₁₀ light extinction after rollback to just meet a scenario with daily maximum of 112 Mm⁻¹ for the 90th percentile excluding hours with relative humidity greater than 90 percent.



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Table 4-4. PM₁₀ light extinction design values for “just meeting” secondary NAAQS scenarios based on measured PM₁₀ light extinction (excluding hours with relative humidity greater than 90 percent)

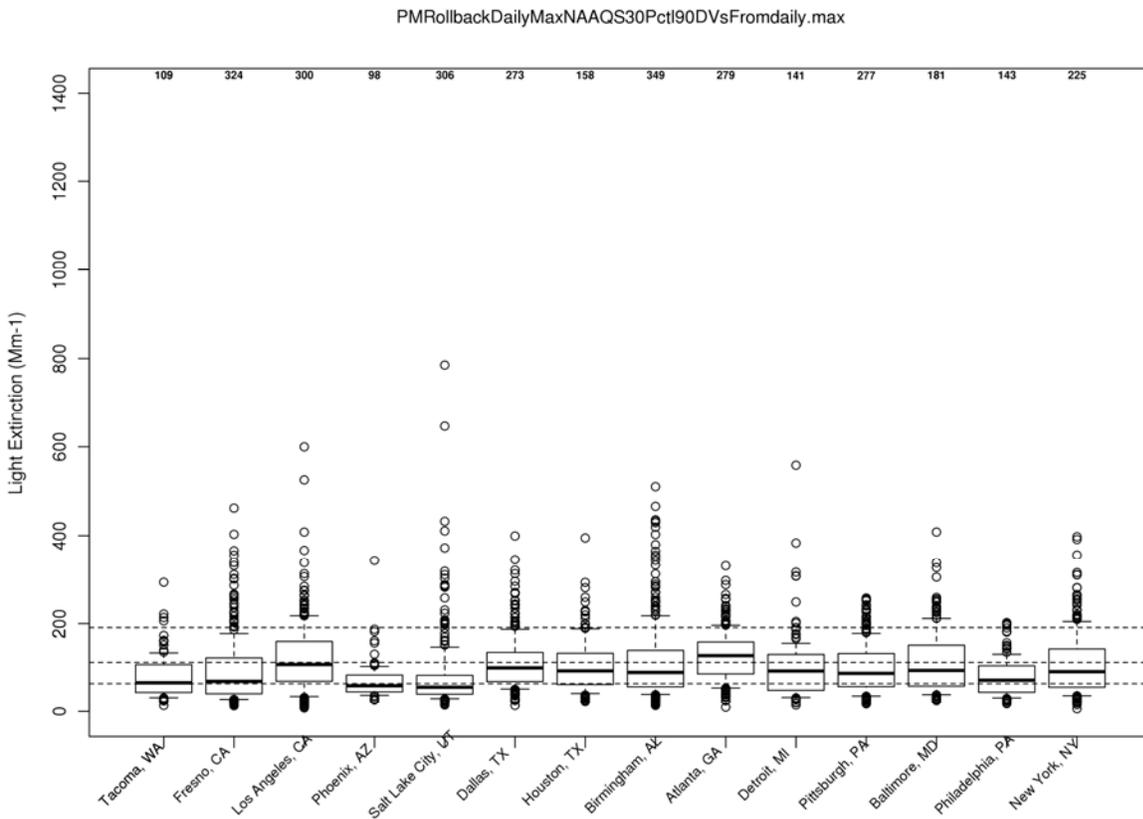
	Secondary NAAQS Scenarios Based on Daily Maximum								
	(a)	(b)	(c)	(d)	(e)	(f)	(g)	(h)	(i)
Level (Mm ⁻¹)	191	191	191	112	112	112	64	64	64
Percentile Form	90 th	95 th	98 th	90 th	95 th	98 th	90 th	95 th	98 th
	PM light extinction Design Value (based on same percentile form as the NAAQS scenario)								
Tacoma, WA	140	157	191	112	112	108	66	70	60
Fresno, CA	191	191	191	112	112	112	64	64	64
Los Angeles, CA	191	191	191	112	112	112	65	64	64
Phoenix, AZ	105	144	185	105	112	112	64	64	64
Salt Lake City, UT	164	191	191	112	112	112	64	64	64
Dallas, TX	183	191	191	113	113	112	64	66	66
Houston, TX	191	191	191	115	112	112	67	61	67
Birmingham, AL	191	192	191	113	114	112	64	66	64
Atlanta, GA	191	191	191	112	111	112	64	63	65
Detroit, MI	191	191	191	112	112	112	64	64	65
Pittsburgh, PA	191	191	191	112	112	112	64	64	64
Baltimore, MD	191	191	191	111	112	112	63	64	65
Philadelphia, PA	191	191	191	112	112	112	65	64	64
New York, NY	192	191	191	113	112	112	65	64	64

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A complete description of the rollback assessment for the ten maximum daily 1-hour PM_{2.5} mass concentration scenarios is available in Appendix 4A of this document. The process used is very similar to that used in the UFVA for rollback of the PM₁₀ light extinction based scenarios.

1 Since PM_{2.5} mass does not have a one-to-one correspondence to light extinction, the PM₁₀
 2 light extinction conditions resulting from just meeting standards based on PM_{2.5} mass are not as
 3 uniform as those shown above for PM₁₀ light extinction scenarios. This is demonstrated in the
 4 box and whisker plot of the maximum daily 1-hour PM₁₀ light extinction based on 1-hr PM_{2.5}
 5 mass for the 30 µg/m³, 90th percentile scenario shown below (Figure 4-7), which shows greater
 6 variation across the urban areas than is seen in the PM₁₀ light extinction based scenarios. Similar
 7 plots for all ten PM_{2.5} mass based scenarios are shown in Appendix 4A (Figure 4A-2).

8 **Figure 4-7. Maximum daily daylight 1-hour PM₁₀ light extinction under "just meet"**
 9 **conditions for a NAAQS scenario based on 1-hour PM_{2.5} mass of 30 µg/m³, 90th percentile,**
 10 **excluding relative humidity >90%.**



11 The 90th percentile PM₁₀ light extinction design values corresponding to the 90th
 12 percentile PM_{2.5} mass concentration based scenarios for the five mass concentration levels are
 13 shown in Table 4-5. Values in the table that exceed each of the CPL values are highlighted using
 14 different colors. The values in the table are the same for some urban areas for the less restrictive
 15 PM_{2.5} mass concentration standard (e.g. Tacoma at 60 µg/m³ and 40 µg/m³, or Phoenix at 60
 16 µg/m³, 40 µg/m³, 30 µg/m³ and 20 µg/m³), because those areas required no rollback to meet the
 17 less restrictive PM_{2.5} mass concentration standards. In order for most or all urban areas to
 18

1 achieve even the highest CPL at the 90th percentile, a PM_{2.5} mass based standard set below 40
2 µg/m³ would be necessary. Based on the CPL's considered, the higher PM_{2.5} mass levels of 60
3 µg/m³ and 40 µg/m³ clearly are not sufficiently protective, since they permit 10 and 9 of the 14
4 areas, respectively, to have design values larger than the high CPL value (i.e., 191 Mm⁻¹). The 30
5 µg/m³ standard level has PM₁₀ light extinction values that are marginally above the high CPL at
6 3 of the 14 urban areas and has a mean value below the high CPL. At the 20 µg/m³ PM_{2.5}
7 standard level, all areas meet the highest CPL benchmark, and the mean PM₁₀ light extinction
8 value across the 14 urban areas is nearer to the middle CPL (i.e., 112 Mm⁻¹) than to the high
9 CPL. The 10 µg/m³ standard level produces a mean PM₁₀ light extinction value of 79.1 Mm⁻¹,
10 which is greater than the low CPL (i.e., 64 Mm⁻¹) but nearer to it than the middle CPL.

1 **Table 4-5. 90th percentile maximum daily 1-hour PM light extinction design values (Mm⁻¹)**
 2 **after rollback to meet alternative standards of 60 µg/m³, 40 µg/m³, 30 µg/m³, 20 µg/m³ and**
 3 **10 µg/m³ maximum daylight 1-hour PM mass concentration for the 90th percentile.***
 4

	60 µg/m ³	40 µg/m ³	30 µg/m ³	20 µg/m ³	10 µg/m ³
Tacoma, WA	140 Mm ⁻¹	140 Mm ⁻¹	140 Mm ⁻¹	128 Mm ⁻¹	82 Mm ⁻¹
Fresno, CA	338 Mm ⁻¹	248 Mm ⁻¹	190 Mm ⁻¹	132 Mm ⁻¹	74 Mm ⁻¹
Los Angeles, CA	403 Mm ⁻¹	284 Mm ⁻¹	220 Mm ⁻¹	156 Mm ⁻¹	105 Mm ⁻¹
Phoenix, AZ	105 Mm ⁻¹	105 Mm ⁻¹	105 Mm ⁻¹	105 Mm ⁻¹	86 Mm ⁻¹
Salt Lake City, UT	164 Mm ⁻¹	164 Mm ⁻¹	153 Mm ⁻¹	107 Mm ⁻¹	59 Mm ⁻¹
Dallas, TX	183 Mm ⁻¹	183 Mm ⁻¹	183 Mm ⁻¹	146 Mm ⁻¹	80 Mm ⁻¹
Houston, TX	194 Mm ⁻¹	194 Mm ⁻¹	179 Mm ⁻¹	125 Mm ⁻¹	73 Mm ⁻¹
Birmingham, AL	357 Mm ⁻¹	266 Mm ⁻¹	208 Mm ⁻¹	152 Mm ⁻¹	102 Mm ⁻¹
Atlanta, GA	249 Mm ⁻¹	249 Mm ⁻¹	191 Mm ⁻¹	134 Mm ⁻¹	76 Mm ⁻¹
Detroit, MI	291 Mm ⁻¹	202 Mm ⁻¹	157 Mm ⁻¹	120 Mm ⁻¹	88 Mm ⁻¹
Pittsburgh, PA	278 Mm ⁻¹	243 Mm ⁻¹	185 Mm ⁻¹	127 Mm ⁻¹	69 Mm ⁻¹
Baltimore, MD	246 Mm ⁻¹	246 Mm ⁻¹	201 Mm ⁻¹	138 Mm ⁻¹	76 Mm ⁻¹
Philadelphia, PA	258 Mm ⁻¹	175 Mm ⁻¹	134 Mm ⁻¹	98 Mm ⁻¹	63 Mm ⁻¹
New York, NY	306 Mm ⁻¹	281 Mm ⁻¹	212 Mm ⁻¹	141 Mm ⁻¹	74 Mm ⁻¹
Mean Values	250.9 Mm ⁻¹	212.9 Mm ⁻¹	175.6 Mm ⁻¹	129.2 Mm ⁻¹	79.1 Mm ⁻¹

5
 6 **Colored highlighting shows which of the CPL levels the values are near, using the*
 7 *following definitions: PM light extinction equal or above the high CPL ≥191 Mm⁻¹; above the*
 8 *middle CPL, 112 Mm⁻¹ – 190 Mm⁻¹; above the low CPL, 64 Mm⁻¹ – 111 Mm⁻¹; below the low*
 9 *CPL, < 64 Mm⁻¹. Values without color highlighting indicate no rollback, so no information*
 10 *about the effectiveness of the hourly PM mass based alternative standards*
 11

12 The range of 90th percentile PM_{2.5} light extinction design levels resulting from rollback to
 13 just meet a 90th percentile 20 µg/m³ PM_{2.5} mass concentration is from 98 Mm⁻¹ for Philadelphia
 14 to 156 Mm⁻¹ for Los Angeles with most values near the middle CPL value of 112 Mm⁻¹. A
 15 similar range for the 30 µg/m³ PM mass concentration is from 134 Mm⁻¹ to 220 Mm⁻¹ for the
 16 same two cities. This demonstrates the amount of variability in visibility conditions that would
 17 likely result from using a standard with PM_{2.5} mass concentration instead of PM_{2.5} light
 18 extinction as the indicator. While this degree of variation is not particularly large, it does mean
 19 that some areas would be required to further reduce ambient levels to meet a secondary standard
 20 though they have visibility conditions that are as good or better than other areas which meet the
 21 PM_{2.5} mass based standard. Some measure of the extent to which this occurs may be gained by

1 comparing the percent rollback values required to meet the various PM mass and PM light
2 extinction values.

3 Based on inspection of the design values for individual urban areas and means shown in
4 Table 4-5, among the PM_{2.5} mass concentration levels that were assessed, 30 µg/m³ provides
5 protection somewhat comparable to the high CPL (191 Mm⁻¹), 20 µg/m³ provides protection
6 somewhat comparable to the middle CPL (112 Mm⁻¹) and 10 µg/m³ provides protection
7 somewhat comparable to the low CPL (64 Mm⁻¹). Additional assessments could be conducted to
8 refine the PM_{2.5} mass concentration values to better correspond to the three CPL PM light
9 extinction values for these 14 urban areas, though this would not much change the variations in
10 performance across urban areas.

11 Care must be taken to avoid misinterpretation of this suggestion of approximate
12 comparability of protection afforded the various urban areas by PM mass at the three
13 concentration levels in terms of the three CPLs. The values in Table 4-5 are the 90th percentile
14 values of the PM_{2.5} light extinction distribution that results from transforming the hourly PM_{2.5}
15 mass and composition values of the rolled back PM distributions for each urban area. Hours
16 with values above the 90th percentile of PM mass do not necessarily have PM light extinction
17 above the 90th percentile and visa versa. As discussed earlier (section 4.3.1) for any individual
18 hourly PM mass concentration there is a substantial range of corresponding PM light extinction
19 values possible.

20 **4.4 STAFF CONCLUSIONS ON ALTERNATIVE SECONDARY STANDARDS FOR** 21 **VISIBILITY-RELATED EFFECTS**

22 In reaching conclusions on potential alternative standards to provide requisite protection
23 of PM-related visibility impairment, staff has considered the basic elements of the NAAQS:
24 indicator, averaging time, form and level. In considering the scientific and technical
25 information, we considered the information available in the last review integrated with
26 information that is newly available as assessed and presented in the ISA and UFVA and as
27 summarized in sections 4.2, 4.3, and Appendices 4A, 4B and 4C.

28 As outlined in section 4.1.3, we emphasize a policy approach that broadens the general
29 approaches used in the last two PM NAAQS reviews by utilizing, to the extent available,
30 enhanced tools, methods, and data to more comprehensively characterize visibility impacts. As
31 such, we have taken into account both evidence-based and impact assessment-based
32 considerations to inform our conclusions related to the adequacy of the current PM_{2.5} secondary
33 standards and alternative standards that are appropriate for consideration in this review. In so
34 doing, we are seeking to provide as broad an array of options as is supportable by the available
35 information, recognizing that the selection of a specific approach to reaching final decisions on

1 the secondary PM standards for protection from PM-related visibility impairment will reflect the
2 judgments of the Administrator. We recognize that selecting from among alternative standards
3 will necessarily reflect consideration of the qualitative and quantitative uncertainties inherent in
4 the relevant evidence and in the assumptions that underlie the quantitative visibility impact
5 assessment.

6 (1) Consideration should be given to revising the current suite of PM_{2.5} secondary standards to
7 provide increased public welfare protection from PM_{2.5}-related visibility impairment,
8 primarily in urban areas. This conclusion is based in general on the evaluation in the ISA of
9 the currently available information, including a more extensive characterization of the
10 sources contributing to visibility impairment in both rural and urban locations; a refined
11 understanding of the contributions of various PM components in such areas; exposure-related
12 evidence supporting a causal relationship between ambient PM and impaired VAQ, evidence
13 that a significant number of days with levels of VAQ that could reasonably be considered
14 unacceptable would continue to occur in areas where the current standards were met; and
15 judgments as to the public welfare significance of these occurrences upon just meeting the
16 current suite of PM_{2.5} standards.

17 (2) Consideration should be given to establishing a new speciated PM_{2.5} mass-calculated light
18 extinction indicator. This conclusion takes into consideration the available evidence that
19 demonstrates a one-to-one correspondence between directly measured or calculated PM light
20 extinction and PM-related visibility impairment as well as the significant degree of
21 variability in visibility protection across the U.S. allowed by a PM_{2.5} mass indicator. While a
22 secondary standard that uses a PM_{2.5} mass indicator could be set to provide additional
23 protection from PM_{2.5}-related visibility impairment, we conclude that the advantages of using
24 a speciated PM_{2.5} mass-calculated light extinction indicator (as described above in section
25 4.3.1) make it the preferred choice.

26 (3) Consideration should be given to a one hour averaging time to protect against daytime PM_{2.5}
27 -related visibility impairment that takes into account the short-term (instantaneous) nature of
28 the perception of visibility impairment, short-term variability in PM-related VAQ (partial
29 hour to hourly), and the short-term nature of relevant exposure periods for the viewing public
30 (partial hour to multiple hours). Due to limitations in the science related to assessing public
31 welfare impacts associated with nighttime visibility impairment, consideration should be
32 given to including daylight hours only. Further, consideration should be given to applying a
33 90% relative humidity screen to remove hours in which fog or precipitation is much more
34 likely to contribute to the observed visibility impairment.

35 (4) In conjunction with a 1-hour averaging time, consideration should be given to either a
36 maximum daily daylight 1-hour form or a form that includes all daylight hours.
37 Consideration should be given to a form defined in terms of the 90th, 95th or 98th percentile of
38 the distribution of calculated PM_{2.5} light extinction, averaged over three years.

39 (5) Consideration should be given to selecting a target level of acceptable PM-related VAQ in
40 terms of PM_{2.5} light extinction. Further, consideration should be given to a range of
41 alternative candidate protection levels from down to 64 Mm⁻¹ and up to 191 Mm⁻¹, with a

1 midpoint level of 112 Mm^{-1} to provide appropriate protection against $\text{PM}_{2.5}$ -related visibility
2 impairment.

3 (6) Consideration should be given to the following alternative secondary $\text{PM}_{2.5}$ standards to
4 provide protection against $\text{PM}_{2.5}$ -related visibility impairment during daylight hours:

5 (a) Consideration should be given to establishing a new 1-hour daily maximum
6 speciated $\text{PM}_{2.5}$ mass-calculated light extinction secondary standard set at a level
7 within the range of 64 to 191 Mm^{-1} in combination with a 90th or 95th percentile
8 form, averaged over 3 years.

9 (b) Alternatively, consideration could be given to a new all daylight hours speciated
10 $\text{PM}_{2.5}$ mass-calculated light extinction secondary standard set at a level within the
11 range of 64 to 191 Mm^{-1} in combination with a 98th percentile form, averaged
12 over 3 years.

13 **4.5 KEY UNCERTAINTIES AND AREAS FOR FUTURE RESEARCH AND DATA** 14 **COLLECTION**

15 This section will be organized into two overarching topic areas: refining current
16 understanding of visibility preferences and characterization of ambient urban visibility
17 conditions. The first deals principally with how the public reacts to and values visibility
18 conditions, while the second is more concerned with determining ambient visibility conditions
19 and the relationships between PM component concentrations and light extinction.

20 Visibility Preferences

- 21 • Levels: The results of the reanalysis of the four urban preference studies (UFVA Chapter 2)
22 demonstrated well defined though significant statistical differences in visibility impairment
23 levels that divide acceptable from unacceptable conditions across the study areas. A number
24 of hypothesis concerning why the results differed for each area are discussed in chapter 2 of
25 the UFVA, but the current state of knowledge does not support a definitive explanation for
26 the range of results. A better understanding of the reasons for the differences in preference
27 response among the studies of the four urban areas could influence the design of future
28 visibility preference survey studies and the interpretation of their results ultimately leading to
29 a better defined range of CPLs for the next PM NAAQS review.
- 30 • Averaging Times/Forms: Additional information would also be helpful in deciding among
31 the various forms and averaging times to develop an effective visibility based secondary PM
32 NAAQS and to assess the overall benefits of visibility improvements. Our current
33 understanding of urban visibility effects does not provide insights concerning:
 - 34 ○ relative importance of intensity versus frequency of visibility impairment,
 - 35 ○ strength of preference for different distributions of visibility conditions, and
 - 36 ○ public exposure rates and mechanisms.

37 Future research to address these deficiencies should include designing and conducting
38 additional preference, valuation and exposure studies to:

- 1 ○ expand the number and geographic coverage of urban area preference results,
- 2 ○ evaluate the sensitivity of results to the differences in survey study methodology,
- 3 ○ apply consistent methodology at multiple urban areas to better understand reasons for
- 4 preference difference among results in different urban areas,
- 5 ○ develop information on the strength of preference and relative importance of intensity
- 6 versus frequency of visibility impairment
- 7 ○ identify the types of scenic elements that are most influential for informing public
- 8 visibility impact awareness, and
- 9 ○ provide insights concerning visibility impact exposure duration, intensity and timing
- 10 and their relationship to the degree and longevity of public welfare effects.

11 Additionally, prior to the next PM NAAQS review and as part of the planning for
12 additional preference and valuation survey studies, a literature review of recent social science
13 literature could usefully be conducted to assess the state of knowledge of view exposure
14 mechanisms, and the psychological and behavioral effects associated with viewed stimuli.

15 Urban Visibility Conditions

16 The paucity of light extinction monitoring data for urban areas lead to the use of the
17 IMPROVE algorithm to calculate hourly light extinction from continuous PM mass and 24-hour
18 PM_{2.5} component and relative humidity data (UFVA Chapter 3). The steps used to apportion 24-
19 hour PM_{2.5} component to calculate hourly averaged values used monthly mean diurnal PM
20 component variations from regional air quality modeling. The mass balance methods used to
21 estimate organic concentration and the loss of nitrate are reasonable but not likely to be precise.
22 The IMPROVE algorithm was originally developed for remote area application to estimate 24-
23 hour light extinction and it was not verified for use in generating urban hourly estimates. The
24 resulting hourly PM light extinction data set is thought to be reasonably representative of the
25 hourly PM light extinction levels that is adequate for assessment purposes, but these data are less
26 accurate than those from direct measurements.

27 A pilot PM light extinction monitoring program could usefully be designed and deployed
28 at some number of locations selected to cover a range of PM air quality conditions with emphasis
29 given to locations with continuous PM mass and speciation monitoring as well as 24-hour mass
30 and speciation sampling. Information from such a pilot monitoring program could be used to:

- 31 ○ evaluate the performance of PM light extinction monitoring methods that could
- 32 ultimately be use as an FRM,
- 33 ○ evaluate and refine approaches for apportioning 24-hour PM species to hourly values
- 34 (needed for sites without continuous PM speciation monitoring),
- 35 ○ evaluate and refine light extinction calculation algorithms for use in urban settings,
- 36 and
- 37 ○ conduct the visibility effects assessment for the next PM secondary NAAQS.

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1 **5 REVIEW OF THE SECONDARY STANDARDS FOR OTHER**
2 **WELFARE EFFECTS**

3 This chapter presents staff conclusions with regard to the current suite of secondary PM
4 standards to protect against PM-related welfare effects other than visibility impairment.
5 Specifically, staff has assessed the relevant information related to effects of atmospheric PM on
6 the environment, including effects on climate, ecological effects, and effects on materials. Our
7 assessment is framed by a series of key policy-relevant questions, which expand upon those
8 presented in the Integrated Review Plan (IRP) (US EPA, 2008a, section 3.2). The answers to
9 these questions will inform decisions on whether to retain or revise the current suite of secondary
10 PM standards.

11 In presenting staff conclusions with regard to the current secondary standards relative to
12 PM-related effects on climate, ecological effects, and materials, we note that the final decision is
13 largely a public welfare policy judgment. A final decision must draw upon scientific information
14 and analyses about non-visibility PM-related effects and related impacts on public welfare, as
15 well as judgments about how to deal with the range of uncertainties that are inherent in the
16 scientific evidence and analyses. Our approach to informing these judgments is discussed more
17 fully below. This approach is consistent with the requirements of the NAAQS provisions of the
18 Act and with how EPA and the courts have historically interpreted the Act. These provisions
19 require the Administrator to establish secondary standards that, in the Administrator’s judgment,
20 are requisite to protect public welfare from any known or anticipated adverse effects associated
21 with the presence of the pollutant in the ambient air. In so doing, the Administrator seeks to
22 establish standards that are neither more nor less stringent than necessary for this purpose. The
23 Act does not require that secondary standards be set at a zero-risk level, but rather at a level that
24 avoids unacceptable public welfare impacts.

25 Information on the approaches used to set the secondary PM standards in past reviews as
26 well as our current approach for this review are presented in section 5.1. A discussion of the
27 scope of the review as related to non-visibility welfare effects of PM is included in section 5.1.2.
28 This chapter considers each of the non-visibility welfare effects separately. The discussion of
29 PM-associated effects on climate (section 5.2), ecological effects (section 5.3), and materials
30 (section 5.4) are each followed by a consideration of key uncertainties and areas for future
31 research and data collection.

32 **5.1 APPROACH**

33 Background information on the approaches used to establish the PM secondary standards
34 in 1997 and revisions to those standards in 2006 are summarized below. This section also

1 includes a discussion of the ongoing joint review of ecological effects of oxides of nitrogen and
2 sulfur (NO_x/SO_x secondary review) for clarity, since depositional effects of PM components of
3 NO_x and SO_x to ecosystems were historically considered as a component of the PM secondary
4 review. Lastly, there is a discussion of the current approach for evaluating the effects of PM on
5 climate, ecosystems, and materials using evidence-based considerations to inform our
6 understanding of the key policy-relevant issues.

7 **5.1.1 Approaches Used in Previous Reviews**

8 **5.1.1.1 Review Completed in 1997**

9 In the 1997 review, as discussed in section 2.1.1.1, EPA determined that for the primary
10 standard the fine and coarse fractions of PM₁₀ should be considered separately and added a suite
11 of new primary standards, using PM_{2.5}, as the indicator for fine particles, and retaining PM₁₀ as
12 the indicator for regulating thoracic coarse particles. The EPA established two new PM_{2.5}
13 standards: an annual standard of 15 µg/m³, based on the 3-year average of annual arithmetic
14 mean PM_{2.5} concentrations from single or multiple community-oriented monitors; and a 24-hour
15 standard of 65 µg/m³, based on the 3-year average of the 98th percentile of 24-hour PM_{2.5}
16 concentrations at each population-oriented monitor within an area (62 FR 38652, July 18, 1997).

17 With respect to the secondary PM standards, EPA concluded in 1997, that the available
18 evidence on effects of PM on non-visibility welfare endpoints was not sufficient to warrant a
19 separate secondary standard. Therefore, the secondary standards were set equal to the primary
20 PM_{2.5} and PM₁₀ standards in the final rule to provide protection against effects on visibility as
21 well as materials damage and soiling effects related to fine and coarse particles (62 FR 38683).

22 **5.1.1.2 Review Completed in 2006**

23 In 2006, the Administrator concluded that there was insufficient information to consider a
24 distinct secondary standard based on PM-related impacts to ecosystems, materials damage and
25 soiling, and climatic and radiative processes (71 FR 61144, October 17, 2006). Specifically,
26 there was a lack of evidence linking various non-visibility welfare effects to specific levels of
27 ambient PM. To provide a level of protection for welfare-related effects, the secondary
28 standards were set equal to the revised primary standards to directionally improve the level of
29 protection afforded vegetation, ecosystems and materials (71 FR 61210).

30 In the last review, the 2004 AQCD concluded that regardless of size fraction, particles
31 containing nitrates and sulfates have the greatest potential for widespread environmental
32 significance (US EPA, 2004, sections 4.2.2 and 4.2.3.1). Considerable supporting evidence was
33 available that indicated a significant role of NO_x, SO_x, and transformation products in

1 acidification and nutrient enrichment of terrestrial and aquatic ecosystems (71 FR 61209). The
2 recognition of these ecological effects, coupled with other considerations detailed below, led
3 EPA to initiate a joint review of the NO₂ and SO₂ secondary NAAQS that will consider the
4 gaseous and particulate species of NO_x and SO_x with respect to the ecosystem-related welfare
5 effects that result from the deposition of these pollutants and transformation products.

6 **5.1.2 Scope of Current NAAQS Reviews**

7 Non-visibility welfare-based effects of oxides of nitrogen and sulfur are divided between
8 two NAAQS reviews; (1) the PM NAAQS review and, (2) the joint NO_x/SO_x secondary
9 NAAQS review. The scope of each document and the components of N and S considered in
10 each review are detailed in this section and summarized in Table 5-1.

11 **5.1.2.1 Scope of the Current Secondary PM NAAQS Review**

12 In reviewing the current suite of secondary PM standards to address visibility impairment
13 (chapter 4), climate forcing effects (section 5.2), and other welfare-related effects (sections 5.3
14 and 5.4), all PM-related effects that are not being covered in the NO_x/SO_x review are
15 considered. With regard to the materials section (5.4), the discussion has been expanded to
16 include particles and gases that are associated with the presence of ambient NO_x and SO_x, as
17 well as NO_y, NH₃ and NH_x for completeness. By excluding the effects associated with
18 deposited particulate matter components of NO_x and SO_x and their transformation products
19 which are addressed fully in the NO_x/SO_x secondary review, as outlined below, the discussion
20 of ecological effects of PM has been narrowed to focus on effects associated with the deposition
21 of metals and, to a lesser extent, organics (section 5.3).

1 **Table 5-1. Scope of the current secondary PM NAAQS review and current NOx/SOx secondary review.**

	NOx/SOx Secondary Review		PM Secondary Review				
Welfare Effect	Acidifying deposition, nutrient enrichment	Direct effects of gas-phase NOx/SOx on vegetation	Visibility impairment	Climate Forcing effects	Ecological effects	Materials	
						Damage	Soiling
Documents							
ISA	NOx/SOx	NOx/SOx	PM	PM	PM	PM and NOx/SOx Annex E	PM
REA	NOx/SOx	NOx/SOx	PM (Urban focused visibility assessment)				
PA	NOx/SOx	NOx/SOx	PM	PM	PM	PM	PM
Components	Deposited particulate and gaseous forms of oxides of nitrogen and sulfur and related N and S containing compounds.	Gaseous forms of oxides of nitrogen and sulfur and related N and S containing compounds in the ambient air.	All particles 10 microns or smaller in the ambient air.	Climate-related particles (aerosols) in the ambient air.	Deposited components of PM, including metals and organics but not N and S containing compounds.	Particles and gases associated with ambient NOx and SOx including NOy, NH ₃ and NHx.	Deposited particles

2

5.1.2.2 Scope of the Current NO_x/SO_x Secondary NAAQS Review

This is the first time since the NAAQS were established in 1971 that a joint review of the secondary NAAQS for NO_x and SO_x has been conducted. This review is being conducted because the atmospheric chemistry and environmental effects of NO_x, SO_x, and their associated transformation products are linked, and because the National Research Council (NRC) has recommended that EPA consider multiple pollutants, as appropriate, in forming the scientific basis for the NAAQS. The NO_x/SO_x secondary review focuses on the welfare effects associated with exposures from deposited particulate and gaseous forms of oxides of nitrogen and sulfur and related N and S containing compounds and transformation products on ecosystem receptors. An assessment of the complex ecological effects associated with N deposition requires consideration of multiple forms of N. These include evaluation of data on inorganic reduced forms of N (e.g., ammonia [NH₃] and ammonium ion [NH₄⁺]), inorganic oxidized forms (e.g., NO_x, nitric acid [HNO₃], nitrous oxide [N₂O], nitrate [NO₃⁻]), and organic N compounds (e.g., urea, amines, proteins, nucleic acids). In addition to acidification and N-nutrient enrichment, other welfare effects related to deposition of N- and S-containing compounds are discussed, such as SO_x interactions with mercury (Hg) methylation. In addition, the NO_x/SO_x secondary review includes evidence related to direct ecological effects of gas-phase NO_x and SO_x since the direct effects of gas-phase SO_x on vegetation formed a primary basis for the initial establishment of the secondary NAAQS for SO₂.

Effects of acidifying deposition associated with particulate N and S are covered in the recent *Integrated Science Assessment for Oxides of Nitrogen and Sulfur-Ecological Criteria (Final Report)* (US EPA, 2008c). The *Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur (Final)* (NO_x/SO_x REA) (US EPA, 2009h) considers four main targeted ecosystem effects considered in the review of secondary effects of NO_x and SO_x: (1) aquatic acidification due to N and S, (2) terrestrial acidification due to N and S, (3) aquatic nutrient enrichment, including eutrophication and (4) terrestrial nutrient enrichment. In the draft *Policy Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur* (US EPA 2010c) ecologically-based indicators that link atmospheric concentrations to deposition are being considered.

5.1.3 Current Approach

The remainder of this chapter summarizes and highlights key aspects of the policy relevant information from the ISA to help inform the Administrator's judgments regarding the adequacy of the current suite of secondary PM NAAQS in relation to climate processes,

1 ecological effects, and materials damage. The ISA uses a five-level hierarchy that classifies the
2 weight of evidence for causation, not just association, into a qualitative statement about the
3 overall weight of evidence and causality (US EPA, 2009a, section 1.5.5, Table 1-3): causal
4 relationship; likely to be a causal relationship; suggestive of a causal relationship; inadequate to
5 infer a causal relationship; not likely to be a causal relationship (see US EPA, 2009a, Table 1-3).

6 Staff is evaluating evidence-based considerations primarily by assessing the evidence of
7 associations identified in the ISA. All relationships between PM and climate, ecological effects,
8 and materials damage effects identified in the ISA are considered to be either “likely causal” or
9 “causal”. The staff’s approach in this review of non-visibility welfare effects of PM is to
10 consider information regarding particulate matter effects on climate, ecological endpoints and
11 materials. This includes new literature available since the last review as well as existing,
12 relevant information as presented in the ISA (US EPA 2009a).

13 **5.2 CLIMATE**

14 **5.2.1 Scope**

15 Information and conclusions about what is currently known about the role of PM in
16 climate is summarized in Chapter 9 of the PM ISA (US EPA, 2009a). The ISA concludes; “that
17 a causal relationship exists between PM and effects on climate, including both direct effects on
18 radiative forcing and indirect effects that involve cloud feedbacks that influence precipitation
19 formation and cloud lifetimes” (US EPA, 2009a, section 9.3.10). Material from the climate
20 section of the ISA is principally drawn from the U.S. Climate Change Science Program
21 Synthesis and Assessment Product 2.3, *Atmospheric Aerosol Properties and Climate Impacts*, by
22 Chin et al., (CCSP 2009) and Chapter 2, *Changes in Atmospheric Constituents and in Radiative*
23 *Forcing*, (Forster et al., 2007) in the comprehensive *Working Group I report in the Fourth*
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25 *Change 2007: The Physical Science Basis*. Sections 9.3.7 (Fire as a Special Source of PM
26 Welfare Effects), 9.3.9 (Other Special Sources and Effects), 9.3.9.1 (Glaciers and Snowpack)
27 and 9.3.9.3 (Effects on Local and Regional Climate) of the ISA were written by NCEA staff.
28 This section of the PA summarizes and synthesizes the policy-relevant science in the ISA for the
29 purpose of helping to inform consideration of climate aspects in the review of the secondary PM
30 NAAQS.

31 Atmospheric PM (referred to as aerosols¹ in the remainder of this section to be consistent
32 with the ISA) affects multiple aspects of climate. These include absorbing and scattering of

¹ In the sections of the ISA included from IPCC AR4 and CCSP SAP2.3, ‘aerosols’ is more frequently used than “PM” and that word is retained.

1 incoming solar radiation, alterations in terrestrial radiation, effects on the hydrological cycle, and
2 changes in cloud properties (US EPA, 2009a, section 9.3.1). Major aerosol components that
3 contribute to climate processes include black carbon (BC), organic carbon (OC), sulfates, nitrates
4 and mineral dusts. There is a considerable ongoing research effort focused on understanding
5 aerosol contributions to changes in global mean temperature and precipitation patterns. The
6 Climate Change Research Initiative identified research on atmospheric concentrations and effects
7 of aerosols as a high research priority (National Research Council, 2001) and the IPCC 2007
8 *Summary for Policymakers* states that anthropogenic contributions to aerosols remain the
9 dominant uncertainty in radiative forcing (IPCC 2007). The current state of the science of
10 climate alterations attributed to PM is in flux as a result of continually updated information.

11 **5.2.2 Adequacy of the Current Standard**

12 In considering the adequacy of the suite of secondary standards, staff addresses the
13 following overarching question:

14 **Does currently available scientific information, as reflected in the ISA, support or call into**
15 **question the adequacy of the protection for climate effects afforded by the current suite of**
16 **secondary PM standards?**

17 To inform the answer to this overarching question, staff has posed specific questions to
18 aid in assessing the available scientific evidence as related to climate effects attributed to
19 aerosols. In considering the currently available scientific and technical information, we included
20 both the information available from the last review and information that is newly available since
21 the last review synthesized in Chapter 9 of the ISA (US EPA, 2009a).

22 • **What new techniques are available to improve our understanding of climate effects of** 23 **aerosols?**

24 Global climate change has increasingly been the focus of intense international research
25 endeavors. Major efforts are underway to understand the complexities inherent in atmospheric
26 aerosol interactions and to decrease uncertainties associated with climate estimations. Two recent
27 reports, the US CCSP Product 2.3 and sections of the IPCC AR4 were combined to form the
28 climate discussion in the ISA (CCSP 2009; Forster et al., 2007). A review of the most recently
29 available techniques for assessing climate-aerosol relationships is presented in the ISA. Aerosol
30 measurement capabilities reviewed in the ISA include a discussion of the increasingly
31 sophisticated instrumentation and techniques available for quantifying aerosols, the enhanced
32 sensing capabilities of satellites, development of remote sensing networks and synergy of
33 measurements with model simulations (US EPA 2009a, section 9.3.2). Advances in measured
34 aerosol properties as related to modeling as well as outstanding issues remaining in these
35 measurement-based studies are elaborated in the ISA (US EPA 2009a sections 9.3.3 and 9.3.4).

1 Section 9.3.6 of the ISA, “Global Aerosol Modeling” considers the capabilities of climate
2 modeling that have developed over the last decade and limitations of the techniques currently in
3 use (US EPA 2009a).

4 • **To what extent does newly available evidence improve our understanding of the nature
5 and magnitude of climate responses to PM (aerosols)?**

6 Aerosols have direct and indirect effects on climate processes. The direct effects of
7 aerosols on climate result mainly from particles scattering light away from earth into space,
8 directly altering the radiative balance of the Earth-atmosphere system. This reflection of solar
9 radiation back to space decreases the transmission of visible radiation to the surface of the earth
10 and results in a decrease in the heating rate of the surface and the lower atmosphere. At the same
11 time, absorption of either incoming solar radiation or outgoing terrestrial radiation by particles,
12 primarily BC, results in an increased heating rate in the lower atmosphere. Global estimates of
13 aerosol direct radiative forcing (RF) were recently summarized using a combined model-based
14 estimate (Forster et al., 2007). The overall, model-derived aerosol direct RF was estimated in the
15 IPCC AR4 as -0.5 (-0.9 to -0.1) watts per square meter (W/m^2), with an overall level of scientific
16 understanding of this effect as “medium low” (Forster et al., 2007), indicating a net cooling
17 effect in contrast to greenhouse gases (GHGs) which have a warming effect.

18 The contribution of individual aerosol components to total aerosol direct radiative forcing
19 is more uncertain than the global average (US EPA, 2009a, section 9.3.6.6). The direct effect of
20 radiative scattering by atmospheric particles exerts an overall net cooling of the atmosphere,
21 while particle absorption of solar radiation leads to warming. For example, the presence of OC
22 and sulfates decrease warming from sunlight by scattering shortwave radiation back into space.
23 Such a perturbation of incoming radiation by anthropogenic aerosols is designated as aerosol
24 climate forcing, which is distinguished from the aerosol radiative effect of the total aerosol
25 (natural plus anthropogenic). The aerosol climate forcing and radiative effect are characterized
26 by large spatial and temporal heterogeneities due to the wide variety of aerosol sources, the
27 spatial non-uniformity and intermittency of these sources, the short atmospheric lifetime of
28 aerosols (relative to that of the greenhouse gases), and processing (chemical and microphysical)
29 that occurs in the atmosphere. For example, OC can be warming (positive forcer) when
30 deposited on or suspended over a highly reflective surface such as snow or ice but, on a global
31 average, is a negative forcer in the atmosphere.

32 More information has also become available on indirect effects of aerosols. Particles in
33 the atmosphere indirectly affect both cloud albedo (reflectivity) and cloud lifetime by modifying
34 the cloud amount, and microphysical and radiative properties (US EPA, 2009a, section 9.3.6.4).
35 The RF due to these indirect effects (cloud albedo effect) of aerosols is estimated in the IPCC
36 AR4 to be -0.7(-1.8 to -0.3) W/m^2 with the level of scientific understanding of this effect as

1 “low” (Forster et al., 2007). Aerosols act as cloud condensation nuclei (CCN) for cloud
2 formation. Increased particulates in the atmosphere available as CCN with no change in
3 moisture content of the clouds have resulted in an increase in the number and decrease in the size
4 of cloud droplets in certain clouds that can increase the albedo of the clouds (the Twomey
5 effect). Smaller particles slow the onset of precipitation and prolong cloud lifetime. This effect,
6 coupled with changes in cloud albedo, increase the reflection of solar radiation back into space.
7 The altitude of clouds also effects cloud radiative forcing. Low clouds reflect incoming sunlight
8 back to space but do not effectively trap outgoing radiation, thus, cooling the planet, while higher
9 elevation clouds reflect some sunlight but more effectively can trap outgoing radiation and act to
10 warm the planet (US EPA, 2009a, section 9.3.3.5).

11 The total negative RF due to direct and indirect effects of aerosols computed from the top
12 of the atmosphere, on a global average, is estimated at -1.3 (-2.2 to -0.5) W/m² in contrast to the
13 positive RF of +2.9 (+3.2 to +2.6) W/m² for anthropogenic GHGs (IPCC 2007, pg. 200).

14 The understanding of the magnitude of aerosol effects on climate has increased
15 substantially in the last decade. Data on the atmospheric transport and deposition of aerosols
16 indicate a significant role for PM components in multiple aspects of climate. Aerosols can
17 impact glaciers, snowpack, regional water supplies, precipitation and climate patterns (US EPA
18 2009a section 9.3.9). Aerosols deposited on ice or snow can lead to melting and subsequent
19 decrease of surface albedo (US EPA 2009a, section 9.3.9.2). Aerosols are potentially important
20 agents of climate warming in the Arctic and other locations (US EPA, 2009a, section 9.3.9).
21 Incidental fires and biomass burning are being recognized as having a significant impact on
22 PM_{2.5} concentrations and climate forcing. Intermittent fires can occur at large enough scales to
23 affect hemispheric aerosol concentrations (US EPA 2009a, section 9.3.7).

24 A series of studies available since the last review examine the role of aerosols on local
25 and regional scale climate processes (US EPA, 2009a, section 9.3.9.3). Studies on the South
26 Coast Air Basin (SCAB) in California indicate aerosols may reduce near-surface wind speeds,
27 which, in turn reduce evaporation rates and increase cloud lifetimes. The overall impact can be a
28 reduction in local precipitation (Jacobson and Kaufmann, 2006). Conditions in the SCAB impact
29 ecologically sensitive areas including the Sierra Nevadas. Precipitation suppression due to
30 aerosols in California (Givati and Rosenfield, 2004) and other similar studies in Utah and
31 Colorado found that orographic precipitation decreased by 15-30% downwind of pollution
32 sources. Evidence of regional-scale impacts of aerosols on meteorological conditions in other
33 regions of the U.S. are lacking.

1 • **To what extent does the currently available information provide evidence of association**
2 **between specific PM constituents (i.e. BC, OC, sulfates) and climate-related effects?**

3 Advances in the understanding of aerosol components and how they contribute to climate
4 change have enabled refined global forcing estimates of individual PM constituents. The global
5 mean radiative effect from individual components of aerosols was estimated for the first time in
6 the IPCC AR4 where they were reported to be (all in W/m² units): -0.4 (± 0.2) for sulfate, -0.05
7 (± 0.05) for fossil fuel-derived OC, +0.2 (+0.15) for fossil fuel derived BC, +0.03 (± 0.12) for
8 biomass burning, -0.1 (± 0.1) for nitrates, and -0.1 (± 0.2) for mineral dust (US EPA, 2009a,
9 section 9.3.10). Sulfate and fossil fuel-derived OC cause negative forcing whereas BC causes
10 positive forcing because of its highly absorbing nature (US EPA, 2009a, 9.3.6.3). Although BC
11 comprises only a small fraction of anthropogenic aerosol mass load and aerosol optical depth
12 (AOD), its forcing efficiency (with respect to either AOD or mass) is an order of magnitude
13 stronger than sulfate and particulate organic matter (POM), so its positive shortwave forcing
14 largely offsets the negative direct forcing from sulfate and POM (IPCC, 2007; US EPA 2009a,
15 9.3.6.3). Global loadings for nitrates and anthropogenic dust remain very difficult to estimate,
16 making the radiative forcing estimates for these constituents particularly uncertain (US EPA,
17 2009a, section 9.3.7).

18 Improved estimates of anthropogenic emissions of some aerosols, especially BC and OC,
19 have promoted the development of improved global emissions inventories and source-specific
20 emissions factors useful in climate modeling (Bond et al. 2004). Recent data suggests that BC is
21 one of the largest individual warming agents after carbon dioxide (CO₂) and perhaps methane
22 (CH₄) (Jacobson 2000; Sato et al., 2003; Bond and Sun 2005). There are several studies
23 modeling BC effects on climate and/or considering emission reduction measures on
24 anthropogenic warming detailed in section 9.3.9 of the ISA. Fires release large amounts of BC,
25 CO₂, CH₄ and OC (US EPA, 2009a, section 9.3.7).

26 **5.2.3 Staff Conclusions**

- 27 • Aerosols alter climate processes directly through radiative forcing and by indirect
28 effects on cloud brightness, changes in precipitation and possible changes in cloud
29 lifetimes.
- 30 • Individual components of aerosols differ in their reflective properties, and direction of
31 climate forcing. Overall, based on current estimates of aerosol radiative forcing,
32 aerosols have a net climate cooling effect.
- 33 • Most climate model simulations are based on global scale scenarios. These models
34 may fail to consider the local variations in climate forcing due to emissions sources and
35 local meteorological patterns.

- Most of the warming aerosols in the U.S. are emitted by biomass burning and internal engine combustion. Much of the cooling aerosols are formed in the atmosphere by oxidation of SO₂ or VOC's. The relative mix and sources of warming and cooling components will vary in areas across the U.S. and over time. Thus, a set of controls to reduce warming PM would not necessarily reduce cooling PM and vice versa.

Collectively taking into consideration the responses to specific questions regarding the adequacy of the current secondary PM standards for climate effects, we revisit the overarching question: “does available scientific information, as reflected in the ISA, support or call into question the adequacy of the protection for climate effects afforded by the current suite of secondary PM standards?” As an initial matter, we considered the appropriateness of the current secondary standard defined in terms of PM_{2.5} and PM₁₀ indicators, for providing protection against potential climate effects of aerosols. Newly available scientific information on climate-aerosol relationships has improved our understanding of direct and indirect effects of aerosols and aerosol properties. The major aerosol components that contribute to climate processes include BC, OC, sulfate, nitrate and mineral dusts. These components vary in their reflectivity, forcing efficiencies and even in the direction of climate forcing. The current standards that are defined in terms of aggregate size mass cannot be expected to appropriately target controls on components of fine and coarse particles that are related to climate forcing effects. Thus, the current mass-based PM_{2.5} and PM₁₀ secondary standards are not an appropriate or effective means of focusing protection against PM-associated climate effects due to these differences in components.

Overall, there is a net climate cooling associated with aerosols in the global atmosphere (US EPA, 2009a, section 9.2.10). Staff recognizes that some individual aerosol components, such as BC, are positive climate forcers, whereas others, such as OC and sulfates, are negative climate forcers. The relative mix of components will vary in areas across the U.S. and over time. Due to the spatial and temporal heterogeneity of PM components that contribute to climate forcing, uncertainties in the measurement of aerosol components, inadequate consideration of aerosol impacts in climate modeling, insufficient data on local and regional microclimate variations and heterogeneity of cloud formations, it is not currently feasible to conduct a quantitative analysis for the purpose of informing revisions of the current NAAQS PM standard based on climate. Based on these considerations, we conclude that there is insufficient information at this time to base a national ambient standard on climate impacts associated with current ambient concentrations of PM or its constituents².

² Given the reasons discussed above, this conclusion would apply for both the secondary (welfare based) and the primary (health based) standards.

1 **5.2.4 Key Uncertainties and Areas for Future Research and Data Collection**

2 Although considerable progress is being made in estimating aerosol contributions to
3 radiative forcing and climate fluctuations, significant uncertainties remain that preclude
4 consideration of climate effects as a basis for establishing a separate NAAQS secondary
5 standard. Further research into the effects of aerosols on climate could provide important
6 information to reduce these uncertainties.

7 A major impediment at this time to establishing a secondary standard for PM based on
8 climate is the lack of accurate measurement of aerosol contributions, specifically quantification
9 of aerosol absorption and inability to separate the anthropogenic component from total aerosol
10 forcing. Section 9.3.4 of the ISA details the current limitations in aerosol measurement. Most
11 measurement studies focus on the sum of natural and anthropogenic contributions under clear
12 sky conditions, however, this scenario is simplistic when effects of cloud cover and differing
13 reflective properties of land and ocean are considered. Satellite measurements do not currently
14 have the capability to distinguish anthropogenic from natural aerosols. Due to a lack of data on
15 the vertical distribution of aerosols, above-cloud aerosols and profiles of atmospheric radiative
16 heating are poorly understood (US EPA, 2009a, section 9.3.4).

17 Another uncertainty in considering climate effects of PM in the NAAQS review is the
18 spatial and temporal heterogeneity of aerosols. In regions having high concentrations of
19 anthropogenic aerosols, aerosol forcing is greater than the global average, and can exceed
20 warming by GHGs, locally reversing the sign of the forcing (US EPA, 2009a, section 9.3.1).
21 The contributions of policy-relevant background (PRB) concentrations to aerosol climate forcing
22 are not sufficiently characterized (US EPA, 2009a, section 3.7). Emissions of carbonaceous
23 aerosols from intermittent fires and volcanic activity can further complicate regional climate
24 forcing estimates (US EPA, 2009a, sections 9.3.7 and 9.3.8). Individual components of aerosols
25 may either be positive or negative climate forcers. Airborne PM components may be directly
26 emitted or undergo a variety of physical and chemical interactions and transformations. These
27 result in changes in particle size, structure and composition which alter aerosol reflective
28 properties. Aerosols can grow in size in the atmosphere because ambient water vapor condenses
29 on individual particles, a phenomenon known as hygroscopic growth (US EPA, 2009a, section
30 9.3.6.2). Atmospheric lifetimes of individual aerosol components vary greatly confounding
31 tracking source receptor relationships.

32 Improved representation of aerosols in climate models is essential to more accurately
33 predict the role of PM in climate forcing (US EPA, 2009a, section 9.3.6.7). The influence of
34 aerosols on climate is not yet adequately taken into account in computer predictions although
35 considerable progress in being made in this area. For example, PM components underrepresented
36 or missing from many models include nitrate aerosols and anthropogenic secondary aerosols (US

1 EPA, 2009a, section 9.3.6.7). The modeling of aerosol indirect effects and absorption are
2 difficult due to the high level of uncertainty associated with these climate factors.

3 The interaction of PM with clouds remains a large source of uncertainty in climate
4 estimates. The interactions of aerosols with clouds and linkages between clouds and the overall
5 climate system are complex and limit the feasibility of conducting quantitative analysis for the
6 purpose of establishing a secondary PM standard based on welfare effects on climate processes.

7 There are uncertainties associated with the potential effects of the alternative standards
8 for visibility discussed in Chapter 4 on regional radiative forcing and climate. A secondary
9 standard for visibility based on light extinction would result in reduced emissions that affect PM
10 in areas where monitoring shows exceedance of the standard. The extinction budget work
11 conducted for the UFVA (Figure 3-13, U.S. EPA 2010b) and second draft PA (Appendix B)
12 indicates that most of the current visibility impact contributions on worst days comes from light
13 scattering particles (e.g., nitrates, sulfates) that are negative climate forcers, and a smaller portion
14 from absorbing aerosols (e.g., black carbon) that are positive climate forcers. The relative
15 proportions of scattering and absorbing particles vary by location and some major contributing
16 emission sources contribute to both scattering and absorbing PM, so it is unclear how the ratio of
17 scattering to absorption might change in response to a secondary standard for visibility affects.
18 However, since the prevailing mixture of aerosol is thought to have a net cooling effect on
19 regional climate, reducing PM and light scattering aerosols could lead to increased radiative
20 forcing and regional climate warming while having a beneficial effect on visibility.

21 **5.3 ECOLOGICAL EFFECTS**

22 **5.3.1 Scope**

23 Information on what is currently known about ecological effects of PM is summarized in
24 Chapter 9 of the ISA (US EPA 2009a). Four main categories of ecological effects are identified
25 in the ISA: direct effects, effects of PM-altered radiative flux, indirect effects of trace metals and
26 indirect effects of organics. Exposure to PM for direct effects occur via deposition (e.g. wet, dry
27 or occult) to vegetation surfaces, while indirect effects occur via deposition to ecosystem soils or
28 surface waters where the deposited constituents of PM then interacts with biological organisms.
29 Both fine and coarse-mode particles may affect plants and other organisms; however, PM size
30 classes do not necessarily relate to ecological effects (U.S. EPA, 1996). More often the chemical
31 constituents drive the ecosystem response to PM (Grantz et al., 2003). The trace metal
32 constituents of PM considered in the ecological effects section of the ISA are cadmium (Cd),
33 copper (Cu), chromium (Cr), mercury (Hg), nickel (Ni) and zinc (Zn). Ecological effects of lead
34 (Pb) in particulate form are covered in the Air Quality Criteria Document for Lead (US EPA,
35 2006). The organics included in the ecological effects section of the ISA are persistent organic

1 pollutants (POPs), polyaromatic hydrocarbons (PAHs) and polybromiated diphenyl ethers
2 (PBDEs).

3 Ecological effects of PM include direct effects to metabolic processes of plant foliage;
4 contribution to total metal loading resulting in alteration of soil biogeochemistry and
5 microbiology, plant and animal growth and reproduction; and contribution to total organics
6 loading resulting in bioaccumulation and biomagnification across trophic levels. It is important
7 to emphasize that the metal and organic constituents of PM contribute to total metal and organic
8 loads in ecosystems.

9 The ISA states that overall, ecological evidence is sufficient to conclude that a causal
10 relationship is likely to exist between deposition of PM and a variety of effects on individual
11 organisms and ecosystems based on information from the previous review and limited new
12 findings in this review (US EPA 2009a, sections 2.5.3 and 9.4.7). However the ISA also finds,
13 in many cases, it is difficult to characterize the nature and magnitude of effects and to quantify
14 relationships between ambient concentrations of PM and ecosystem response due to significant
15 data gaps and uncertainties as well as considerable variability that exists in the components of
16 PM and their various ecological effects.

17 Ecological effects of PM must then be evaluated to determine if they are known or
18 anticipated to have an adverse impact on public welfare. Characterizing a known or anticipated
19 adverse effect to public welfare is an important component of developing any secondary
20 NAAQS. The most recent secondary NAAQS reviews have assessed changes in ecosystem
21 structure or processes using a weight-of-evidence approach that uses both quantitative and
22 qualitative data. For example, the 2008 ozone (O₃) final rule and 2010 O₃ proposal conclude that
23 a determination of what constitutes an “adverse” welfare effect in the context of secondary
24 NAAQS review can appropriately occur by considering effects at higher ecological levels
25 (populations, communities, ecosystems) as supported by recent literature. In the 2008
26 rulemaking and current ozone proposal, the interpretation of what constitutes an adverse effect
27 on vegetation can vary depending on the location and intended use of the plant. The degree to
28 which O₃-related effects are considered adverse depends on the intended use of the vegetation
29 and its significance to public welfare (73 FR 16496). Therefore, effects (e.g. biomass loss, foliar
30 injury, impairment of intended use) may be judged to have a different degree of impact on public
31 welfare depending, for example, on whether that effect occurs in a Class I area, a city park,
32 commercial cropland or private land.

33 A paradigm useful in evaluating ecological adversity is the concept of ecosystem
34 services. Ecosystem services identify the varied and numerous ways that ecosystems are
35 important to human welfare. Ecosystems provide many goods and services that are of vital
36 importance for the functioning of the biosphere and provide the basis for the delivery of tangible

1 benefits to human society. An EPA initiative to consider how ecosystem structure and function
2 can be interpreted through an ecosystem services approach has resulted in the inclusion of
3 ecosystem services in the NO_x/SO_x REA (US EPA, 2009h). The Millennium Ecosystem
4 Assessment (MEA) defines these to include supporting, provisioning, regulating and cultural
5 services (Hassan et al., 2005):

- 6 • Supporting services are necessary for the production of all other ecosystem services.
7 Some examples include biomass production, production of atmospheric O₂, soil
8 formation and retention, nutrient cycling, water cycling, and provisioning of habitat.
9 Biodiversity is a supporting service that is increasingly recognized to sustain many of
10 the goods and services that humans enjoy from ecosystems. These provide a basis for
11 three higher-level categories of services.
- 12 • Provisioning services, such as products (Gitay et al., 2001) i.e., food (including game,
13 roots, seeds, nuts, and other fruit, spices, fodder), fiber (including wood, textiles), and
14 medicinal and cosmetic products (including aromatic plants, pigments).
- 15 • Regulating services that are of paramount importance for human society such as (a) C
16 sequestration, (b) climate and water regulation, (c) protection from natural hazards
17 such as floods, avalanches, or rock-fall, (d) water and air purification, and (e) disease
18 and pest regulation.
- 19 • Cultural services that satisfy human spiritual and aesthetic appreciation of ecosystems
20 and their components.

21 An important consideration in evaluating biologically adverse effects of PM and linkages
22 to ecosystem services is that many of the MEA categories overlap and any one pollutant may
23 impact multiple services. For example, deposited PM may alter the composition of soil-
24 associated microbial communities, which may affect supporting services such as nutrient
25 cycling. Changes in available soil nutrients could result in alterations to provisioning services
26 such as timber yield and regulating services such as climate regulation. If enough information is
27 available, these alterations can be quantified based upon economic approaches for estimating the
28 value of ecosystem services. Valuation may be important from a policy perspective because it
29 can be used to compare the benefits of altering versus maintaining an ecosystem. Knowledge
30 about the relationships linking ambient concentrations and ecosystem services can be used to
31 inform a policy judgment on a known or anticipated adverse public welfare effect.

32 This review seeks to build upon and focus this body of science using the concept of
33 ecosystem services to qualitatively evaluate linkages between biologically adverse effects and
34 particulate deposition. This approach is similar to that taken in the NO_x/SO_x REA in which the
35 relationship between air quality indicators, deposition of N and S, ecologically relevant
36 indicators and effects on sensitive receptors are linked to changes in ecosystem structure and
37 services (US EPA, 2009h). This approach considers the benefits received from the resources and

1 processes that are supplied by ecosystems. Ecosystem components (e.g. plants, soils, water,
2 wildlife) are impacted by PM air pollution, which may alter the services provided by the
3 ecosystems in question. The goals of this policy assessment are to (1) identify ecological effects
4 associated with PM deposition that can be linked to ecosystem services and (2) qualitatively
5 evaluate ecological endpoints when possible. Keeping these goals and guidelines in mind,
6 limited new data on PM effects on plants, soil and nutrient cycling, wildlife and water are
7 evaluated in the context of ecosystem services to qualitatively evaluate linkages between
8 biologically adverse effects and particulate deposition for the purpose of evaluating the adequacy
9 of the current standard.

10 **5.3.2 Adequacy of the Current Standard**

11 In considering the adequacy of the suite of secondary standards, staff addresses the
12 following overarching question:

13 **Does available scientific information, as reflected in the ISA support or call into question**
14 **the adequacy of the protection afforded by the current suite of secondary PM standards for**
15 **vegetation and ecosystems from the effects of deposited particulate metals and organics?**

16 To inform the answer to this overarching question, staff has posed specific questions to
17 aid in assessing the available scientific evidence as related to ecosystem effects attributed to PM
18 deposition as presented in the ISA (US EPA, 2009a).

- 19 • **To what extent has key scientific evidence become available to improve our**
20 **understanding of the nature and magnitude of ecosystem responses, the variability**
21 **associated with these responses, and the impact of PM on ecosystem services?**

22 Key scientific evidence regarding PM effects on plants, soil and nutrient cycling, wildlife
23 and water available since the last review is summarized below to evaluate how this information
24 has improved our understanding of ecosystem responses to PM.

25 Plants

26 As primary producers, plants play a pivotal role in energy flow through ecosystems.
27 Ecosystem services derived from plants include all of the categories (supporting, provisioning,
28 regulating, cultural) identified in the MEA (Hassan et al., 2005). Vegetation supports other
29 ecosystem processes by cycling nutrients through food webs and serving as a source of organic
30 material for soil formation and enrichment. Trees and plants provide food, wood, fiber, and fuel
31 for human consumption. Flora help to regulate climate by sequestering CO₂, control flooding by
32 stabilizing soils and cycling water via uptake and evapotranspiration. Plants are significant in
33 aesthetic, spiritual and recreational aspects of human interactions.

1 Particulate matter can adversely impact plants and ecosystem services provided by plants
2 by deposition to vegetative surfaces (US EPA, 2009a, section 9.4.3). Particulates deposited on
3 the surfaces of leaves and needles can block light, altering the radiation received by the plant.
4 PM deposition can obstruct stomata limiting gas exchange, damage leaf cuticles and increase
5 plant temperatures. This level of PM accumulation is typically observed near sources of heavy
6 deposition such as smelters and mining operations (US EPA, 2009a, section 9.4.3). Plants
7 growing on roadsides exhibit impact damage from near-road PM deposition, having higher levels
8 of organics and heavy metals, and accumulate salt from road de-icing during winter months (US
9 EPA, section 2009a, sections 9.4.3.1 and 9.4.5.7).

10 In addition to damage to plant surfaces, deposited PM can be taken up by plants from soil
11 or foliage. The ability of vegetation to take up heavy metals and organics is dependent upon the
12 amount, solubility and chemical composition of the deposited PM. Uptake of PM by plants from
13 soils and vegetative surfaces can disrupt photosynthesis, alter pigments and mineral content,
14 reduce plant vigor, decrease frost hardiness and impair root development. The ISA indicates that
15 there are little or no effects on foliar processes at ambient levels of PM (sections 9.4.3 and 9.4.7)
16 however, damage due to atmospheric pollution can occur near point-sources or under conditions
17 where plants are subjected to multiple stressors.

18 Though all heavy metals can be directly toxic at sufficiently high concentrations, only
19 Cu, Ni, and Zn have been documented as being frequently toxic to plants (U.S. EPA, 2004),
20 while toxicity due to Cd, Co, and Pb has been observed less frequently (Smith, 1990; US EPA
21 2009a, section 9.4.5.3). In general, plant growth is negatively correlated with trace metal and
22 heavy metal concentration in soils and plant tissue (Audet and Charest, 2007). Trace metals,
23 particularly heavy metals, can influence forest growth. Growth suppression of foliar microflora
24 has been shown to result from Fe, Al, and Zn. These three metals can also inhibit fungal spore
25 formation, as can Cd, Cr, Mg, and Ni (see Smith, 1990). Metals cause stress and decreased
26 photosynthesis (Kucera et al., 2008) and disrupt numerous enzymes and metabolic pathways
27 (Strydom et al., 2006). Excessive concentrations of metals result in phytotoxicity through: (i)
28 changes in the permeability of the cell membrane; (ii) reactions of sulfhydryl (-SH) groups with
29 cations; (iii) affinity for reacting with phosphate groups and active groups of ADP or ATP; and
30 (iv) replacement of essential ions (Patra et al., 2004).

31 New information since the last review provides additional evidence of plant uptake of
32 organics (US EPA, 2009a, section 9.4.6). An area of active study is the impact of PAHs on
33 provisioning ecosystem services due to the potential for human and other animal exposure via
34 food consumption (US EPA, 2009a, section 9.4.6 page 9-190). The uptake of PAHs depends on
35 the plant species, site of deposition, physical and chemical properties of the organic compound
36 and prevailing environmental conditions. It has been established that most bioaccumulation of

1 PAHs by plants occurs via leaf uptake, and to a lesser extent, through roots. Differences
2 between species in uptake of PAHs confound attempts to quantify impacts to ecosystem
3 provisioning services. For example, zucchini (*Cucurbita pepo*) accumulated significantly more
4 PAHs than related plant species (Parrish et al., 2006).

5 Plants as ecosystem regulators can serve as passive monitors of pollution (US EPA,
6 2009a, section 9.4.2.3). Lichens and mosses are sensitive to pollutants associated with PM and
7 have been used with limited success to show spatial and temporal patterns of atmospheric
8 deposition of metals (US EPA, 2009a, section 9.4.2.3). For example, the presence or absence of
9 a specific species of lichen can be used as a bioindicator of metal or organics contamination.
10 PBDEs detected in moss and lichens in Antarctica indicate long-range transport of PM
11 components (Yogui and Sericano 2008). In the U.S. Blue Ridge Mountains, a study linked metal
12 concentrations in mosses to elevation and tree canopy species at some sites but not with
13 concentrations of metals in the O horizon of soil (Schilling, 2002). A limitation to employing
14 mosses and lichens to detect for the presence of air pollutants is the difference in uptake
15 efficiencies of metals between species. The European Moss Biomonitoring Network has been
16 shown to be useful in Europe for estimating general trends in metal concentrations and
17 identification of some sources of trace contaminants, however, quantification of ecological
18 effects is not possible due to the variability of species responses (US EPA, 2009a, section
19 9.4.2.3).

20 A potentially important regulating ecosystem service of plants is their capacity to
21 sequester contaminants (US EPA, 2009a, section 9.4.5.3). Ongoing research on the application
22 of plants to environmental remediation efforts are yielding some success in removing heavy
23 metals and organics from contaminated sites (phytoremediation) with tolerant plants such as the
24 willow tree (*Salix* spp.) and members of the family Brassicaceae (US EPA, 2009a, section
25 9.4.5.3). Tree canopies can be used in urban locations to capture particulates and improve air
26 quality (Freer-Smith et al., 2004). Plant foliage is a sink for Hg and other metals and this
27 regulating ecosystem service may be impacted by atmospheric deposition of trace metals.

28 An ecological endpoint (phytochelatin concentration) associated with presence of metals
29 in the environment has been correlated with the ecological effect of tree mortality (Grantz et al.,
30 2003). Metal stress may be contributing to tree injury and forest decline in the Northeastern U.S.
31 where red spruce populations are declining with increasing elevation. Quantitative assessment of
32 PM damage to forests potentially could be conducted by overlaying PM sampling data and
33 elevated phytochelatin levels. However, limited data on phytochelatin levels in other species
34 currently hinders use of this peptide as a general biomarker for PM.

35 The presence of PM in the atmosphere affects ambient radiation as discussed in the ISA
36 which can impact the amount of sunlight received by plants (US EPA, 2009a, section 9.4.4).

1 Atmospheric PM can change the radiation reaching leaf surfaces through attenuation and by
2 converting direct radiation to diffuse radiation. Diffuse radiation is more uniformly distributed in
3 a tree canopy, allowing radiation to reach lower leaves. The net effect of PM on photosynthesis
4 depends on the reduction of photosynthetically active radiation (PAR) and the increase in the
5 diffuse fraction of PAR. Decreases in crop yields (provisioning ecosystem service) have been
6 attributed to regional scale air pollution, however, global models suggest that the diffuse light
7 fraction of PAR can increase growth (US EPA, 2009a, section 9.4.4).

8 Soil and Nutrient Cycling

9 Many of the major indirect plant responses to PM deposition are chiefly soil-mediated
10 and depend on the chemical composition of individual components of deposited PM. Major
11 ecosystem services impacted by PM deposition to soils include support services such as nutrient
12 cycling, products such as crops and regulating flooding and water quality. Upon entering the soil
13 environment, PM pollutants can alter ecological processes of energy flow and nutrient cycling,
14 inhibit nutrient uptake to plants, change microbial community structure and, affect biodiversity.
15 Accumulation of heavy metals in soils depends on factors such as local soil characteristics,
16 geologic origin of parent soils, and metal bioavailability. It can be difficult to assess the extent
17 to which observed heavy metal concentrations in soil are of anthropogenic origin (US EPA,
18 2009a, section 9.4.5.1). Trace element concentrations are higher in some soils that are remote
19 from air pollution sources due to parent material and local geomorphology.

20 Heavy metals such as Zn, Cu, and Cd and some pesticides can interfere with
21 microorganisms that are responsible for decomposition of soil litter, an important regulating
22 ecosystem service that serves as a source of soil nutrients (US EPA, 2009a, sections 9.4.5.1 and
23 9.4.5.2). Surface litter decomposition is reduced in soils having high metal concentrations. Soil
24 communities have associated bacteria, fungi, and invertebrates that are essential to soil nutrient
25 cycling processes. Changes to the relative species abundance and community composition can
26 be quantified to measure impacts of deposited PM to soil biota. A mutualistic relationship exists
27 in the rhizosphere (plant root zone) between plant roots, fungi, and microbes. Fungi in
28 association with plant roots form mycorrhizae that are essential for nutrient uptake by plants.
29 The role of mycorrhizal fungi in plant uptake of metals from soils and effects of deposited PM
30 on soil microbes is discussed in section 9.4.5.2 of the ISA.

31 Wildlife

32 Animals play a significant role in ecosystem function including nutrient cycling and crop
33 production (supporting ecosystem service), and as a source of food (provisioning ecosystem
34 service). Cultural ecosystem services provided by wildlife include bird and animal watching,
35 recreational hunting and fishing. Impacts on these services are dependent upon the

1 bioavailability of deposited metals and organics and their respective toxicities to ecosystem
2 receptors. Pathways of PM exposure to fauna include ingestion, absorption and trophic transfer.
3 Bioindicator species (known as sentinel organisms) can provide evidence of contamination due
4 to atmospheric pollutants. Use of sentinel species can be of particular value because chemical
5 constituents of deposited PM are difficult to characterize and have varying bioavailability (US
6 EPA, 2009a, section 9.4.5.5). Snails readily bioaccumulate contaminants such as PAHs and
7 trace metals. These organisms have been deployed as biomonitors for urban pollution and have
8 quantifiable biomarkers of exposure including growth inhibition, impairment of reproduction,
9 peroxidomal proliferation and induction of metal detoxifying proteins (metallothioneins)
10 (Gomet-de Vaufleury, 2002; Regoli, 2006). Earthworms have also been used as sensitive
11 indicators of soil metal contamination.

12 Evidence of deposited PM effects on animals is limited (US EPA, 2009a, section 9.4.5.5).
13 Trophic transfer of pollutants of atmospheric origin has been demonstrated in limited studies.
14 PM may also be transferred between aquatic and terrestrial compartments. There is limited
15 evidence for biomagnifications of heavy metals up the food chain except for Hg which is well
16 known to move readily through environmental compartments (US EPA, 2009a section 9.4.5.6).
17 Bioconcentration of POPs and PBDEs in the Arctic and deep-water oceanic food webs indicates
18 the global transport of particle-associated organics (US EPA, 2009a, section 9.4.6). Salmon
19 migrations are contributing to metal accumulation in inland aquatic systems, potentially
20 impacting the provisioning and cultural ecosystem service of fishing (US EPA, 2009a, section
21 9.4.6). Stable isotope analysis can be applied to establish linkages between PM exposure and
22 impacts to food webs, however, the use of this evaluation tool is limited for this ecological
23 endpoint due to the complexity of most trophic interactions (US EPA 2009a, section 9.4.5.6).
24 Foraging cattle have been used to assess atmospheric deposition and subsequent bioaccumulation
25 of Hg and trace metals and their impacts on provisioning services (US EPA, 2009a, section
26 9.4.2.3).

27 Water

28 New limited information on impacts of deposited PM on receiving water bodies indicate
29 that the ecosystem services of primary production, provision of fresh water, regulation of climate
30 and floods, recreational fishing and water purification are adversely impacted by atmospheric
31 inputs of metals and organics (US EPA, 2009a, sections 9.4.2.3 and 9.4.5.4). Deposition of PM
32 to surfaces in urban settings increases the metal and organic component of storm water runoff
33 (US EPA, 2009a, sections 9.4.2.3). This atmospherically-associated pollutant burden can then be
34 toxic to aquatic biota.

1 Atmospheric deposition can be the primary source of some organics and metals to
2 watersheds. The contribution of atmospherically deposited PAHs to aquatic food webs was
3 demonstrated in high elevation mountain lakes with no other anthropogenic contaminant sources
4 (US EPA, 2009a, section 9.4.6). Metals associated with PM deposition limit phytoplankton
5 growth, impacting aquatic trophic structure. Long-range atmospheric transport of 47 pesticides
6 and degradation products to the snowpack in seven national parks in the Western U.S. was
7 recently quantified indicating PM-associated contaminant inputs to receiving waters during
8 spring snowmelt (Hageman et al., 2006).

9 • **What new techniques are available to improve our understanding of ecosystem effects**
10 **associated with metal and organic components of PM?**

11 The recently completed Western Airborne Contaminants Assessment Project (WACAP)
12 is the most comprehensive database on contaminant transport and PM depositional effects on
13 sensitive ecosystems in the U.S. In this project, the transport, fate, and ecological impacts of
14 anthropogenic contaminants from atmospheric sources were assessed from 2002 to 2007 in seven
15 ecosystem components (air, snow, water, sediment, lichen, conifer needles and fish) in eight
16 core national parks (Landers et al., 2008). The goals of the study were to identify where the
17 pollutants were accumulating, identify ecological indicators for those pollutants causing
18 ecological harm, and to determine the source of the air masses most likely to have transported
19 the contaminants to the parks (US EPA, 2009a, section 9.4.6). Collected data were analyzed to
20 identify probable local, regional and/or global sources of deposited PM components and their
21 concurrent effects on ecological receptors. The study concluded that bioaccumulation of semi-
22 volatile organic compounds (VOCs) was observed throughout park ecosystems (Landers et al.,
23 2008). Findings from this study included the observation of an elevational gradient in PM
24 deposition with greater accumulation at higher altitude areas of the parks. Furthermore, specific
25 ecological indicators were identified in the WACAP that can be useful in assessing
26 contamination on larger spatial scales. For example, quantification of concentrations of selected
27 pesticides in second-year conifer needles served as a method for regional-scale comparison of
28 pollutant distribution (Landers et al., 2008).

29 In the WACAP study, bioaccumulation and biomagnification of airborne contaminants
30 were demonstrated on a regional scale in remote ecosystems in the Western United States.
31 Contaminants were shown to accumulate geographically based on proximity to individual
32 sources or source areas, primarily agriculture and industry (Landers et al., 2008). This finding
33 was counter to the original working hypothesis that most of the contaminants found in western
34 parks would originate from Eastern Europe and Asia (Landers et al., 2008 p 6-8). The WACAP
35 study represents an experimental design in which ecological effects could be correlated to
36 ambient pollutant levels on a regional scale. Although this assessment focuses on chemical

1 species that are components of PM, it does not specifically assess the effects of particulates
2 versus gas-phase forms; therefore, in most cases it is difficult to apply the results to this
3 assessment based on particulate concentration and size fraction (US EPA, 2009a, section 9.4.6).
4 There is a need for ecological modeling of PM components in different environmental
5 compartments to further elucidate links between PM and ecological indicators.

6 Europe and other countries are using the critical load approach to assess pollutant effects
7 at the level of the ecosystem. This type of assessment requires site-specific data and information
8 on individual species responses to PM. In respect to trace metals and organics, there are
9 insufficient data for the vast majority of U.S. ecosystems to calculate critical loads, however, a
10 methodology is being presented in the NO_x/SO_x Secondary REA (US EPA 2009h) to calculate
11 atmospheric concentrations from deposition that may be applicable to other environmental
12 contaminants.

13 • **Is there currently available information on ambient levels of PM that cause adverse**
14 **effects on ecosystem components?**

15 As reviewed above, there is considerable data on impacts of PM on ecological receptors,
16 but few studies that link ambient PM levels to observed effect. This is due, in part, to the nature,
17 deposition, transport and fate of PM in ecosystems. PM is not a single pollutant, but a
18 heterogeneous mixture of particles differing in size, origin and chemical composition (US EPA,
19 2009a, section 9.4.1). The heterogeneity of PM exists not only within individual particles or
20 samples from individual sites, but to even a greater extent, between samples from different sites.
21 Since vegetation and other ecosystem components are affected more by particulate chemistry
22 than size fraction, exposure to a given mass concentration of airborne PM may lead to widely
23 differing plant or ecosystem responses, depending on the particular mix of deposited particles.

24 Many of the PM components bioaccumulate over time in organisms or plants, making
25 correlations to ambient levels of PM difficult. For example, in the WACAP study, SOC
26 accumulation in vegetation and air showed different patterns, possibly because each medium
27 absorbs different types of SOCs with varying efficiencies (Landers et al., 2008).

28 Bioindicator organisms demonstrated biological effects including growth inhibition,
29 metallothionein induction and reproductive impairment when exposed to complex mixtures of
30 ambient air pollutants (US EPA 2009a, section 9.4.5.5). Other studies quantify uptake of metals
31 and organics by plants or animals. However, due to the difficulty in correlating individual PM
32 components to a specific physiological response, these studies are limited. Furthermore, there
33 may be differences in uptake between species such as differing responses to metal uptake
34 observed in mosses and lichens (US EPA 2009a, section 9.4.2.3). PM may also biomagnify
35 across trophic levels confounding efforts to link atmospheric concentrations to physiological
36 endpoints (US EPA, 2009a, section 9.4.5.6).

1 Evidence of PM effects that are linked to a specific ecological endpoint can be observed
2 when ambient levels are exceeded. Most direct ecosystem effects associated with particulate
3 pollution occur in severely polluted areas near industrial point sources (quarries, cement kilns,
4 metal smelting) (US EPA, 2009a, sections 9.4.3 and 9.4.5.7). Extensive research on biota near
5 point sources provide some of the best evidence of ecosystem function impacts and demonstrates
6 that deposited PM has the potential to alter species composition over long time scales.
7 Ecological field studies conducted in proximity to Cu-Ni smelter in Harjavalta, Finland indicated
8 ecological structure and community composition are altered in response to PM and these effects
9 decrease with increasing distance from the point source (US EPA 2009a, section 9.4.5.7). The
10 ISA indicates at 4 km distance, species composition of vegetation, insects, birds, and soil
11 microbiota changed, and within 1 km only the most resistant organisms were surviving (US
12 EPA, 2009a, section 9.4.5.7). Heavy metal concentrations were quantified in understory plant
13 species growing at varying distance from the Harjavalta smelter (Salemaa et al., 2004). Heavy
14 metal concentrations were highest in bryophytes, followed by lichens and were lowest in
15 vascular plants. At the Harjavalta smelter there are clear links between PM deposition levels,
16 ecological endpoints and compromised ecosystem structure. However, these conditions are not
17 reflective of ambient concentrations of PM in the majority of US ecosystems (US EPA, 2009a,
18 section 9.4.7).

19 **5.3.3 Staff Conclusions**

- 20 • A number of significant environmental effects that either have already occurred or are
21 currently occurring are linked to deposition of chemical constituents found in ambient
22 PM.
- 23 • Ecosystem services can be adversely impacted by PM in the environment, including
24 supporting, provisioning, regulating and cultural services.
- 25 • The lack of sufficient information to relate specific ambient concentrations of
26 particulate metals and organics to a degree of impairment of a specific ecological
27 endpoint hinders our ability to identify a range of appropriate indicators, levels, forms
28 and averaging times of a distinct secondary standard to protect against associated
29 effects.
- 30 • Data from regionally-based ecological studies can be used to establish probable local,
31 regional and/or global sources of deposited PM components and their concurrent
32 effects on ecological receptors.

33 Taking into consideration the responses to specific questions regarding the adequacy of
34 the current secondary PM standards for ecological effects, we revisit the overarching question:
35 “does available scientific information, as reflected in the ISA, support or call into question the
36 adequacy of the protection for ecosystems afforded by the current suite of secondary PM
37 standards?” Staff concludes that the available information is insufficient to assess the adequacy

1 of the protection for ecosystems afforded by the current suite of PM secondary standards.
2 Ecosystem effects linked to PM are difficult to determine because the changes may not be
3 observed until pollutant deposition has occurred for many decades. Because the high levels
4 necessary to cause injury occur only near a few limited point sources and/or on a very local
5 scale, protection against these effects alone may not provide sufficient basis for considering a
6 separate secondary NAAQS based on the ecological effects of particulate metals and organics.
7 Data on ecological responses clearly linked with atmospheric PM is not abundant enough to
8 perform a quantitative analysis although the WACAP study may represent an opportunity for
9 quantification at a regional scale. At this time, we conclude that available evidence is not
10 sufficient for establishing a distinct national standard for ambient PM based on ecosystem effects
11 of particulates not addressed in the NO_x/SO_x secondary review (e.g. metals, organics).

12 Staff considered the appropriateness of continuing to use the PM_{2.5} and PM₁₀ size
13 fractions as the indicators for protection of ecological effects of PM. Though the chemical
14 constitution of individual particles can be strongly correlated with size, the relationship between
15 particle size and particle composition can also be quite complex, making it difficult in most cases
16 to use particle size as a surrogate for chemistry. At this time it remains to be determined as to
17 what extent PM secondary standards focused on a given size fraction would result in reductions
18 of the ecologically relevant constituents of PM for any given area. Nonetheless, in the absence of
19 information that provides a basis for specific standards in terms of particle composition,
20 observations continue to support retaining an appropriate degree of control on both fine and
21 coarse particles to help address effects to ecosystems and ecosystem components associated with
22 PM.

23 **5.3.4 Key Uncertainties and Areas for Future Research and Data Collection**

24 The above discussions identify linkages between ecological effects of deposited PM and
25 potential impacts to ecosystem services. Unfortunately, our ability to relate ambient
26 concentrations of PM to ecosystem response is hampered by a number of significant data gaps
27 and uncertainties. These limitations include the presence of multiple ecological stressors
28 confounding attempts to link specific ecosystem responses to PM deposition. These stressors
29 can be anthropogenic (e.g. habitat destruction, eutrophication, other pollutants) or natural (e.g.
30 drought, fire, disease). Deposited PM interacts with other stressors to affect ecosystem patterns
31 and processes. Furthermore, the environmental effects of deposited PM are decoupled in space
32 and time from the point of emission confounding efforts to identify ecological perturbations
33 attributed to PM deposition.

34 A second source of uncertainty lies in predicting the amount of PM deposited to sensitive
35 receptors from measured concentrations of PM in the ambient air. This makes it difficult to

1 relate a given air concentration to a receptor response, an important factor in being able to set a
2 national ambient air quality standard. A multitude of factors such as the mode of deposition
3 (wet, dry and occult), wind speed, surface roughness or stickiness, elevation, particle
4 characteristics (e.g. size, shape, chemical composition), and relative humidity exert varying
5 degrees of influence on the deposition velocities for different PM components in any point in
6 time. Composition of ambient PM varies in time and space and the particulate mixture may have
7 synergistic, antagonistic or additive effects on ecological receptors depending upon the chemical
8 species present. Furthermore, presence of co-occurring pollutants make it difficult to attribute
9 observed effects to ecological receptors to PM alone or one component of deposited PM.

10 Third, each ecosystem has developed within a context framed by the topography,
11 underlying bedrock, soils, climate, meteorology, hydrologic regime, natural and land use history,
12 and species composition that make it unique from all others. Sensitivity of ecosystem response
13 is highly variable in space and time. Because of this variety and lack of sufficient baseline data
14 on each of these features for most ecosystems, it is currently not possible to extrapolate with
15 confidence any effect from one ecosystem to another. Further research is needed to decrease the
16 uncertainties associated with ambient PM effects on ecosystems and ecosystem components.

17 **5.4 MATERIALS**

18 **5.4.1 Scope**

19 Welfare effects on materials associated with deposition of PM include both physical
20 damage (materials damage effects) and impaired aesthetic qualities (soiling effects). Because the
21 effects of PM are exacerbated by the presence of acidic gases and can be additive or synergistic
22 due to the complex mixture of pollutants in the air and surface characteristics of the material, this
23 discussion will also include those particles and gases that are associated with the presence of
24 ambient NO_x and SO_x, as well as NH₃ and NH_x for completeness. Building upon the
25 information presented in the last Staff Paper (US EPA, 2005), and including the limited new
26 information presented in Chapter 9 of the PM ISA (US EPA, 2009a) and *Annex E. Effects of*
27 *NO_y, NH_x, and SO_x on Structures and Materials of the Integrated Science Assessment for*
28 *Oxides of Nitrogen and Sulfur-Ecological Criteria (NO_x/SO_x ISA)* (US EPA, 2008c) the
29 following sections consider the policy-relevant aspects of physical damage and aesthetic soiling
30 effects of PM on materials including metal and stone.

31 The ISA concludes that evidence is sufficient to support a causal relationship between
32 PM and effects on materials (US EPA, 2009a, sections 2.5.4 and 9.5.4). The deposition of PM
33 can physically affect materials, adding to the effects of natural weathering processes, by
34 potentially promoting or accelerating the corrosion of metals, by degrading paints and by
35 deteriorating building materials such as stone, concrete and marble (US EPA, 2009a, section

1 9.5). Particles contribute to these physical effects because of their electrolytic, hygroscopic and
2 acidic properties, and their ability to sorb corrosive gases (principally SO₂). In addition, the
3 deposition of ambient PM can reduce the aesthetic appeal of buildings and objects through
4 soiling. Particles consisting primarily of carbonaceous compounds cause soiling of commonly
5 used building materials and culturally important items such as statues and works of art. Soiling
6 is the deposition of particles on surfaces by impingement, and the accumulation of particles on
7 the surface of an exposed material results in degradation of its appearance (US EPA 2009a,
8 section 9.5). Soiling can be remedied by cleaning or washing, and depending on the soiled
9 material, repainting.

10 **5.4.2 Adequacy of the Current Standard**

11 In considering the adequacy of the suite of secondary standards, staff addresses the
12 following overarching question:

13 **Does available scientific information, as reflected in the ISA support or call into question**
14 **the adequacy of the protection for materials afforded by the current suite of secondary PM**
15 **standards?**

16 To inform the answer to this overarching question, staff has posed a specific question to
17 aid in assessing the available scientific evidence as related to materials damage and soiling
18 attributed to PM deposition as presented in the ISA (US EPA, 2009a).

19 • **What new evidence is available to improve our understanding of effects of PM on**
20 **materials and linking ambient concentrations to materials damage?**

21 The majority of available new studies on materials effects of PM are from outside the
22 U.S., however, they provide limited new data for consideration of the secondary standard.

23 Metal and stone are susceptible to damage by ambient PM. Considerable research has
24 been conducted on the effects of air pollutants on metal surfaces due to the economic importance
25 of these materials, especially steel, zinc, aluminum, and copper. Chapter 9 of the PM ISA and
26 Annex E of the NO_x/SO_x ISA summarize the results of a number of studies on the corrosion of
27 metals (US EPA, 2009a; US EPA, 2008c). Moisture is the single greatest factor promoting metal
28 corrosion, however, deposited PM can have additive, antagonistic or synergistic effects. In
29 general, SO₂ is more corrosive than NO_x although mixtures of NO_x, SO₂ and other particulate
30 matter corrode some metals at a faster rate than either pollutant alone (US EPA, 2008c, Annex
31 E.5.2). Information from both the PM ISA and NO_x/SO_x ISA suggest that the extent of damage
32 to metals due to ambient PM is variable and dependent upon the type of metal, prevailing
33 environmental conditions, rate of natural weathering and presence or absence of other pollutants.

34 The PM ISA and NO_x/SO_x ISA summarize the results of a number of studies on PM and
35 stone surfaces. While it is clear from the available information that gaseous air pollutants, in

1 particular SO₂, will promote the deterioration of some types of stones under specific conditions,
2 carbonaceous particles (non-carbonate carbon) and particles containing metal oxides may help to
3 promote the decay process. Studies on metal and stone summarized in the ISA do not show an
4 association between particle size, chemical composition and frequency of repair.

5 A limited number of new studies available on materials damage effects of PM since the
6 last review consider the relationship between pollutants and biodeterioration of structures
7 associated with microbial communities that colonize monuments and buildings (US EPA 2009a,
8 section 9.5). Presence of air pollutants may synergistically enhance microbial deterioration
9 processes. The role of heterotrophic bacteria, fungi and cyanobacteria in biodeterioration varied
10 by local meteorological conditions and pollutant components. In a comparative study of
11 biodeterioration processes on monuments in Latin America, limestone deterioration at the Mayan
12 site of Uxmal was enhanced by biosolubilization by metabolic acids from bacteria and fungi
13 while destruction of the Cathedral of La Plata was attributed primarily to atmospheric pollutants
14 (Herrera and Videla, 2004).

15 PM deposition onto surfaces such as metal, glass, stone and paint can lead to soiling.
16 Soiling results when PM accumulates on an object and alters the optical characteristics
17 (appearance). The reflectivity of a surface may be changed or presence of particulates may alter
18 light transmission. These effects can impact the aesthetic value of a structure or result in
19 reversible or irreversible damage to statues, artwork and architecturally or culturally significant
20 buildings. Due to soiling of building surfaces by PM, the frequency and duration of cleaning
21 may be increased. Soiling affects the aesthetic appeal of painted surfaces. In addition to natural
22 factors, exposure to PM may give painted surfaces a dirty appearance. Pigments in works of art
23 can be degraded or discolored by atmospheric pollutants, especially sulfates (US EPA, 2008c,
24 Annex E-15).

25 Formation of black crusts due to carbonaceous compounds and buildup of microbial
26 biofilms results in discoloration of surfaces. Black crust includes a carbonate component derived
27 from building material and organic carbon (OC) and elemental carbon (EC). In limited new
28 studies quantifying the OC and EC contribution to soiling by black crust, OC predominated over
29 EC at almost all locations (Bonazza et al., 2005). Limited new studies suggest that traffic is the
30 major source of carbon associated with black crust formation (Putaud et al., 2004) and that
31 soiling of structures in Oxford, UK showed a relationship with traffic and NO₂ concentrations
32 (Viles and Gorbushina, 2003). These findings attempt to link atmospheric concentrations of PM
33 to observed damage. However, no data on rates of damage are available and all studies were
34 conducted outside of the U.S.

1 **5.4.3 Staff Conclusions**

2 Available evidence in regards to materials damage and soiling supports the following
3 observations:

- 4 • Materials damage and soiling that occur through natural weathering processes are
5 enhanced by exposure to atmospheric pollutants, most notably SO₂ and particulate
6 sulfates.
- 7 • While ambient particles play a role in the corrosion of metals and in the weathering of
8 materials, no quantitative relationships between ambient particle concentrations and
9 rates of damage have been established.
- 10 • While soiling associated with fine and course particles can result in increased cleaning
11 frequency and repainting of surfaces, no quantitative relationships between particle
12 characteristics and the frequency of cleaning or repainting have been established.
- 13 • Limited new data on the role of microbial colonizers in biodeterioration processes and
14 contributions of black crust to soiling are not sufficient for quantitative analysis.
- 15 • While several studies in the PM ISA and NO_x/SO_x ISA suggest that particles can
16 promote corrosion of metals there remains insufficient evidence to relate corrosive
17 effects to specific particulate levels or to establish a quantitative relationship between
18 ambient PM and metal degradation. With respect to damage to calcareous stone,
19 numerous studies suggest that wet or dry deposition of particles and dry deposition of
20 gypsum particles can enhance natural weathering processes.

21 Revisiting the overarching policy question as to whether the available scientific evidence
22 supports or calls into question the adequacy of the protection for materials afforded by the
23 current suite of secondary PM standards, we conclude that no new evidence in this review calls
24 into question the adequacy of the protection for materials afforded by the current standard. PM
25 effects on materials can play no quantitative role in considering whether any revisions of the
26 secondary PM NAAQS are appropriate at this time. Nonetheless, in the absence of information
27 that provides a basis for establishing a different level of control, observations continue to support
28 retaining an appropriate degree of control on both fine and coarse particles to help address
29 materials damage and soiling associated with PM.

30 **5.4.4 Key Uncertainties and Areas for Future Research and Data Collection**

31 Quantitative relationships are needed between particle size, concentration, chemical
32 concentrations and frequency of repainting and repair. Deposition rates of airborne PM to
33 surfaces would provide an indication of rate and degree of damage to surfaces. There is
34 considerable uncertainty with regard to interaction of co-pollutants in regards to materials
35 damage and soiling processes.

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APPENDICES

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Appendix 2A

Table 2A-1. Predicted Percent of Counties with Monitors (and percent of Population in counties with monitors) Not Likely to Meet Alternative Annual and 24-hour PM _{2.5} Standards											
Region >		All U.S.	Northeast	Southeast	Industrial Midwest	Upper Midwest	Southwest	Northwest	Southern California	Outlying areas	
Total # of counties >		526	87	150	134	44	21	63	16	12	
Total population (x 1,000)>		182,075	42,841	40,904	38,646	7,361	9,311	18,446	22,610	2,099	
Current levels											
annual µg/m ³	24-hour µg/m ³	Statistic		Numbers of counties, populations, and percentages of total							
15	35	# counties	62	9	6	15	0	4	19	9	1
		population	47,896	5,639	5,157	8,551	0	1,040	9,847	17,721	83
		% # counties	12%	10%	4%	11%	0%	19%	30%	56%	8%
		% population	26%	13%	13%	22%	0%	11%	53%	78%	4%
Alternative levels											
annual µg/m ³	24-hour µg/m ³	Statistic		Numbers of counties, populations, and percentages of total							
13	35	# counties	151	19	45	53	0	4	20	10	1
		population	77,390	10,202	13,678	22,128	0	1,040	9,866	20,535	83
		% # counties	29%	22%	30%	40%	0%	19%	32%	63%	8%
		% population	43%	24%	33%	57%	0%	11%	53%	91%	4%
13	30	# counties	211	40	49	72	0	4	34	11	2
		population	104,539	26,228	14,063	28,417	0	1,040	14,266	20,553	114
		% # counties	40%	46%	33%	54%	0%	19%	54%	69%	17%
		% population	57%	61%	34%	74%	0%	11%	77%	91%	5%
12	35	# counties	239	36	79	90	0	4	20	10	1
		population	102,274	21,725	18,913	30,255	0	1,040	9,866	20,535	83
		% # counties	45%	41%	53%	67%	0%	19%	32%	63%	8%
		% population	56%	51%	46%	78%	0%	11%	53%	91%	4%
12	30	# counties	270	46	79	95	0	4	34	11	2
		population	114,610	27,991	18,913	31,876	0	1,040	14,266	20,553	114
		% # counties	51%	53%	53%	71%	0%	19%	54%	69%	17%
		% population	63%	65%	46%	82%	0%	11%	77%	91%	5%
11	30	# counties	343	53	115	119	5	5	34	11	2
		population	132,785	30,053	26,400	36,179	1,250	4,112	14,266	20,553	114
		% # counties	65%	61%	77%	89%	11%	24%	54%	69%	17%
		% population	73%	70%	65%	94%	17%	44%	77%	91%	5%
11	25	# counties	393	71	116	130	11	7	44	12	3
		population	147,943	39,345	26,547	38,494	2,164	4,317	15,739	21,306	173
		% # counties	75%	82%	77%	97%	25%	33%	70%	75%	25%
		% population	81%	92%	65%	100%	29%	46%	85%	94%	8%
10	30	# counties	391	62	133	128	14	7	34	12	2
		population	146,671	35,932	29,392	38,335	2,611	4,856	14,266	21,306	114
		% # counties	74%	71%	89%	96%	32%	33%	54%	75%	17%
		% population	81%	84%	72%	99%	35%	52%	77%	94%	5%
10	25	# counties	417	71	133	130	17	8	44	12	3
		population	151,975	39,345	29,392	38,494	2,782	4,887	15,739	21,306	173
		% # counties	79%	82%	89%	97%	39%	38%	70%	75%	25%
		% population	83%	92%	72%	100%	38%	52%	85%	94%	8%

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Appendix 3A

Table 3A-1. Predicted Percent of Counties with Monitors (and percent of Population in counties with monitors) Not Likely to Meet Alternative 24-hour PM ₁₀ Standards										
Region >	All U.S.	Northeast	Southeast	Industrial Midwest	Upper Midwest	Southwest	Northwest	California	Outlying areas	
Total # of counties >	318	40	60	57	35	24	79	18	8	
Total population >	122,582	15,753	27,009	23,646	5,143	11,037	16,188	22,695	1,525	
Current 3-Year Expected Exceedance Equivalent Design Value										
µg/m ³	Statistic	Numbers of counties, populations, and percentages of total								
150	# counties	45	0	3	1	1	14	15	11	2
	population	32,169	0	4,159	348	28	7,675	2,075	17,724	320
	% # counties	14%	0%	5%	2%	3%	58%	19%	61%	25%
	% population	26%	0%	15%	1%	1%	70%	13%	78%	21%
Alternative 24-hour level (3-year average 98th percentile)										
µg/m ³	Statistic	Numbers of counties, populations, and percentages of total								
87	# counties	38	0	3	3	3	12	8	10	1
	population	22,450	0	4,159	1,789	710	6,106	1,366	8,220	260
	% # counties	12%	0%	5%	5%	9%	50%	10%	56%	13%
	% population	18%	0%	15%	8%	14%	55%	8%	36%	17%
85	# counties	41	0	3	3	4	12	9	11	1
	population	22,736	0	4,159	1,789	734	6,106	1,381	8,467	260
	% # counties	13%	0%	5%	5%	11%	50%	11%	61%	13%
	% population	19%	0%	15%	8%	14%	55%	9%	37%	17%
80	# counties	48	0	4	4	5	12	11	12	2
	population	25,914	0	4,176	1,838	781	6,106	1,499	11,313	360
	% # counties	15%	0%	7%	7%	14%	50%	14%	67%	25%
	% population	21%	0%	15%	8%	15%	55%	9%	50%	24%
75	# counties	60	1	6	4	6	12	18	13	2
	population	37,835	74	4,441	1,838	885	6,106	3,458	20,832	360
	% # counties	19%	3%	10%	7%	17%	50%	23%	72%	25%
	% population	31%	0%	16%	8%	17%	55%	21%	92%	24%
70	# counties	75	1	7	7	8	15	22	13	4
	population	47,280	74	4,462	9,094	986	7,745	3,798	20,832	450
	% # counties	24%	3%	12%	12%	23%	63%	28%	72%	50%
	% population	39%	0%	17%	38%	19%	70%	23%	92%	29%
65	# counties	100	4	8	12	9	15	35	14	5
	population	54,170	666	4,776	12,527	1,003	7,745	5,538	21,585	490
	% # counties	31%	10%	13%	21%	26%	63%	44%	78%	63%
	% population	44%	4%	18%	53%	20%	70%	34%	95%	32%

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Chapter 4 Appendices

Appendix 4A: Information Regarding the 1-Hour PM_{2.5} Mass Indicator

Appendix 4B: Simplified Approaches to Calculate Hourly PM Light Extinction Values from Hourly PM_{2.5} Mass and Relative Humidity Data plus 24-hour Mean PM Composition Data

Appendix 4C: Assessment of the PM Components Responsible for the Largest Hourly PM Light Extinction and PM_{2.5} Mass Hours Selected as Maximum Daily and using All Hours for 14 Urban Areas

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Appendix 4A

Information Regarding the 1-hour PM_{2.5} Mass Indicator

This Appendix presents information on 2005-2007 levels of 1-hour PM_{2.5} mass concentrations in the 14 urban study areas and on the “what if” PM₁₀ light extinction conditions that would exist if the study areas met each of 10 alternative secondary PM NAAQS scenarios based on a 1-hour PM_{2.5} mass indicator. With respect to the latter subject, this Appendix is therefore similar to Chapter 4 of the Particulate Matter Urban-Focused Visibility Assessment (UFVA), which presented similar information for 18 secondary PM NAAQS scenarios based on PM₁₀ light extinction as the indicator, for the current annual and 24-hour PM_{2.5} NAAQS, and for a scenario with an annual NAAQS of 12 µg/m³ and a 24-hour NAAQS of 25 µg/m³.

1.0 Indicator and Monitoring Method

As in Chapter 4 of the UFVA, this Appendix excludes from all NAAQS scenarios and results all non-daylight hours and all daylight hours with relative humidity greater than 90 percent. This applies to both the definition of 10 secondary NAAQS scenarios, and to graphics and tables that characterize ambient conditions. While ambient humidity should not affect conventional measurement approaches for 1-hour PM_{2.5} mass, the issue of co-occurrence of high humidity levels with light extinction due to natural conditions would still apply. See section 3.3.5 of the UFVA. The assumed hours of daylight are the same as those used in the UFVA, as shown in Table 3-5 of the UFVA.

All values for 1-hour PM_{2.5} mass concentration in this appendix come from the continuous instruments at the 14 urban study sites, with no adjustment to make these values consistent with the collocated 24-hour FRM measurement of PM_{2.5} mass. Appendix A of the UFVA provides details on the type of continuous instrument at each study site. TEOMs were used at all sites except for beta attenuation instruments in Fresno and Philadelphia, nephelometer instruments in Tacoma and Phoenix, and an FDMS instrument in Salt Lake City.

For conciseness in this second external review draft, only the daily maximum daylight 1-hour PM_{2.5} mass concentration indicator is considered in this Appendix. It would also be possible to construct alternative NAAQS scenarios of an all-hours type, which could be analyzed in the same manner as presented in this Appendix.

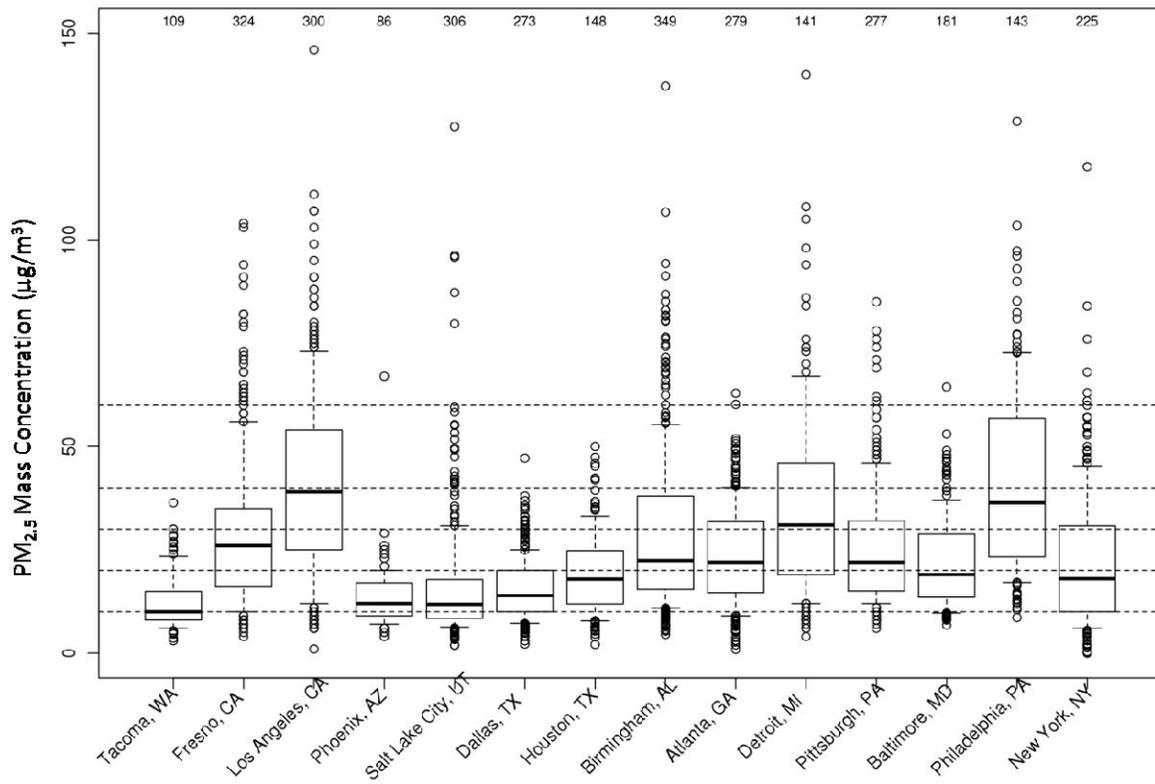
2.0 Current Conditions of 1-hour PM_{2.5} Mass

Figure 4A-1 is a box plot of 2005-2007 daily maximum daylight 1-hour PM_{2.5} mass concentrations for the 14 study areas, excluding hours with relative humidity greater than 90 percent, to give a sense of the range and central tendency of this parameter. The horizontal reference lines are at 10, 20, 30, 40 and 60 µg/m³. The relative positions of the 90 percentile concentrations (indicated by the horizontal stroke at the top of the whisker) are generally consistent with the relative ranking of these sites according to their design values for the 24-hour PM_{2.5} NAAQS (see Table 3-2 of the UFVA); similarly, the relative positions of the median

1 concentrations are generally consistent with the annual PM_{2.5} design values. Table 4A-1, based
 2 on the same data as Figure 4A-1, presents the percentage of days in 2005-2007 on which the
 3 daily maximum daylight 1-hour PM_{2.5} concentration exceeded the reference levels represented
 4 by the horizontal lines in Figure 4A-1.

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Figure 4A-1. 2005-2007 daily maximum daylight 1-hour PM_{2.5} mass concentrations (µg/m³) for the 14 study areas (excluding hours with relative humidity greater than 90 percent)



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1 **Table 4A-1. Percentage of days with daily maximum daylight 1-hour PM_{2.5} mass**
 2 **concentration exceeding reference levels in 2005-2007 (excluding hours with relative**
 3 **humidity greater than 90 percent)**
 4

Study Area	Number of Days with Estimates	1-hour PM _{2.5} Mass Reference Level (µg/m ³)				
		10	20	30	40	60
Tacoma	109	50	11	1	0	0
Fresno	324	88	62	37	20	8
Los Angeles	300	92	81	67	46	20
Phoenix	86	60	8	1	1	1
Salt Lake City	306	64	20	11	7	2
Dallas	273	75	25	5	0	0
Houston	148	80	42	14	5	0
Birmingham	349	92	60	37	23	8
Atlanta	279	86	56	28	10	1
Detroit	141	92	72	52	36	13
Pittsburgh	277	94	57	28	15	3
Baltimore	181	90	46	22	8	1
Philadelphia	143	99	84	63	45	20
New York	225	75	43	25	13	3

5
 6 **3.0 Alternative NAAQS Scenarios Based on 1-hour PM_{2.5} Mass as the Indicator**
 7

8 To ensure examination of a wide enough range of alternative standards based on 1-hour
 9 PM_{2.5} mass to encompass the range of standards that might be considered as alternatives to the
 10 PM₁₀ light extinction NAAQS scenarios examined in Chapter 4 of the UFVA, we considered
 11 levels of 10, 20, 30, 40, and 60 µg/m³. Only the daily maximum daylight hour form was
 12 considered. Each level was combined with two statistical forms: the three-year average of the
 13 annual 90th percentile value and the three-year average of the annual 95th percentile value. For
 14 ease of reference, these scenarios are designated by letters from “aa” to “jj” and listed in Table
 15 4A-2. Looking somewhat ahead to results presented below, the scenarios are arranged in Table
 16 4A-2 in order of least to most stringent in terms of the reductions in ambient PM_{2.5} needed from
 17 current levels to meet the current and alternative NAAQS levels and forms.
 18

19 **Table 4A-2. Alternative NAAQS scenarios based on daily maximum daylight 1-hour PM_{2.5}**
 20 **mass, averaged over three years (excluding hours with relative humidity greater than 90**
 21 **percent)**
 22

NAAQS Scenario	Level (µg/m ³)	Statistical Form
aa	60	3-year average of 90 th percentile
bb	60	3-year average of 95 th percentile
cc	40	3-year average of 90 th percentile
dd	40	3-year average of 95 th percentile
ee	30	3-year average of 90 th percentile
ff	30	3-year average of 95 th percentile
gg	20	3-year average of 90 th percentile
hh	20	3-year average of 95 th percentile
ii	10	3-year average of 90 th percentile
jj	10	3-year average of 95 th percentile

1 **4.0 Approach to Modeling “What If” Conditions of PM₁₀ Light Extinction for**
2 **Alternative Secondary NAAQS Based on 1-hour PM_{2.5} Mass**
3

4 Before modeling “what if” conditions, we augmented the data set described in Table 4 of
5 the UFVA in the same manner as described in Section 4.1.4 of the UFVA, to achieve seasonal
6 balance despite the lack of monitoring data for one quarter in each of Houston and Phoenix. In
7 Tacoma and Phoenix, which had data only for two years in the 2005-2007 period, we averaged
8 the percentile values from the only two available years rather than the three years defined for the
9 statistical form of the NAAQS scenarios.

10
11 The modeling of daily maximum daylight 1-hour PM_{2.5} mass under each of the scenarios
12 listed in Table 4A-2 used a rollback approach that combined relevant concepts and steps from
13 the rollback methods described in sections 4.1.4 (for PM₁₀ light extinction scenarios) and 4.2.2
14 (for scenarios based on annual average and 24-hour average PM_{2.5}) of the UFVA. The following
15 are the steps in the modeling.

16
17 1. Identify the 90th percentile daily maximum daylight 1-hour PM_{2.5} mass value in each of
18 2005, 2006, and 2007 for a study area. Average these to determine the 3-year average design
19 value for that percentile form. Repeat for the 95th percentile form. These design values are
20 presented in Table 4A-3. They range from 22 to 81 µg/m³, indicating that some study areas meet
21 some of the NAAQS scenarios under current conditions. In such cases, PM_{2.5} concentrations
22 were not adjusted, i.e., there was no “roll up” for any area in any scenario.

23
24 2. Using the same days and hours as contributed by the three annual 90th percentile values
25 for actual 1-hour PM_{2.5} mass, find the three corresponding values of policy relevant background
26 (PRB) 1-hour PM_{2.5} mass. Average these three annual values of PRB 1-hour PM_{2.5} to obtain the
27 3 year average PRB portion of the actual 1-hour PM_{2.5} design value for the 90th percentile form.
28 Repeat for the 95th percentile form.

29
30 In the modeling for the NAAQS scenarios examined in the UFVA, PRB for 1-hour PM_{2.5}
31 mass was not explicitly calculated because it was not needed in the rollback modeling for the
32 scenarios addressed in the UFVA. Therefore, it was necessary to reconstruct this parameter by
33 adding the values for the PRB concentrations of the five components of PM_{2.5}: nitrate, sulfate,
34 elemental carbon, organic carbon material, and soil. The method for estimating PRB for these
35 five components is described in Appendix C of the UFVA.

36
37 3. Subtract the value from step 2 from the value from step 1, to determine the non-PRB
38 portion of the 1-hour PM_{2.5} mass design value.

39
40 4. Calculate the percentage reduction required in non-PRB 1-hour PM_{2.5} mass in order to
41 reduce the design value to the level that defines the NAAQS scenario, using the following
42 equation:

43
44
$$\text{Percent reduction required} = 1 - (\text{NAAQS level} - \text{PRB portion of the design value}) / (\text{non-PRB}$$

45
$$\text{portion of the design value})$$

46

1 The percentage reductions determined in step 4 are shown in Table 4A-4. Note that for
2 some combinations of area and scenario no reduction is required because the 2005-2007 design
3 value already meets the NAAQS scenario.

4
5 5. Turning to the entire set of day/hour-specific actual and PRB daylight 1-hour
6 concentrations of the five PM_{2.5} components for the three (or two) year period, determine the
7 non-PRB portion of each of the five components in an hour by subtracting the PRB value from
8 actual value, reduce it by the percentage determined in step 4, and add back in the PRB 1-hour
9 concentration of the component.

10
11 6. Finally, calculate PM₁₀ light extinction using the reduced values of the five components,
12 the original value of 1-hour PM_{10-2.5}, and the 1-hour value of f(RH), according to the following
13 equation for PM₁₀ light extinction (see section 3.2.3 of the UFVA for an explanation of the
14 variables in this equation).

$$\begin{aligned} & b_{\text{extPM}} = 3 \times f(\text{RH}) \times [\text{Sulfate}] \\ & + 3 \times f(\text{RH}) \times [\text{Nitrate}] \\ & + 4 \times [\text{Organic Mass}] \\ & + 10 \times [\text{Elemental Carbon}] \\ & + 1 \times [\text{Fine Soil}] \\ & + 0.6 \times [\text{Coarse Mass}] \end{aligned}$$

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23 These steps assume that in order to meet a PM NAAQS scenario based on 1-hour PM_{2.5}
24 as the indicator, each component of PM_{2.5} is reduced by an equal percentage, across the five
25 components and across all hours. In actual implementation of such a NAAQS, each state would
26 develop an attainment strategy, which might result in unequal percentage reductions of the
27 components. If the strategy emphasized reductions in the fine soil component, for example, PM
28 light extinction levels would remain high relative to those estimated by these steps, because fine
29 soil is not efficient in terms of reducing visibility compared to the other four components on a
30 dry mass-to-mass basis. On the other hand, a strategy that involves relatively large reductions in
31 sulfate or nitrate would achieve greater reductions in PM light extinction than estimated by these
32 steps. The uncertainty in how the results of this rollback method compare to the results of actual
33 attainment strategies should be kept in mind when comparing the results of “what if” scenarios
34 for NAAQS based on PM_{2.5} mass as the indicator versus scenarios based on PM₁₀ light
35 extinction. Unlike the effect of humidity variation between areas, this source of uncertainty is
36 not reflected in any of the results presented in this Appendix and will not be apparent in
37 comparisons of results in this Appendix to results presented in the UFVA for NAAQS scenarios
38 based on PM₁₀ light extinction.

39
40 These steps also assume no change in PM_{10-2.5} concentrations between current conditions
41 and “what if” conditions. While reductions in PM_{10-2.5} would not be needed to meet a secondary
42 NAAQS based on 1-hour PM_{2.5} mass, it is possible that strategies to control PM_{2.5} concentrations
43 might also achieve reductions in PM_{10-2.5} concentrations because some sources emit both and
44 some control methods achieve some reductions in both. However, in most of the 14 study areas,
45 PM_{10-2.5} makes a small contribution to estimated PM₁₀ light extinction, in part because in many
46 of the areas no local data on PM_{10-2.5} concentrations were available and the method used to fill

1 this gap (application of a factor to PM_{2.5} concentration) simply could not produce a high estimate
 2 of PM_{10-2.5}.

3
 4 **Table 4A-3. 2005-2007 design values for 1-hour PM_{2.5} mass (µg/m³)**
 5

Study Area	Percentile Form	
	90th	95th
Tacoma	22	27
Fresno	55	66
Los Angeles	72	81
Phoenix	20	24
Salt Lake City	32	45
Dallas	26	29
Houston	33	37
Birmingham	55	74
Atlanta	40	45
Detroit	64	79
Pittsburgh	46	51
Baltimore	37	43
Philadelphia	67	77
New York	44	55

6
 7
 8 **Table 4A-4. Percentage reductions in non-PRB PM_{2.5} components required to meet**
 9 **NAAQS scenarios based on 1-hour PM_{2.5} mass**
 10

Scenario	aa	bb	cc	dd	ee	ff	gg	hh	ii	jj
Level (µg/m ³)	60	60	40	40	30	30	20	20	10	10
Percentile Form	90	95	90	95	90	95	90	95	90	95
Study Area	Percentage Reduction									
Tacoma	0	0	0	0	0	0	11	27	60	69
Fresno	0	10	28	40	46	55	65	71	83	86
Los Angeles	17	26	45	51	59	64	73	76	87	88
Phoenix	0	0	0	0	0	0	0	15	51	58
Salt Lake City	0	0	0	12	7	34	39	56	70	78
Dallas	0	0	0	0	0	0	23	34	64	69
Houston	0	0	0	0	9	20	40	49	71	78
Birmingham	0	19	28	46	46	60	65	74	84	87
Atlanta	0	0	0	12	25	34	51	57	77	80
Detroit	7	24	38	50	54	63	70	75	85	88
Pittsburgh	0	0	13	22	35	42	57	62	79	81
Baltimore	0	0	0	8	19	31	47	55	74	78
Philadelphia	10	22	40	49	55	62	71	75	86	88
New York	0	0	8	28	32	46	55	65	78	83

11
 12

1 **5.0. 1-hour PM_{2.5} Mass Results for “Just Meeting” Alternative Secondary NAAQS**
2 **Scenarios Based on 1-hour PM_{2.5} Mass**
3

4 As a check on the reasonableness of the rollback method described in section 4.0 and on
5 the accuracy of the code used to implement it, it is of interest to examine the distribution of the
6 levels of 1-hour PM_{2.5} that result from the method. Ideally, after rollback any area that had a
7 non-zero required reduction should have a post-rollback design value for 1-hour PM_{2.5} mass that
8 is exactly equal to the target design value. Also, there should be a progression of reductions in 1-
9 hour PM_{2.5} medians and other percentile points on the distribution as progressively more
10 stringent scenarios are modeled.
11

12 Table 4A-5 shows the post-rollback 1-hour PM_{2.5} mass design values for the scenarios,
13 with percentile forms matched. Design values for area-scenario combinations for which the
14 required reductions were zero have been omitted, because the current conditions design values
15 for these combinations would not be expected to reflect the target design value. It can be seen
16 that the design values progress as expected and are in the vicinity of the target design values, but
17 are not always exactly equal to the targets. EPA staff attributes this to the fact that PRB
18 concentrations of 1-hour PM_{2.5} mass vary from hour to hour. It is possible for the daily
19 maximum PM_{2.5} mass concentration on a certain day in 2005 with a percentile rank of, for
20 example, 96th to have a relatively small PRB portion and a large non-PRB portion compared to
21 the daily maximum concentration that ranks 95th. When an equal reduction is made to the non-
22 PRB portion of each total concentration, the two values may switch rank positions, and so a new
23 day and hour becomes the 2005 contributor to the rolled back three-year design value. Since this
24 day and hour was not used to determine the required percentage reduction, the resulting design
25 value will not exactly meet the target design value. It would be possible to iterate with higher
26 and lower percentage reductions until the rolled back design value exactly matched the target
27 design value, but EPA considered this degree of refinement to be unnecessary in order to meet
28 the objectives of the Policy Assessment Document, given other uncertainties in the underlying
29 data and in the assumptions used to estimate PM₁₀ light extinction values.
30

31 EPA staff also generated and examined box plots of daily maximum daylight 1-hour
32 PM_{2.5} mass concentrations as a check for conceptual or programming errors, and found them to
33 match expectations. They are not included here, for conciseness.

Table 4A-5. Post-rollback design values for daily maximum 1-hour PM_{2.5} mass. Design values are shown only for combinations of study area and scenario for which the study area does not meet the scenario under current conditions, such that reductions were made during the rollback modeling.

Scenario	aa	bb	cc	dd	ee	ff	gg	hh	ii	jj
Level (µg/m ³)	60	60	40	40	30	30	20	20	10	10
Statistical Form	90th	95th	90th	95th	90th	95th	90th	95th	90th	95th
Study Area	Corresponding Design Value (µg/m ³) (same percentile form as the scenario)									
Tacoma							20	21	11	12
Fresno		63	40	42	30	31	20	21	10	10
Los Angeles	53	53	35	35	26	26	18	18	9	9
Phoenix								19	10	10
Salt Lake City				38	29	28	19	19	10	10
Dallas							23	23	12	11
Houston					29	27	19	18	10	9
Birmingham		58	42	39	32	29	21	20	11	10
Atlanta				36	28	27	19	18	10	10
Detroit	52	59	34	39	26	29	17	20	9	10
Pittsburgh			33	33	24	25	16	17	8	9
Baltimore				38	31	28	21	19	10	10
Philadelphia	46	44	31	30	23	22	16	15	8	8
New York			42	40	32	30	21	20	11	10

6.0 PM₁₀ Light Extinction Results for “Just Meeting” Alternative Secondary NAAQS Scenarios Based on 1-hour PM_{2.5} Mass

The rollback steps described in section 4.0 resulted in estimates of PM₁₀ light extinction for each day and hour in each study area, for each of the 10 NAAQS scenarios based on 1-hour PM_{2.5} mass as the indicator. Two summaries of these conditions are presented here.

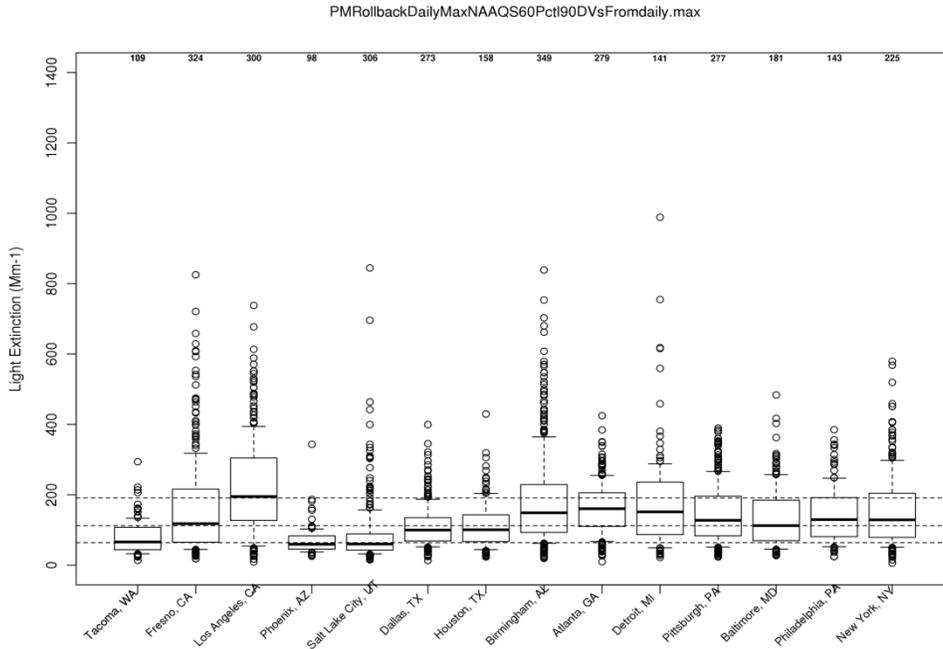
Figure 4A-2 presents a box plot of daily maximum daylight 1-hour PM₁₀ light extinction for each NAAQS scenario based on 1-hour PM_{2.5} mass. These can be compared to Figure 3-8(a) of the UFVA representing pre-rollback daily maximum PM₁₀ light extinction, and to the upper panel of the figures in Appendix F of the UFVA representing the daily maximum PM₁₀ light extinction levels resulting from the 20 NAAQS scenarios examined in the UFVA (18 scenarios based on PM₁₀ light extinction as the indicator, the current annual and 24-hour PM_{2.5} NAAQS, and a scenario with an annual NAAQS of 12 µg/m³ and a 24-hour NAAQS of 25 µg/m³). It can be seen that the distribution of PM_{2.5} mass in a given study area shifts downward as the NAAQS scenarios progress from least to most stringent (as indicated by the required percentage reduction) and in most cases become more similar to other areas (once the progression of scenarios begins to require reductions in a given area).

Table 4A-6 presents the percentage of days in 2005-2007 on which daily maximum 1-hour PM₁₀ light extinction exceeded each of the CPLs, under each of the 10 secondary PM NAAQS scenarios based on 1-hour PM_{2.5} mass. These percentages are necessarily based on the days for which data to estimate PM₁₀ light extinction were available, but are best estimates of the percentage of all days in the year given that the days with data were well distributed across the

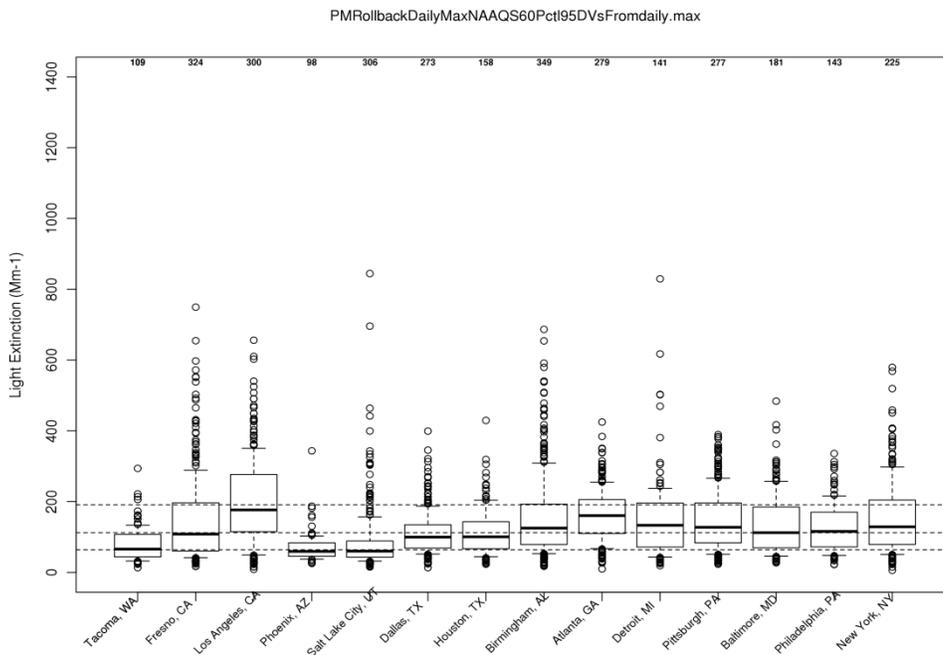
1 year on either a one-in-three or one-in-six sampling schedule. These percentages can be
2 compared to the same-basis percentages presented in Table 4-7 of the UFVA.

1 **Figure 4A-2. Distributions of maximum daily daylight 1-hour PM₁₀ light extinction under**
 2 **“just meet” conditions for NAAQS scenarios based on 1-hour PM_{2.5} mass (excluding hours**
 3 **>90% RH)**
 4
 5

(aa) NAAQS Scenario: 60 µg/m³ and 90th percentile



(bb) NAAQS Scenario: 60 µg/m³ and 95th percentile

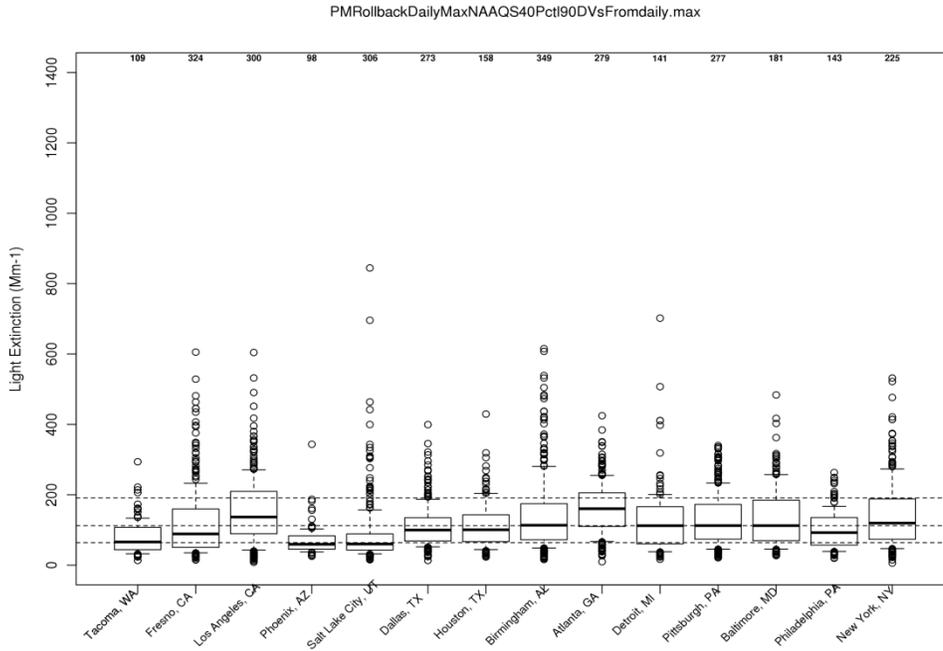


6

8

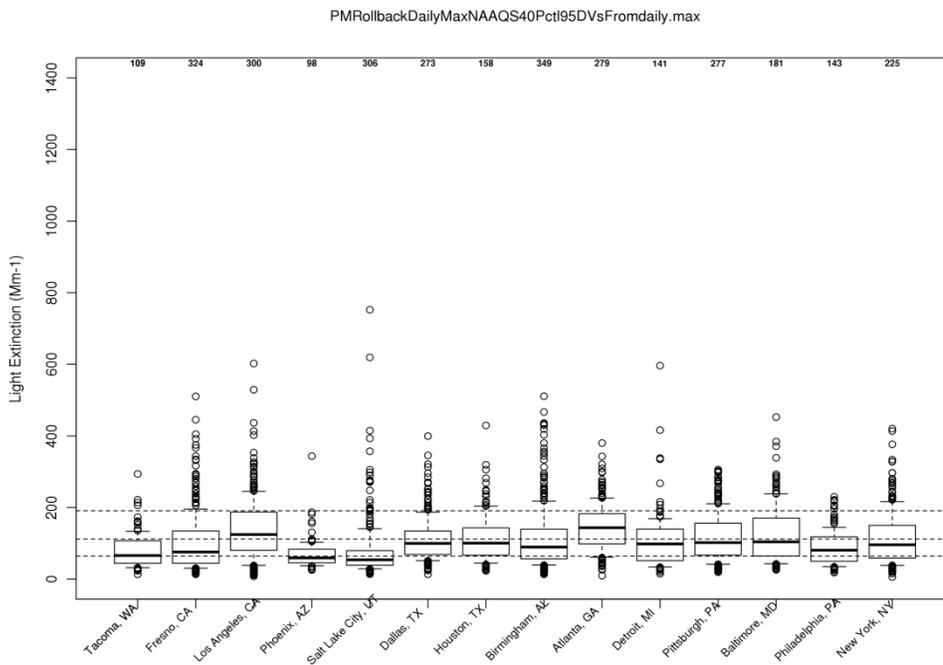
1 **Figure 4A-2 (cont). Distributions of maximum daily daylight 1-hour PM₁₀ light extinction**
 2 **under “just meet” conditions for NAAQS scenarios based on 1-hour PM_{2.5} mass (excluding**
 3 **hours >90% RH) (continued)**
 4
 5

(cc) NAAQS Scenario: 40 µg/m³ and 90th percentile



6

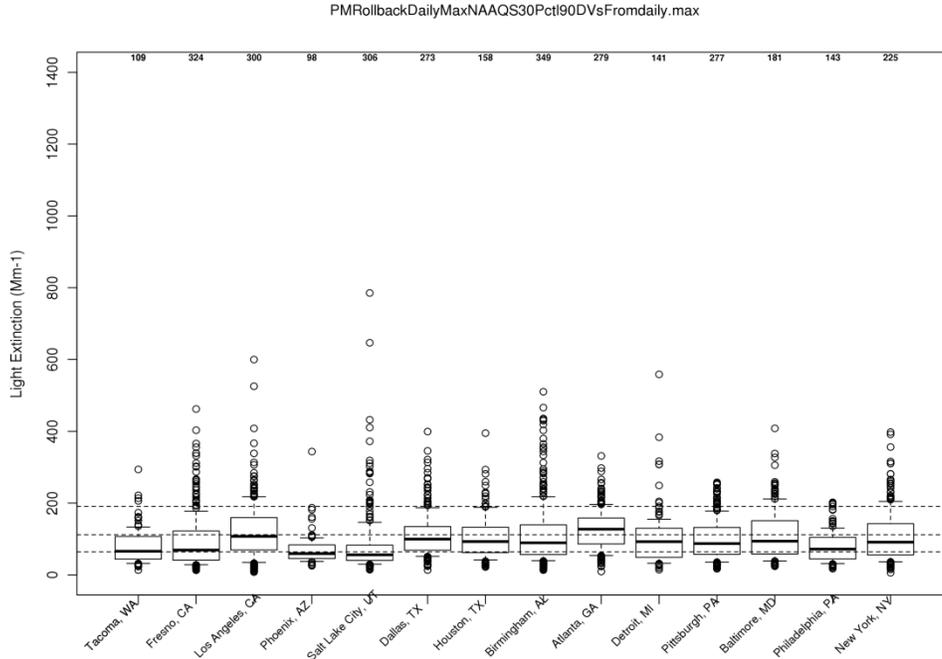
(dd) NAAQS Scenario: 40 µg/m³ and 95th percentile



8

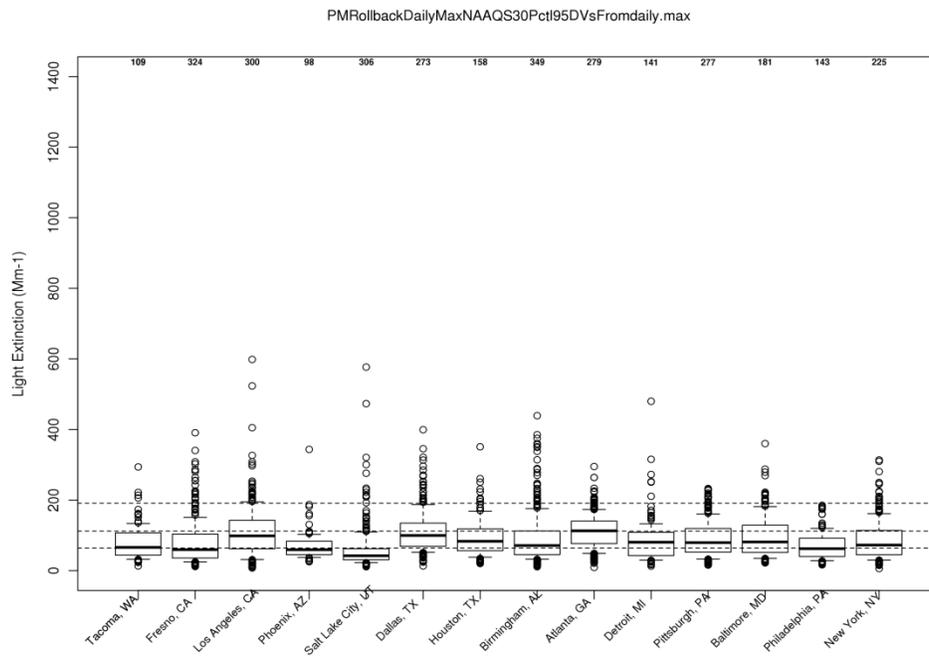
1 **Figure 4A-2 (cont). Distributions of maximum daily daylight 1-hour PM₁₀ light extinction**
 2 **under “just meet” conditions for NAAQS scenarios based on 1-hour PM_{2.5} mass (excluding**
 3 **hours >90% RH) (continued)**
 4
 5

(ee) NAAQS Scenario: 30 µg/m³ and 90th percentile



6

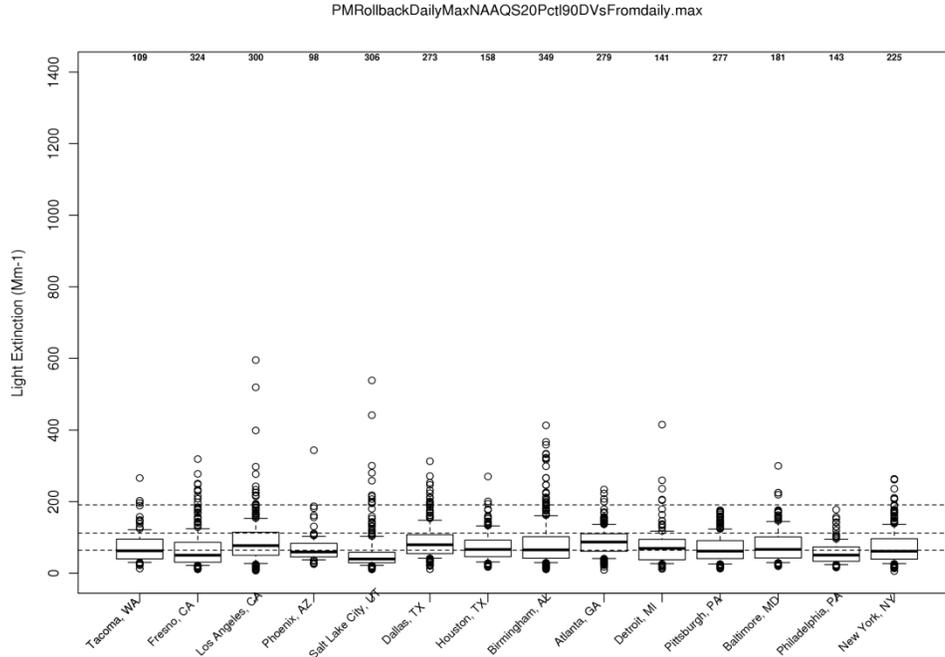
(ff) NAAQS Scenario: 30 µg/m³ and 95th percentile



8

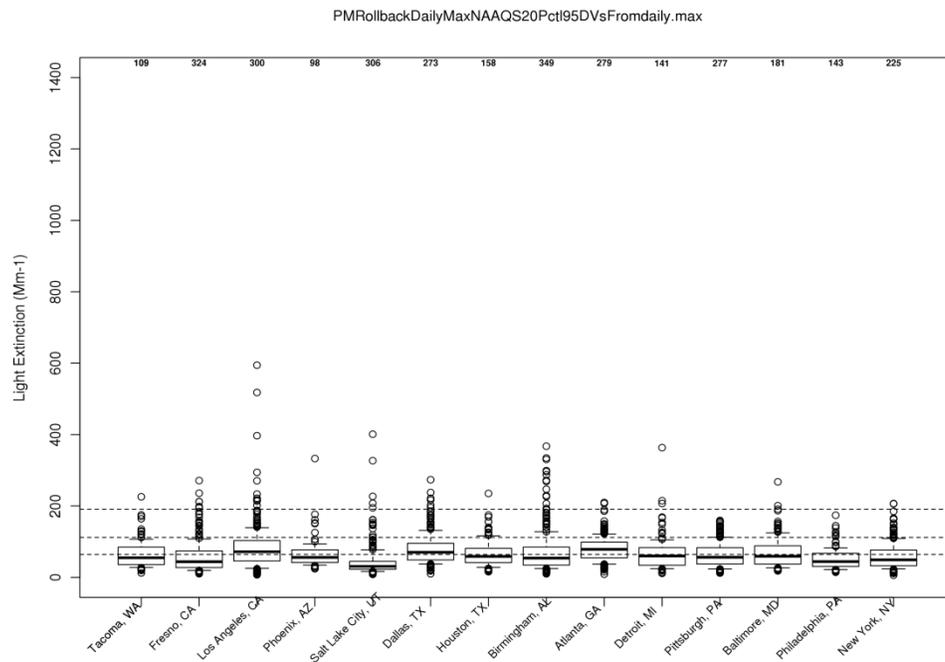
1 **Figure 4A-2 (cont). Distributions of maximum daily daylight 1-hour PM₁₀ light extinction**
 2 **under “just meet” conditions for NAAQS scenarios based on 1-hour PM_{2.5} mass (excluding**
 3 **hours >90% RH) (continued)**
 4
 5

(gg) NAAQS Scenario: 20 µg/m³ and 90th percentile



6
7

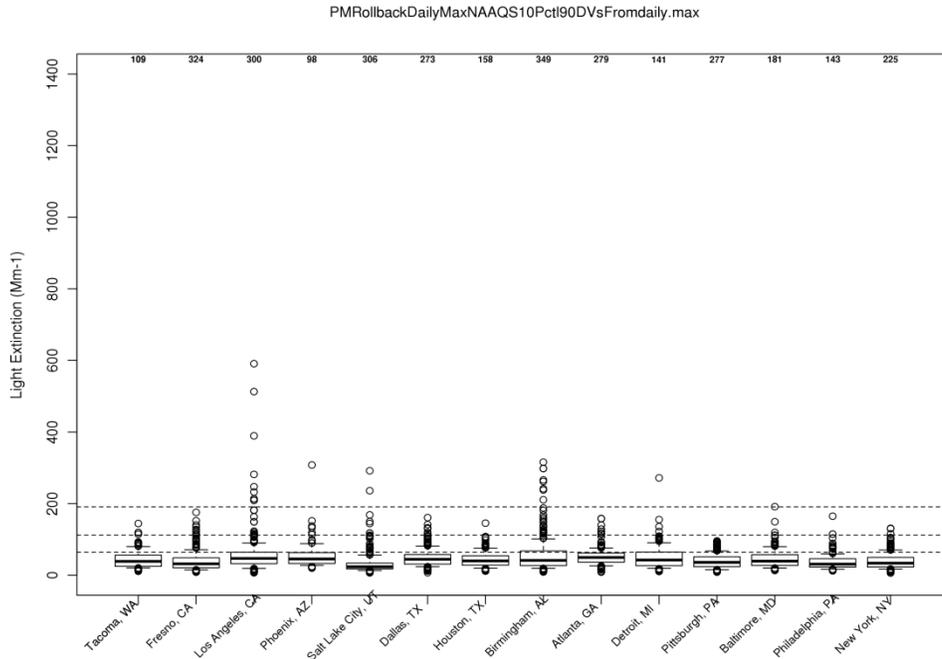
(hh) NAAQS Scenario: 20 µg/m³ and 95th percentile



8

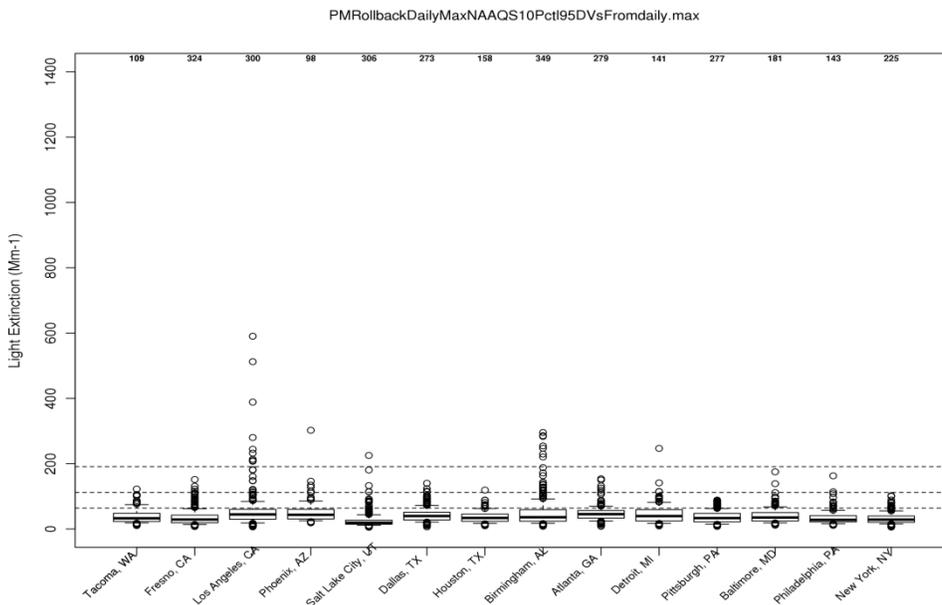
1 **Figure 4A-2 (cont). Distributions of maximum daily daylight 1-hour PM₁₀ light extinction**
 2 **under “just meet” conditions for NAAQS scenarios based on 1-hour PM_{2.5} mass (excluding**
 3 **hours >90% RH) (continued)**
 4
 5

(ii) NAAQS Scenario: Daily Max: 10 µg/m³ and 90th percentile



6
7

(j) NAAQS Scenario: Daily Max: 10 µg/m³ and 95th percentile



8

Table 4A-6. Percentage of days across three years (two years in the case of Phoenix and Houston) with maximum 1-hour daylight PM₁₀ light extinction above CPLs when “just meeting” NAAQS scenarios based on 1-hour PM_{2.5} mass. Blue shading indicates no reduction required from current conditions.

Scenario NAAQS Level (µg/m ³) NAAQS Percentile Form	Days with max hour above 64 Mm ⁻¹										Days with max hour above 112 Mm ⁻¹										Days with max hour above 191 Mm ⁻¹									
	aa	bb	cc	dd	ee	ff	gg	hh	ii	jj	aa	bb	cc	dd	ee	ff	gg	hh	ii	jj	aa	bb	cc	dd	ee	ff	gg	hh	ii	jj
	60	60	40	40	30	30	20	20	10	10	60	60	40	40	30	30	20	20	10	10	60	60	40	40	30	30	20	20	10	10
	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95	90	95
Area	Percentage of days										Percentage of days										Percentage of days									
Tacoma	53	53	53	53	53	53	53	53	43	35	23	23	23	23	23	23	23	23	11	6	4	4	4	4	4	4	4	4	1	1
Fresno	76	73	65	57	69	60	55	44	28	17	52	48	37	31	44	32	29	18	9	4	30	27	17	11	23	12	10	5	1	0
Los Angeles	89	87	84	81	84	79	74	69	41	30	78	76	65	57	65	53	41	31	11	7	52	46	30	24	30	19	11	6	3	3
Phoenix	44	44	44	44	44	44	44	44	37	32	6	6	6	6	6	6	6	6	6	6	1	1	1	1	1	1	1	1	1	1
Salt Lake City	45	45	45	37	45	45	45	26	17	10	17	17	17	15	17	17	17	11	8	5	8	8	8	7	8	8	8	5	2	1
Dallas	81	81	81	81	81	81	81	71	41	29	41	41	41	41	41	41	41	32	8	5	10	10	10	10	10	10	10	7	0	0
Houston	79	79	79	79	79	79	74	65	32	27	44	44	44	44	44	44	35	28	6	3	11	11	11	11	11	11	9	6	1	0
Birmingham	89	85	80	68	87	80	72	62	41	34	65	56	51	36	58	51	40	30	15	12	34	26	21	13	30	20	15	11	4	3
Atlanta	91	91	91	89	91	89	82	77	47	34	75	75	75	68	74	66	51	35	3	3	31	31	31	21	31	19	5	3	0	0
Detroit	84	80	74	72	76	73	65	60	40	33	67	57	51	43	53	48	34	21	9	6	43	28	13	7	14	9	6	4	1	1
Pittsburgh	85	85	81	77	81	77	63	55	27	19	57	57	51	45	52	44	29	22	3	0	26	26	18	14	21	13	6	2	0	0
Baltimore	81	81	81	76	81	74	64	56	31	20	51	51	51	45	51	44	31	23	4	3	23	23	23	18	23	16	8	2	1	1
Philadelphia	84	78	71	62	72	63	55	43	17	10	60	54	33	29	37	31	16	10	3	3	26	17	8	5	8	5	0	0	0	0
New York	83	83	80	71	81	73	63	56	27	19	60	60	56	39	56	40	32	22	6	3	29	29	25	16	25	17	9	5	0	0
Average	76	75	72	68	73	69	64	56	34	25	50	48	43	37	45	38	31	23	7	4	23	21	15	11	17	12	8	4	1	1

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Appendix 4B

Simplified Approaches to Calculate Hourly PM_{2.5} Light Extinction Values from Hourly PM_{2.5} Mass and Relative Humidity Data plus 24-hour Mean PM_{2.5} Composition Data

Overview: The goal of the assessment described in this appendix is to identify a simpler procedure for calculating PM_{2.5} light extinction that produces hourly values that are comparable to those developed for the 15-city¹ assessment (UFVA Section 3.3). The expected benefits of moving to a simpler approach include an increase the numbers of sites at which it could be applied (i.e., by not requiring so many collocated monitors), and periods of time (i.e., by not using sample day-specific CSN data) for which the calculation of PM light extinction could be conducted.

The UFVA approach used the following:

1. SANDWICH mass balance model to estimate organic carbonaceous material and nitrate mass concentrations on the FRM filter for each Chemical Speciation Network (CSN) sample day, and thus also organic carbonaceous material mass on the CSN filter. Other component concentrations on the CSN filter are taken by direct measurement.
2. Monthly mean diurnal PM_{2.5} variations of each of the major PM_{2.5} components from CMAQ air quality simulation modeling results applied to sample day specific CSN 24-hour samples.
3. Hourly FRM-consistent PM_{2.5} mass developed by normalizing continuous PM_{2.5} measurements to the 24-hour FRM filter mass.
4. Hourly sum of the FRM-consistent components scaled to hourly PM_{2.5} mass.
5. An adjustment to the hourly PM_{2.5} nitrate component concentration to reflect actual atmospheric concentration, which is assumed to be represented by the CSN filter nitrate measurement. (This step in effect un-does the FRM nitrate loss introduced by the SANDWICH mass balance model as part of estimating organic carbonaceous material mass.)

¹ The PM_{10-2.5} values for St. Louis were determined by difference between two nearby monitoring sites. Based on comments received on the UFVA to the effect that the PM₁₀ monitoring site was not representative of the St. Louis urban area, EPA does not consider the PM_{10-2.5} concentrations to be credible. However, for this assessment of various simplified methods to calculate PM_{2.5} light extinction, St. Louis data were not excluded since all but one of the comparisons are for calculated PM_{2.5} light extinction.

- 1 6. Hourly $PM_{10-2.5}$ data estimated by difference at sites with continuous monitors for both
2 $PM_{2.5}$ and $PM_{10-2.5}$ or by using typical regional ratios of $PM_{10-2.5}$ to $PM_{2.5}$ times the $PM_{2.5}$
3 concentrations where there are no PM_{10} measurements.
- 4 7. IMPROVE algorithm to estimate hourly $PM_{2.5}$ light extinction from hourly $PM_{2.5}$
5 component and hourly relative humidity values.

6 The process that was followed for this analysis was to examine the difference between
7 calculated hourly PM_{10} light extinction values used in the UFVA and values generated using
8 simpler approaches, explored in a step-wise fashion. All of the simpler approaches (designated
9 by letter of the alphabet and described below) use the IMPROVE algorithm and measured hourly
10 $PM_{2.5}$ mass and relative humidity data.

- 11 A. Omission of $PM_{10-2.5}$ contributions to PM light extinction (Step 6 above). (This was done
12 primarily to simplify comparisons with other simplified approaches; coarse mass
13 contributions were separately examined in approach C.) For clarity, light extinction
14 estimates are referred to as either $PM_{2.5}$ or PM_{10} light extinction in the remainder of this
15 Appendix to indicate whether the $PM_{10-2.5}$ contribution to PM light extinction is included.
- 16 B. Use of a simpler approach to organic carbonaceous mass and nitrate than in the
17 SANDWICH-based UFVA approach briefly described in the steps listed above. Table
18 4B-1 contrasts the steps in this simpler approach B (and approach F, below) to the steps
19 of approach A. The key distinguishing features of approach B are (i) a multiplier of 1.7
20 is applied to the CSN measurement of organic carbon (after a correction for blank filter
21 artifact) rather than estimating organic carbonaceous material by the mass-balance
22 SANDWICH approach, and (ii) the hourly $PM_{2.5}$ mass is normalized to match the FRM
23 value for the 24-hour $PM_{2.5}$ mass and then speciated using the CSN-measured component
24 mix, with no correction for nitrate loss from the FRM Teflon filter.
- 25 C. Same as approach B except that it includes $PM_{10-2.5}$ contributions to PM_{10} light
26 extinction,
- 27 D. Same as B except that the diurnal component distribution is assumed to be flat
28 (unvarying) for each sample day.
- 29 E. Same as B except that monthly averaged $PM_{2.5}$ component distributions are used in place
30 of the CSN sample period-specific values.
- 31 F. Same as B except both flat diurnal $PM_{2.5}$ component variation are assumed and monthly
32 averaged $PM_{2.5}$ component distribution are used in place of sample period specific values
33 (i.e., combines simplification of approaches D and E). Because staff recommends
34 consideration of this particular approach, for greatest clarity the data sources and
35 calculation steps used in this approach are listed in Table 4B-2.

- 1 I. Simply uses the $PM_{2.5}$ mass concentration times a single constant. (This approach was
- 2 included to show the performance of using $PM_{2.5}$ mass without adjusting for particle
- 3 composition and humidity effects on hygroscopic particles.)²

² Note that the letter designations for the approaches are not contiguous. This is because EPA staff tested additional approaches that are not included in this discussion appendix. Note also that all of the figures include approach designators, as plan designators.

Table 4B-1. Detailed Comparative Description of Approaches A, B, and F for Estimating 1-hour PM_{2.5} Light Extinction

UFVA Step [‡]	Aspect of Approach	Approach A	Approach B	Approach F
1	Estimation of 24-hour organic carbonaceous mass	The SANDWICH method (Frank, 2006) is used to subdivide the 24-hour PM _{2.5} mass reported by the FRM for each day and site into sulfate, nitrate elemental carbon, organic carbonaceous mass, and fine soil/crustal mass. This is done using information from the CSN measurements, physical models, and day-specific temperatures and relative humidity. The primary purpose of this SANDWICH step is to estimate organic carbonaceous mass on the FRM Teflon filter.	Organic carbonaceous mass is assumed to equal the organic carbon value reported from CSN sampling, minus a network-wide blank filter correction, times 1.7.	Same as B
1, continued	Estimation of 24-hour elemental carbon mass	CSN elemental carbon concentration	Same as A	Same as A
1, continued	Estimation of 24-hour sulfate mass	CSN sulfate concentration, with day-specific SANDWICH estimates of associated ammonium and water.	Sulfate ion measurement from the CSN filter is multiplied by 1.375 to represent dry, neutralized sulfate.	Same as B
1, continued	Estimation of 24-hour nitrate mass	Nitrate ion on the FRM Teflon filter is estimated by SANDWICH, with day-specific estimates of associated ammonium and water.	Nitrate ion measurement from the CSN filter is multiplied by 1.29 to represent dry, neutralized nitrate.	Same as B

[‡] The numbering of steps follows that used to describe the UFVA approach in section 3.3.1 of the UFVA.

UFVA Step[†]	Aspect of Approach	Approach A	Approach B	Approach F
1, continued	Estimation of 24-hour fine soil/crustal mass	Calculated from CSN elements, but without Al (a difference from the IMPROVE approach)	Same as A	Same as A
2	Diurnal pattern of PM _{2.5} components	The CMAQ-derived monthly normalized diurnal profiles for the sulfate, nitrate, elemental carbon, organic carbon and fine soil/crustal components (which average to 1.0 across 24 hours) were multiplied by the day-specific SANDWICH-based estimates of the 24-hour average concentrations of the five PM _{2.5} components, to get intermediate day-specific hourly estimates of the five components (including ammonium and water associated with sulfate and nitrate ion).	CMAQ-derived profiles were applied to the five components to get intermediate estimates. However, as described above, in Approach B the sulfate, nitrate, and organic carbonaceous material components are defined and estimated differently than in Approach A.	No diurnal profiles are used. This approach assumes that the percentage mix of PM _{2.5} components is the same in all hours of a single day, equal to the mix observed on the CSN filter (with the stated method for OCM).
3	Sum the 5 components	The hourly concentrations of these five components (including day-specific ammonium and water associated with sulfate and nitrate ion when the FRM Teflon filter is weighed) were added together, to get a sum-of-components estimate of hourly PM _{2.5} mass for the day of the FRM/CSN sampling.	The hourly concentrations of the five components were added together, to get a sum-of-components estimate of hourly PM _{2.5} for the day of FRM/CSN sampling. Note that water is not included.	Calculate the monthly-average percentage mix of the 5 PM _{2.5} components, as follows: For each day of 24-hour CSN sampling, sum the five components. Calculate the fraction of sum-of-5 for each component. Average the fraction for each component across the CSN sampling days in the month.

UFVA Step [±]	Aspect of Approach	Approach A	Approach B	Approach F
4	Hourly PM _{2.5} concentration, consistent with 24-hour FRM concentration.	The hourly data from the continuous PM _{2.5} instrument on the day of the FRM sampling were normalized by their 24-hour average, to get a normalized diurnal profile. This profile was applied to the 24-hour PM _{2.5} mass reported by the FRM sampler, to get a preliminary, FRM-consistent estimate of hourly PM _{2.5} mass for the day of the FRM sampling. This keeps the average of the valid 1-hour PM _{2.5} values equal to the 24-hour value from the FRM sampler.	Same as A	Same as A, but see the comment on this topic in Table B-2.
5, 6	Adjust preliminary estimates of hourly PM _{2.5} component concentrations (reflecting CMAQ diurnal profiles and 24-hour measurements) to be consistent with estimate of hourly PM _{2.5} mass.	The two estimates of hourly PM _{2.5} mass from steps 3 and 4 were compared, hour-by-hour. Within each hour, the estimates of all five components from step 2 were increased or decreased by a common percentage (referred to below as A _i where the subscript i indicates the hour) so that the sum of the five components after this adjustment was equal to the estimate of the hourly PM _{2.5} mass from step 4. The adjustment percentage varied from hour-to-hour.	Same as A	Not applicable. Monthly-average percentage mix of PM _{2.5} components is directly applied to the day-specific FRM-consistent hourly PM _{2.5} mass.

UFVA Step [±]	Aspect of Approach	Approach A	Approach B	Approach F
7	Adjust the FRM-consistent estimate of sulfate to the CSN/IMPROVE-consistent basis expected by the IMPROVE algorithm.	Each hourly estimate of sulfate concentration on the FRM filter from step 6 (which includes estimates of associated ammonium and particle bound water) was adjusted so that it excludes water and reflects full neutralization and therefore is consistent with the reporting practices of the IMPROVE program and the IMPROVE algorithm.	No adjustment is needed, given that the factors of 1.375 and 1.29 already assume full neutralization and no water. In effect, the mass of water on the FRM filter is allocated among the 5 components.	Same as B.
8	Adjust the FRM-consistent estimate of nitrate to the CSN/IMPROVE-consistent basis expected by the IMPROVE algorithm.	A similar adjustment as in step 7 (for sulfate) was made to each hour's nitrate concentration from step 6, so that the estimate of hourly nitrate would reflect actual atmospheric conditions and be consistent with the IMPROVE algorithm. This can result in the estimate of nitrate used in the IMPROVE algorithm being higher than the FRM-consistent estimate, for days on which the SANDWICH method predicts a loss of nitrate from the FRM filter.	No adjustment is made. Implication: On warm days when the FRM filter has lost nitrate mass, the estimates of hourly PM _{2.5} will be lower than actual atmospheric mass. All hourly PM _{2.5} components will be reduced, by the fraction that the lost nitrate is of total PM _{2.5} mass.	.Same as B.

UFVA Step[†]	Aspect of Approach	Approach A	Approach B	Approach F
Not numbered in UFVA	Estimation of PM _{2.5} light extinction from estimates of hourly concentrations of PM _{2.5} components.	Original IMPROVE algorithm, including f(RH) determined from hourly RH. Hours with RH >90% were excluded from design values and from most graphical displays of results.	Same as A	Same as A

1
2

Table 4B-2. Data Sources and Calculation Steps for Approach F

<p align="center">Approach F (As performed for results shown in this Appendix)</p>	<p align="center">Comments</p>
Data Sources	
<p>CSN values for sulfate ion (S), nitrate ion (N), elemental carbon (EC), organic carbon (OC), and the crustal elements Si, Ca, Fe, and Ti.</p>	<p>CSN sampling is currently performed at about 200 sites, some with one-in-six-days operation and some with one-in-three-days operation. In 2005-2007, several different sampler models were in use among the 15 study sites. Prospectively, all CSN sampling for S, N, EC, and the crustal elements will be based the Met One SASS or Met One SuperSASS sampler, and all CSN sites now use the URG2000 sampler for carbon measurements.</p> <p>If the revised IMPROVE algorithm were to be used in Step (ix) instead of the original IMPROVE algorithm, an option the Administrator may wish to consider, additional analysis results from CSN would be needed, in particular chlorine or chloride.</p>
<p>Fixed external estimates of organic carbon artifact on quartz filters, based on network-wide field blank data. Estimate depends on sampler model used for carbon sampling. These values ranged from 0.32 to 1.53 $\mu\text{g}/\text{m}^3$, depending on sampler model.</p>	<p>The artifact adjustment for the URG2000 sampler is of most interest prospectively, because it is the single sampler now in use for carbon sampling in CSN. The URG2000 was used only at about one-half of the 15 study sites and only in the second half of 2007. For those sites and days, an organic carbon artifact of 0.5 ug/m^3 was assumed for the purposes of the UFVA and this document, based on early experience with this sampler. Should the Administrator choose to invite public comment on an approach like Approach F, this value could be updated prior to NPRM based on a more recent, larger field blank data set.</p>
<p>Hourly $\text{PM}_{2.5}$ mass concentration from continuous instrument.</p>	<p>Steps (vii) and (viii) involve the scaling of hourly $\text{PM}_{2.5}$ mass concentrations from a continuous instrument to match the same-day 24-hour concentration reported by a collocated FRM/FEM filter-based sampler. Including this scaling in a regulatory program based on Approach F would mean that estimates of $\text{PM}_{2.5}$ light extinction could only be developed</p>
<p>FRM or FEM 24-hour $\text{PM}_{2.5}$ mass concentration</p>	

	<p>for days and locations with FRM/FEM filter-based sampling, implying the need for daily filter-based FRM/FEM sampling if estimates of light extinction are required on a daily basis.</p> <p>The Administrator may wish to consider requiring that only continuous instruments approved as federal equivalent methods (FEM) (for purposes of measuring 24-hour PM_{2.5} mass concentrations) may be used for PM_{2.5} light extinction estimation, and omitting the scaling steps. None of the continuous PM_{2.5} instruments used to collect the 2005-2007 data used in the UFVA analysis had been approved as FEM. Several models have now been approved. If the scaling steps are omitted, then FRM measurements would not be a required data source for this approach.</p>
Steps	
(i) For each CSN sampling day, subtract OC artifact from OC measurement, and multiply by 1.7 to estimate organic carbonaceous material (OCM).	
(ii) For each CSN sampling day, calculate fine soil/crustal PM _{2.5} (FS) from CSN measurements of crustal elements Si, Ca, Fe, and Ti, using the formula Fine soil PM _{2.5} = 3.73 × [Si] + 1.63 × [Ca] + 2.42 × [Fe] + 1.94 × [Ti]	
(iii) For each CSN sampling day, multiply CSN measurement of sulfate ion by 1.375, and multiply CSN measurement of nitrate ion by 1.29, to reflect associated ammonium under an assumption of full neutralization.	
(iv) Sum the above estimates of the 5 components of PM _{2.5} : Sum = 1.375*S + 1.29*N + OCM + EC + FS	
(v) For each CSN sampling day, calculate the 5 component fractions: sulfate fraction = 1.375*S/Sum Nitrate fraction = 1.29*N/Sum OCM fraction = OCM/Sum EC fraction = EC/Sum FS fraction = FS/Sum	
(vi) Average the fraction for sulfate from step	

<p>(v) across the CSN sampling days of that calendar month of that calendar year. Repeatn for the other 4 components. Call these the monthly-average component fractions.</p>	
<p>(vii) For each CSN sampling day, average the 24 values of 1-hour PM_{2.5} mass from the continuous instrument. Divide the 24-hour FRM value for PM_{2.5} mass by this average. Call this the “instrument scaling factor”.</p>	
<p>(viii) For each CSN sampling day, multiply each 1-hour PM_{2.5} mass value from the continuous instrument by the instrument scaling factor.</p>	
<p>(ix) For each daylight hour of each day of that month (including days without CSN sampling) multiply the value of hourly PM_{2.5} mass from step (viii) by the monthly-average component fractions from step (vi).</p>	
<p>(x) Insert the results from step (ix) into the original IMPROVE algorithm, along with f(RH) calculated based on same-hour RH. Omit the Rayleigh scattering term and the contribution from PM_{10-2.5}. Estimates of PM_{2.5} light extinction in hours with RH greater than 90% are not used in design value calculations and graphics presented in this document.</p>	<p>The Administrator may wish to consider proposing or seeking comment on the alternative of using the revised IMPROVE algorithm instead of the original algorithm, omitting the terms for Rayleigh scattering and NO₂ absorption. This approach that would allow explicit consideration of sea salt effects on light extinction.</p> <p>The exclusion of hours with RH>90% originated out of concern that an instrumental measurement of PM light extinction might result in high values of light extinction due to natural conditions of fog or precipitation. In as much as Approach F does not employ a light extinction instrument, the Administrator may wish to consider not excluding these hours from the form of the NAAQS. However, the accuracy of RH measurements at such high values of RH would still be a consideration.</p>

1 As the results below show, approaches B, D, E, and F produce hourly PM_{2.5} light
2 extinction values that are quite comparable to the hourly PM_{2.5} light extinction values in the
3 original UFVA (i.e., approach A). EPA staff favors consideration of approach F because it is the
4 most simplified of the methods tested that make use of CSN measurements, and its use of
5 monthly averaged PM_{2.5} components information means that it can be applied every day, not just
6 on days with CSN monitoring data (i.e., 1 day in 3 or 1 day in 6).⁴

7 Comparative Performance of Simplified PM_{2.5} Light Extinction Approaches: The
8 performance assessment of simplified approaches for calculated PM_{2.5} light extinction was
9 accomplished by comparing hourly values of PM_{2.5} light extinction generated by each approach
10 to their corresponding paired values generated using the original UFVA method, which is labeled
11 above as approach A. Annual box and whisker plots of the percentage differences between the
12 paired values, as well as annual and monthly scatter plots and regression analysis of these paired
13 data, were generated. Selected graphs and summary tables of regression statistics are included
14 below to show the degree of comparability between the various approaches to the original
15 estimates of PM_{2.5} light extinction. The complete set of these figures is available on EPA's web
16 site (http://www.epa.gov/ttn/analysis/pm_pad.htm).

17 The box and whisker plots of the differences between calculated hourly PM_{2.5} light
18 extinction by approaches B, D, E, and F are shown in Figures 4B-1 and 4B-2. In the box and
19 whisker plots of percentage difference, the percentage difference is calculated as follows:

20
21
$$\text{Percentage difference} = [(\text{"A"} \text{ estimate}) - (\text{simpler approach estimate})]/(\text{"A"} \text{ estimate}) * 100\%$$

22

23 The patterns of relatively small bias for the 15 urban areas are notably similar in each of the
24 plots. Keeping in mind that the differences in the approach are iterative with approach B
25 differing from approach A only by not using the SANDWICH model to estimate organic
26 component mass concentration and to adjust nitrate concentrations, while the three other
27 approaches (i.e., D, E, and F) add additional simplifications, it is perhaps surprising that the city-
28 to-city pattern of the box positions and sizes are remarkably similar for these four approaches.
29 This suggests that the simplifications between approaches A and B (i.e., replacing the
30 SANDWICH model) is responsible for the greatest amount of differences in calculated hourly
31 PM_{2.5} light extinction between approach A and D, E, and F, or in other words, the additional

⁴ Note that Approach F as applied for this Appendix required the availability of filter-based FRM/FEM data for 24-hour concentration, and that most filter-based PM_{2.5} monitoring sites do not operate every day. However, see the related comment in Table 4B-2.

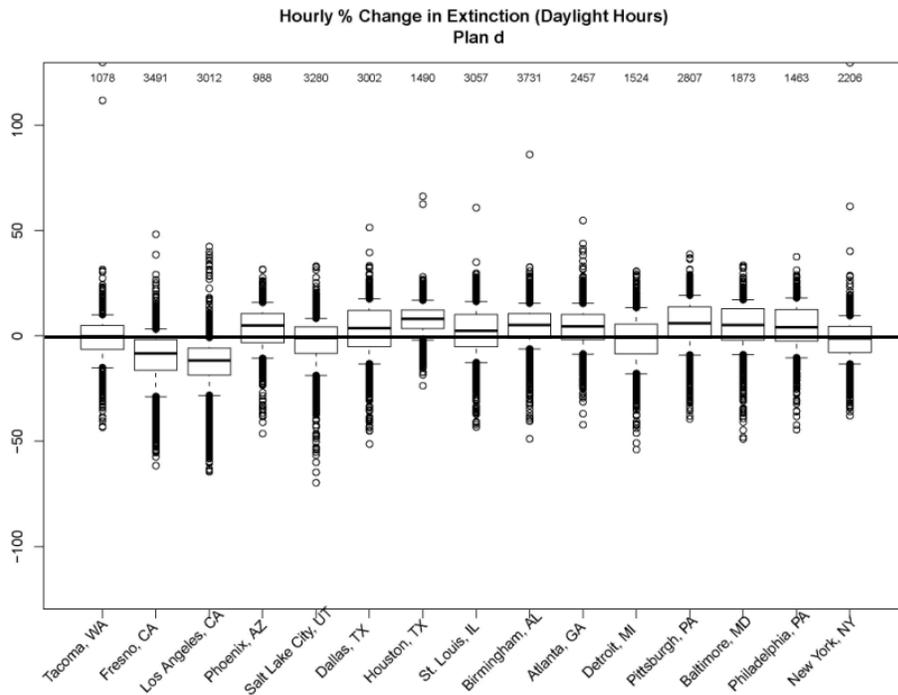
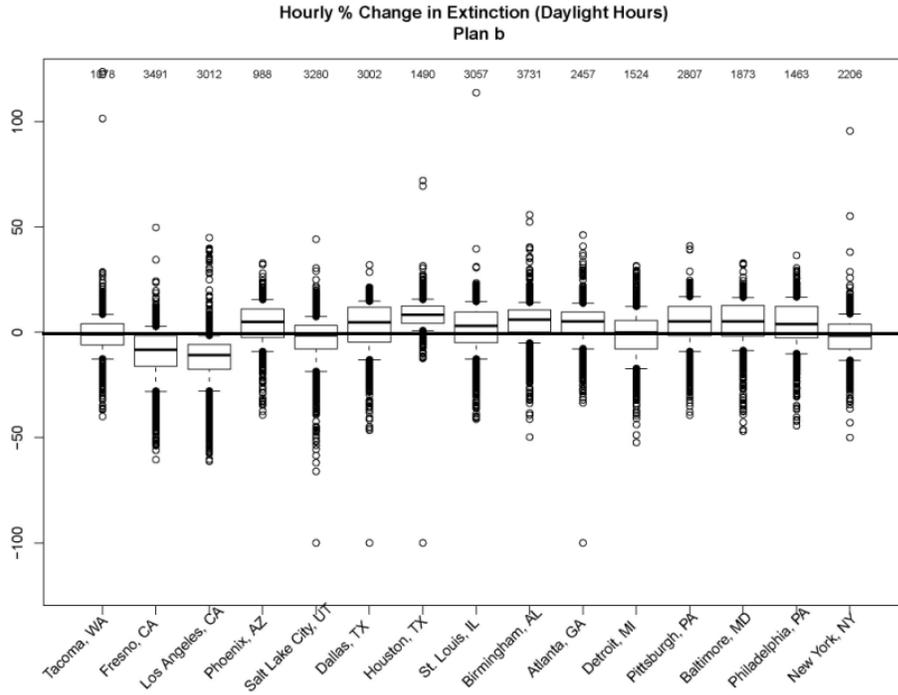
1 changes introduced in the remaining three approaches did not much affect the resulting hourly
2 values.⁵

3 The degree of comparability for paired hourly PM_{2.5} light extinction between approach F
4 and A, and between approach F and B values by month and urban area is evident in regression
5 statistics (Tables 4B-3 and 4B-4). In both tables the regression lines have slopes and R² values
6 near one with small intercepts for most urban areas and months implying that the values are
7 highly comparable. The western urban areas (e.g., Fresno, Houston, Los Angeles, Phoenix, Salt
8 Lake City and Tacoma) have slopes and R² values for some months that imply a bias and/or
9 noisier relationship between values calculated by approaches F and A. As expected the
10 regression relationships show that values calculated by approach F are more similar to those of
11 approach B (Table 4B-4) since neither uses the SANDWICH model, than to those of approach A
12 (Table 4B-3) which included the SANDWICH model estimates. Four scatter plots of calculated
13 hourly PM_{2.5} light extinction for all months that compare approach F to approach A and
14 approach F to approach B for Baltimore and Fresno are show in Figure 4B-3 as examples of the
15 degree of comparability for eastern and western urban areas.

16 A comprehensive assessment of the reasons that there are greater differences between
17 approaches A and B than between approaches B and any of D, E, and F has not been conducted.
18 However, some explanations are suggested by the results themselves. The fact that hour-to-hour
19 and day-to-day variations in the dry PM_{2.5} composition can be replaced by monthly averaged
20 values without much loss of precision in calculated hourly PM_{2.5} light extinction suggests that
21 these shorter term variations within a single month at a single monitoring site are not very
22 influential. This may imply that the differences between approach A and the other approaches
23 considered here are probably not due to the relative composition changes caused by use of the
24 SANDWICH model and the simpler approach B. This suggests that the difference may be due to
25 the adjustment to the FRM measured PM_{2.5} mass concentration that was done to account for
26 negative sampling artifact for ammonium nitrate. This is consistent with the behavior for sites
27 that have high ammonium nitrate (e.g., Fresno and Los Angeles). Depending on the results of
28 this more complete assessment of why the difference in hourly values are higher between
29 approach A and B then between those of B and any of D, E, and F, another approach that retains
30 the SANDWICH model but otherwise has the simplifications of approach F might be developed
31 as the better method to calculate PM_{2.5} light extinction.

⁵ In the box and whisker plots, some extreme values of the percentage change are due to rounding effects when the values involved were very small.

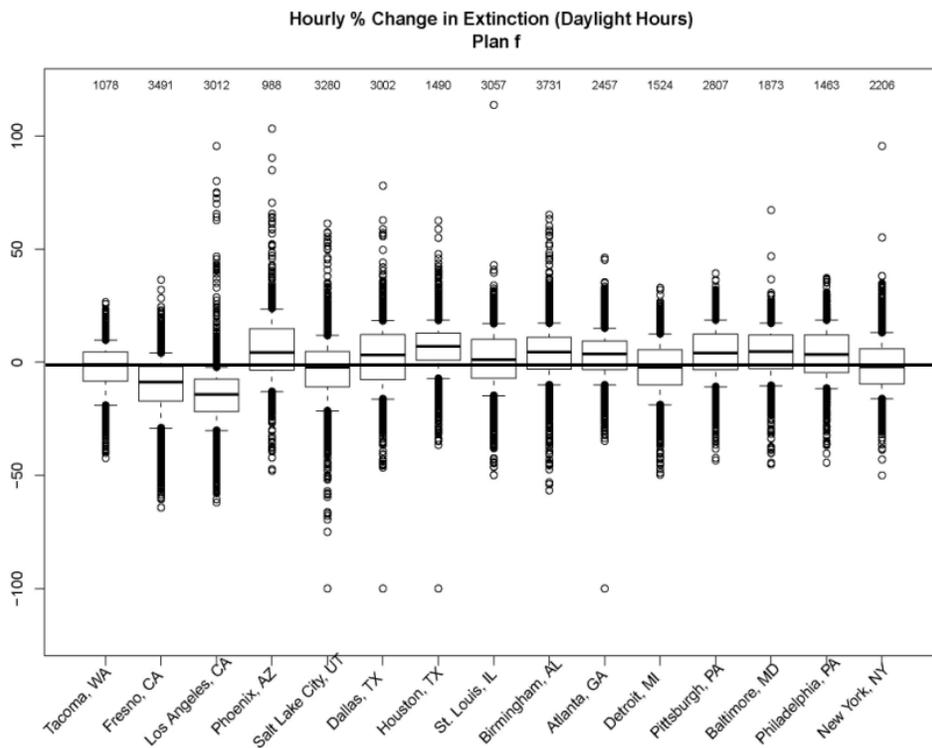
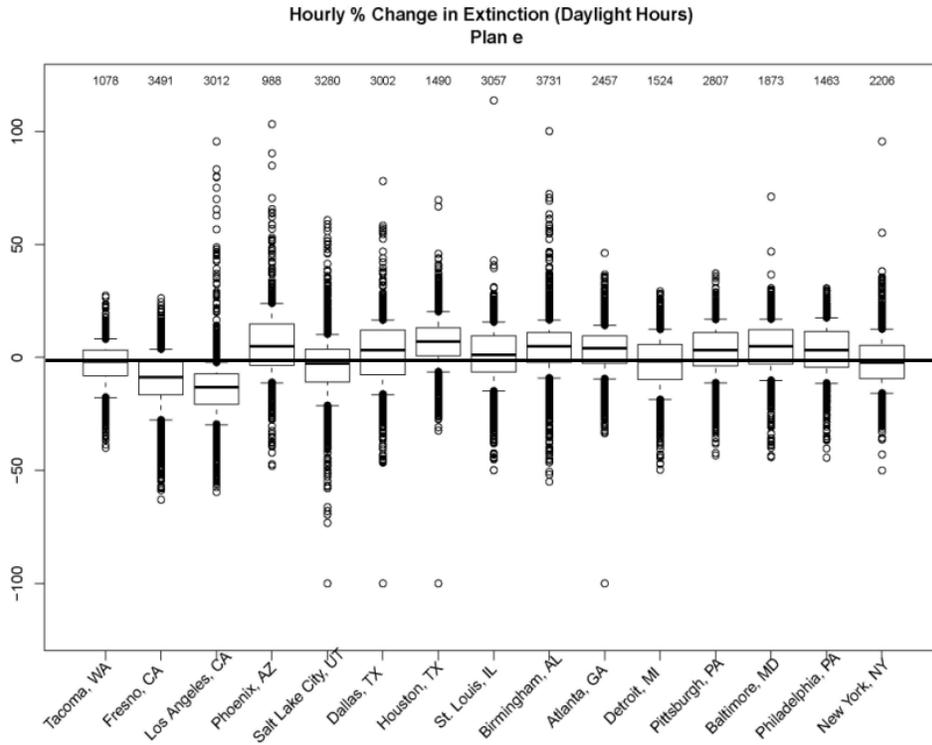
1 **Figure 4B-1 Box and whisker plot of the percent difference in calculated hourly $PM_{2.5}$ light extinction between**
 2 **approaches B and A (top plot) and D and A (bottom plot) by urban area.**



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Figure 4B-2 Box and whisker plot of the percent difference in calculated hourly PM_{2.5} light extinction between approaches E and A (top plot) and F and A (bottom plot) by urban area.

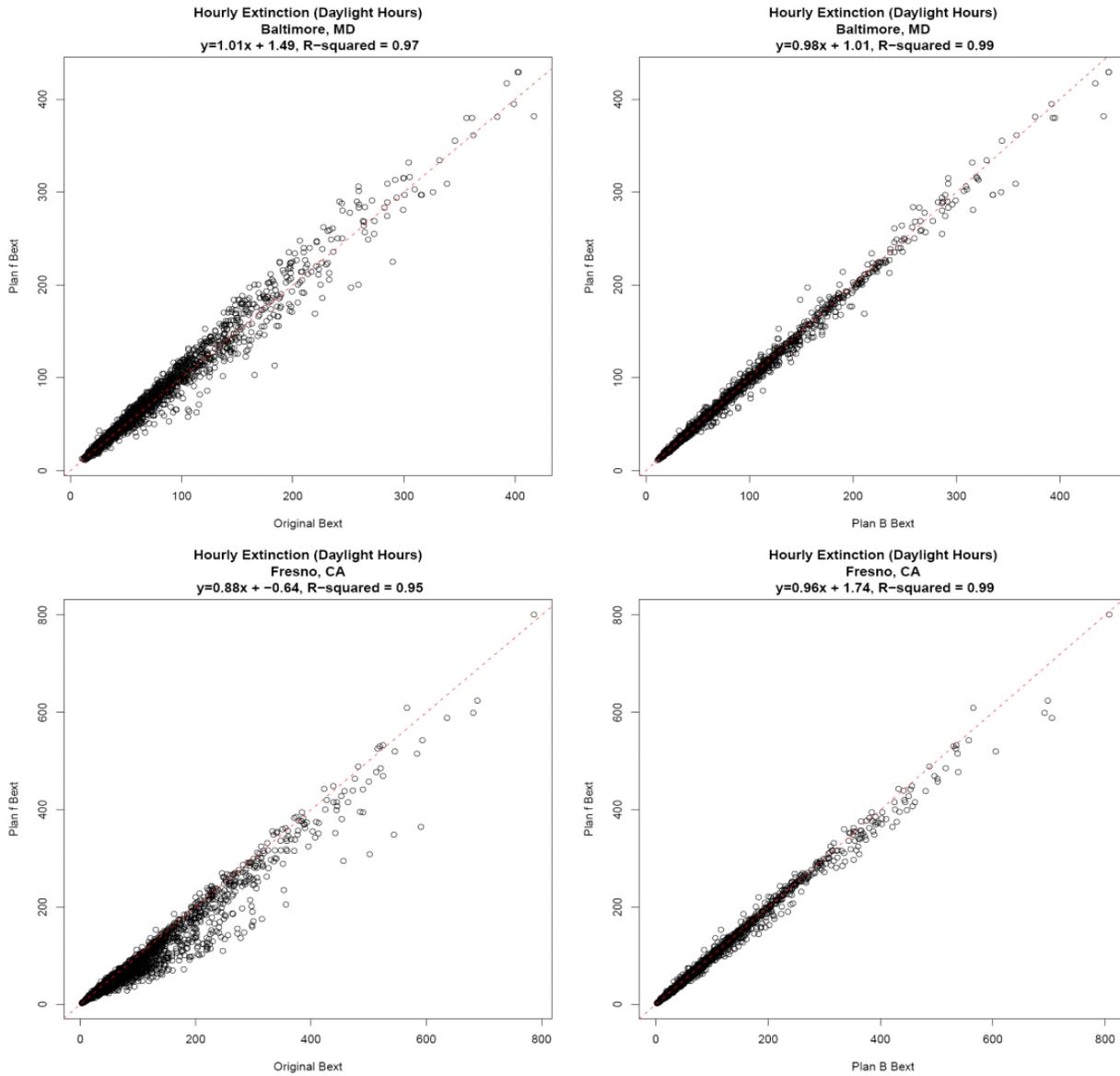


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Figure 4B-3 Scatter plots of calculated PM_{2.5} light extinction by approach F versus approach A (left) and B (right) for Baltimore (top) and Fresno (bottom).



4

Table 4B-3 Linear regression equation and R² values for relating hourly PM_{2.5} light extinction values calculated using approach F (x in the equation) to those using approach A (y in the equation) by month for 15 urban areas.

Month	1	2	3	4	5	6	7	8	9	10	11	12
Atlanta, GA	y=1.09*x+ -0.74; Rsq=0.98	y=1.03*x+ 1.85; Rsq=0.98	y=1.01*x+ -1.53; Rsq=0.97	y=1.06*x+ -4.65; Rsq=0.98	y=1.02*x+ 4.41; Rsq=0.97	y=0.99*x+ -3.52; Rsq=0.97	y=0.96*x+ 1.38; Rsq=0.96	y=0.98*x+ -4.01; Rsq=0.98	y=0.98*x+ -2.8; Rsq=0.98	y=0.97*x+ 0.42; Rsq=0.99	y=1.04*x+ 0.12; Rsq=0.96	y=1.04*x+ 0.67; Rsq=0.98
Baltimore, MD	y=0.99*x+ 1.31; Rsq=0.99	y=1.07*x+ -3.64; Rsq=1	y=0.97*x+ 3.35; Rsq=0.99	y=1.02*x+ 0.36; Rsq=0.96	y=0.99*x+ 0.83; Rsq=0.9	y=0.98*x+ -3.78; Rsq=0.96	y=0.94*x+ 2.59; Rsq=0.98	y=0.88*x+ 2.1; Rsq=0.99	y=0.96*x+ -3.25; Rsq=0.98	y=1.05*x+ -2.77; Rsq=0.98	y=1.01*x+ 5.62; Rsq=0.99	y=1.02*x+ 0.19; Rsq=0.97
Birmingham, AL	y=1.1*x+ 2.32; Rsq=0.97	y=1.04*x+ 0.59; Rsq=0.97	y=1.06*x+ -2.86; Rsq=0.97	y=1.06*x+ -3.91; Rsq=0.97	y=0.94*x+ 2.05; Rsq=0.97	y=0.98*x+ -4.9; Rsq=0.98	y=1.02*x+ 6.87; Rsq=0.97	y=0.9*x+ .31; Rsq=0.93	y=1*x+ 6.31; Rsq=0.98	y=1.06*x+ -3.87; Rsq=0.97	y=1.05*x+ -2.22; Rsq=0.98	y=1.02*x+ 0.37; Rsq=0.99
Dallas, TX	y=1.17*x+ -1; Rsq=0.97	y=1.1*x+ .12; Rsq=0.98	y=1.15*x+ -4.59; Rsq=0.96	y=1.04*x+ -2.64; Rsq=0.97	y=1.03*x+ 3.1; Rsq=0.96	y=0.93*x+ 0.75; Rsq=0.95	y=0.96*x+ 0.01; Rsq=0.96	y=0.98*x+ -0.63; Rsq=0.91	y=1.01*x+ -3.02; Rsq=0.93	y=1.06*x+ -2.35; Rsq=0.97	y=1.04*x+ -1.16; Rsq=0.96	y=1.07*x+ 1.11; Rsq=0.99
Detroit, MI	y=1.02*x+ 2.64; Rsq=0.97	y=0.96*x+ 3.31; Rsq=0.99	y=1*x+ 1.92; Rsq=0.99	y=1.1*x+ 0.43; Rsq=0.96	y=0.92*x+ 5.49; Rsq=0.96	y=0.96*x+ 2.14; Rsq=0.96	y=0.99*x+ 0.17; Rsq=0.93	y=1.29*x+ -18.07; Rsq=0.9	y=1.11*x+ -8.62; Rsq=0.97	y=1.18*x+ -3.49; Rsq=0.95	y=1.05*x+ 5.63; Rsq=1	y=0.95*x+ 3.25; Rsq=0.99
Fresno, CA	y=1.05*x+ -0.61; Rsq=0.98	y=1.06*x+ 2.02; Rsq=0.99	y=1.17*x+ 4.27; Rsq=0.96	y=1.32*x+ -4.32; Rsq=0.86	y=1.27*x+ 2.92; Rsq=0.83	y=1.24*x+ -5.26; Rsq=0.87	y=1.06*x+ 0.97; Rsq=0.97	y=1.3*x+ 6.59; Rsq=0.86	y=1.39*x+ -6.67; Rsq=0.89	y=1.51*x+ -8.53; Rsq=0.94	y=1.08*x+ 5.8; Rsq=0.98	y=0.98*x+ 6.01; Rsq=0.99
Houston, TX	y=0.83*x+ 4.6; Rsq=0.96	y=0.97*x+ -1.12; Rsq=0.99	y=0.9*x+ 0.8; Rsq=0.99	y=1.02*x+ -3.72; Rsq=0.98	y=0.97*x+ 2.87; Rsq=0.97	y=0.84*x+ 7.59; Rsq=0.92	y=0.76*x+ 11.38; Rsq=0.85	y=0.88*x+ 4.71; Rsq=0.94	y=0.99*x+ -2.92; Rsq=0.98	y=1.06*x+ -3.39; Rsq=0.99	y=1.07*x+ -3.64; Rsq=0.95	y=1.06*x+ -2.09; Rsq=0.98
Los Angeles, CA	y=1.21*x+ -0.76; Rsq=0.98	y=1.13*x+ 1.34; Rsq=0.98	y=1.1*x+ .13; Rsq=0.96	y=1.16*x+ 1.82; Rsq=0.96	y=1.05*x+ 13.01; Rsq=0.96	y=1.35*x+ -4.71; Rsq=0.86	y=1.29*x+ 4.96; Rsq=0.91	y=1.4*x+ 14.24; Rsq=0.82	y=1.43*x+ -10.81; Rsq=0.91	y=1.27*x+ -1.2; Rsq=0.95	y=1.1*x+ .4; Rsq=0.97	y=1.1*x+ .08; Rsq=0.97
New York, NY	y=1.04*x+ 1.72; Rsq=0.98	y=1.03*x+ -1.34; Rsq=1	y=1.01*x+ 3.48; Rsq=0.99	y=1.11*x+ -0.74; Rsq=0.98	y=1.06*x+ 1.45; Rsq=0.97	y=1.11*x+ -6.56; Rsq=0.96	y=1.02*x+ 7.96; Rsq=0.96	y=0.98*x+ -1.61; Rsq=0.97	y=1.03*x+ 0.24; Rsq=0.96	y=1.22*x+ -7.88; Rsq=0.97	y=1.09*x+ -0.86; Rsq=0.99	y=1.06*x+ 1.17; Rsq=0.98

Philadel phia, PA	y=0.98*x+ 2.25; Rsq=0.99	y=1.01*x+ -1.64; Rsq=0.99	y=1*x+1.3 2; Rsq=0.99	y=0.96*x+ 2.4; Rsq=0.94	y=0.9*x+1. 44; Rsq=0.98	y=0.92*x+ -0.44; Rsq=0.98	y=0.91*x+ 1.75; Rsq=0.98	y=0.85*x+ 5.59; Rsq=0.98	y=0.89*x+ 1.3; Rsq=0.96	y=0.95*x+ 3.87; Rsq=0.97	y=1.08*x+ 0.34; Rsq=0.99	y=0.97*x+ 2.37; Rsq=0.99
Phoenix, AZ	y=1.21*x+ -2.37; Rsq=0.89	y=1.13*x+ -1.38; Rsq=0.97	y=0.88*x+ 1.25; Rsq=0.96	y=0.85*x+ 1.94; Rsq=0.94	y=0.93*x+ 0.87; Rsq=0.9	y=0.88*x+ 1.65; Rsq=0.97	y=0.77*x+ 4.02; Rsq=0.69	y=0.92*x+ 0.62; Rsq=0.99	y=0.86*x+ 0.48; Rsq=0.97	y=0.92*x+ 1.18; Rsq=0.72	y=1.05*x+ -1.68; Rsq=0.92	y=1.16*x+ -2.68; Rsq=0.94
Pittsburg h, PA	y=1.1*x+ -3.59; Rsq=0.95	y=0.99*x+ 1.91; Rsq=0.97	y=0.99*x+ -0.27; Rsq=0.99	y=1.09*x+ -2.34; Rsq=0.96	y=1.07*x+ 5.67; Rsq=0.92	y=0.95*x+ -1.28; Rsq=0.97	y=1.03*x+ 9.91; Rsq=0.96	y=0.94*x+ -1.93; Rsq=0.97	y=1.04*x+ -7.42; Rsq=0.94	y=1.02*x+ -1.66; Rsq=0.97	y=1.06*x+ -1.26; Rsq=0.99	y=0.99*x+ 2.25; Rsq=0.98
Salt Lake City, UT	y=0.91*x+ 8.57; Rsq=1	y=0.97*x+ 4.76; Rsq=0.99	y=1.12*x+ 1.93; Rsq=0.97	y=1.14*x+ -1.25; Rsq=0.88	y=0.98*x+ 0.92; Rsq=0.83	y=1.02*x+ 0.99; Rsq=0.87	y=1*x+0; Rsq=0.99	y=0.98*x+ -0.43; Rsq=0.97	y=0.88*x+ 5.49; Rsq=0.87	y=1.09*x+ 0.11; Rsq=0.93	y=1.06*x+ 0.77; Rsq=0.97	y=0.91*x+ 5.95; Rsq=0.99
St Louis, MO	y=1.01*x+ 1.63; Rsq=0.99	y=1.03*x+ -0.41; Rsq=1	y=1.07*x+ -0.47; Rsq=0.98	y=1.14*x+ -3.93; Rsq=0.94	y=1.06*x+ 2.9; Rsq=0.92	y=0.92*x+ 0.97; Rsq=0.95	y=0.98*x+ 2.77; Rsq=0.97	y=1*x+ 7.97; Rsq=0.98	y=0.94*x+ -0.89; Rsq=0.97	y=1.2*x+ 4.17; Rsq=0.95	y=1*x+4.6 6; Rsq=0.98	y=1.01*x+ 1.69; Rsq=0.99
Tacoma, WA	y=1.02*x+ -0.05; Rsq=1	y=0.98*x+ 0.71; Rsq=0.94	y=1.07*x+ -0.1; Rsq=0.97	y=1.22*x+ -3.78; Rsq=0.96	y=1.18*x+ 4.43; Rsq=0.92	y=1.09*x+ -0.14; Rsq=0.9	y=1.1*x+ 1.28; Rsq=0.94	y=1.22*x+ -4.35; Rsq=0.95	y=1.06*x+ 0.02; Rsq=0.96	y=0.97*x+ 1.2; Rsq=0.98	y=1.05*x+ 0.83; Rsq=1	y=1.02*x+ 2.4; Rsq=0.97

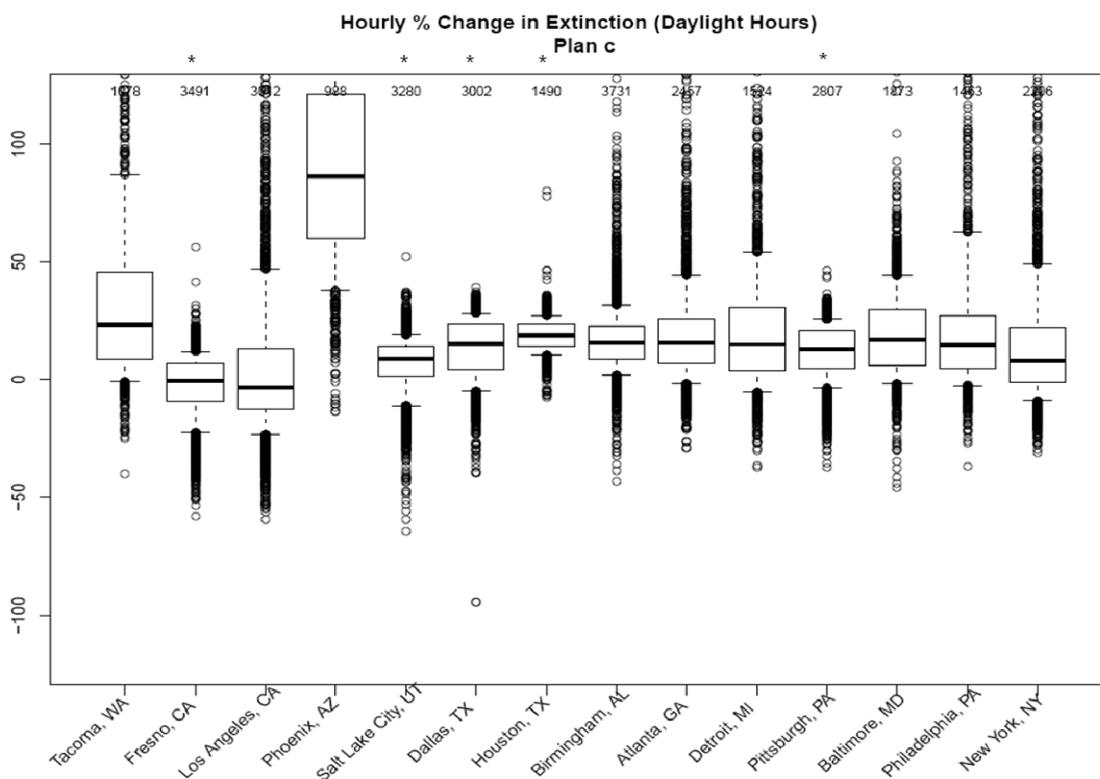
Table 4B-4 Linear regression equation and R² values for relating hourly PM_{2.5} light extinction values calculated using approach F (x in the equation) to those using approach B (y in the equation) by month for 15 urban areas.

Month	1	2	3	4	5	6	7	8	9	10	11	12
Atlanta, GA	y=1.01*x +0.4; Rsqr=0.98	y=1.06*x +0.26; Rsqr=0.96	y=1.02*x +0.15; Rsqr=0.99	y=1.02*x +0.72; Rsqr=0.99	y=1.02*x +0.83; Rsqr=0.99	y=1.02*x +1.06; Rsqr=0.99	y=1.02*x -0.38; Rsqr=0.95	y=1.04*x +3.54; Rsqr=0.98	y=1.02*x +0.74; Rsqr=0.99	y=1*x+1.11; Rsqr=0.99	y=1*x+1.2; Rsqr=0.98	y=0.99*x +0.81; Rsqr=0.99
Baltimore, MD	y=0.96*x +1.84; Rsqr=0.99	y=1.09*x +5.5; Rsqr=0.99	y=1.04*x +1.21; Rsqr=1	y=1.01*x +1.15; Rsqr=0.98	y=1.02*x +1.15; Rsqr=1	y=0.99*x +0.73; Rsqr=0.99	y=1.01*x -0.36; Rsqr=0.98	y=1*x+0.25; Rsqr=1	y=1.02*x +1; Rsqr=1	y=0.97*x +2.23; Rsqr=0.98	y=0.99*x +2.98; Rsqr=0.99	y=1.02*x +0.17; Rsqr=0.98
Birmingham, AL	y=1.08*x +2.37; Rsqr=0.98	y=0.98*x +3.39; Rsqr=0.96	y=1.02*x +0; Rsqr=0.99	y=1.07*x +2.22; Rsqr=0.98	y=1*x+1.38; Rsqr=0.99	y=1.01*x +0.71; Rsqr=0.99	y=1.06*x -4.76; Rsqr=0.98	y=0.98*x +2.12; Rsqr=0.95	y=1.02*x +2.17; Rsqr=0.98	y=1*x+1.46; Rsqr=0.98	y=1.02*x +0.76; Rsqr=0.99	y=1*x+1.59; Rsqr=0.99
Dallas, TX	y=1.08*x +1.32; Rsqr=0.99	y=1.05*x +1.39; Rsqr=0.99	y=1.06*x +1.42; Rsqr=0.99	y=1.08*x +2.66; Rsqr=0.99	y=1.04*x +1.58; Rsqr=0.99	y=0.98*x +1.45; Rsqr=0.95	y=0.99*x +0.46; Rsqr=0.96	y=0.94*x +2.44; Rsqr=0.91	y=1*x+0.52; Rsqr=0.99	y=1.02*x +0.48; Rsqr=0.99	y=1.03*x +0.71; Rsqr=0.99	y=1.13*x +3.74; Rsqr=0.99
Detroit, MI	y=1.06*x +4.03; Rsqr=0.98	y=1.05*x +2.66; Rsqr=1	y=1.03*x +1.35; Rsqr=0.99	y=1.01*x +0.23; Rsqr=0.99	y=1*x+0.08; Rsqr=1	y=0.98*x +0.79; Rsqr=0.99	y=1.02*x -0.65; Rsqr=0.99	y=1.09*x +5.76; Rsqr=0.97	y=1.01*x +0.83; Rsqr=0.99	y=1.05*x +1.29; Rsqr=0.99	y=0.99*x +4.66; Rsqr=1	y=1.09*x +5.58; Rsqr=0.95
Fresno, CA	y=1.06*x +5.57; Rsqr=0.99	y=1.05*x +4.18; Rsqr=0.99	y=1.03*x +0.85; Rsqr=1	y=1.02*x +0.57; Rsqr=0.99	y=1.02*x +0.24; Rsqr=0.98	y=1.03*x +0.55; Rsqr=0.99	y=1*x+0.76; Rsqr=1	y=1.02*x +0.58; Rsqr=0.99	y=1.04*x +1.01; Rsqr=0.99	y=1.1*x+3.18; Rsqr=0.99	y=1.05*x +4.61; Rsqr=0.99	y=1.04*x +4.37; Rsqr=0.99
Houston, TX	y=0.97*x +2.57; Rsqr=0.99	y=1.08*x +1.78; Rsqr=0.97	y=1.01*x +0.8; Rsqr=0.99	y=1.04*x +0.93; Rsqr=0.99	y=1.02*x +0.48; Rsqr=0.98	y=0.9*x+8.05; Rsqr=0.93	y=0.77*x+13.92; Rsqr=0.81	y=0.92*x +6.43; Rsqr=0.94	y=1.01*x +0.91; Rsqr=0.99	y=1.05*x +1.77; Rsqr=0.99	y=1.12*x +3.15; Rsqr=0.99	y=1.14*x +2.39; Rsqr=0.97
Los Angeles, CA	y=1.08*x +3.01; Rsqr=0.99	y=1.04*x +1.62; Rsqr=1	y=1.02*x +0.27; Rsqr=0.99	y=1.05*x +2.99; Rsqr=0.99	y=1*x+0.88; Rsqr=1	y=1.05*x +2.15; Rsqr=0.99	y=1.03*x -0.32; Rsqr=0.99	y=1.03*x +0.34; Rsqr=0.99	y=1.06*x +1.65; Rsqr=0.99	y=1.08*x +3.22; Rsqr=0.99	y=1.06*x +0.62; Rsqr=1	y=1.04*x +0.14; Rsqr=0.99

New York, NY	$y=1*x+0.52$; Rsq=1	$y=1.01*x+0.82$; Rsq=1	$y=0.99*x+0.86$; Rsq=1	$y=1.03*x+0.26$; Rsq=1	$y=1*x+0.17$; Rsq=0.99	$y=1.07*x+3.4$; Rsq=0.99	$y=1.03*x+4.03$; Rsq=0.99	$y=1.02*x+2.43$; Rsq=0.99	$y=1.01*x+0.94$; Rsq=1	$y=1.06*x+3.4$; Rsq=0.99	$y=1.02*x+0.84$; Rsq=1	$y=1*x+1.11$; Rsq=1
Philadelphia, PA	$y=1.03*x+1.63$; Rsq=0.99	$y=1.02*x+2.07$; Rsq=0.99	$y=1.02*x+0.65$; Rsq=1	$y=1.05*x+1.63$; Rsq=0.98	$y=0.99*x+0.1$; Rsq=1	$y=0.99*x+0.42$; Rsq=1	$y=1.02*x+0.32$; Rsq=0.98	$y=0.99*x+2.35$; Rsq=0.99	$y=0.96*x+1.39$; Rsq=0.99	$y=1.01*x+0.84$; Rsq=1	$y=1.02*x+0.83$; Rsq=1	$y=1*x+0.41$; Rsq=1
Phoenix, AZ	$y=1.17*x+3.94$; Rsq=0.93	$y=0.95*x+1.15$; Rsq=0.99	$y=0.84*x+1.91$; Rsq=0.97	$y=0.9*x+1.35$; Rsq=0.94	$y=0.98*x+0.6$; Rsq=0.96	$y=0.98*x+0.63$; Rsq=0.98	$y=0.88*x+3.02$; Rsq=0.81	$y=0.94*x+1.07$; Rsq=0.99	$y=0.97*x+0.28$; Rsq=0.98	$y=0.98*x+1.01$; Rsq=0.89	$y=0.98*x+0.03$; Rsq=0.96	$y=1.06*x+3.32$; Rsq=0.97
Pittsburgh, PA	$y=1.04*x+0.5$; Rsq=0.97	$y=1*x+0.78$; Rsq=0.99	$y=1.01*x+0.29$; Rsq=1	$y=1.05*x+0.59$; Rsq=0.99	$y=1.01*x+0.89$; Rsq=0.99	$y=1*x+0.02$; Rsq=0.99	$y=1.02*x+3.12$; Rsq=0.99	$y=1.04*x+3.96$; Rsq=0.99	$y=1.02*x+1.58$; Rsq=0.99	$y=0.98*x+2.1$; Rsq=0.98	$y=1.05*x+2.23$; Rsq=0.99	$y=0.95*x+4.33$; Rsq=0.98
Salt Lake City, UT	$y=1*x+1$; Rsq=1	$y=1.02*x+0.94$; Rsq=1	$y=1.07*x+0.81$; Rsq=0.99	$y=1.05*x+0.51$; Rsq=0.95	$y=0.98*x+0.79$; Rsq=0.96	$y=1.02*x+0.44$; Rsq=0.99	$y=0.97*x+0.77$; Rsq=0.99	$y=0.96*x+0.74$; Rsq=0.96	$y=0.75*x+8.1$; Rsq=0.8	$y=0.97*x+1.37$; Rsq=0.99	$y=1*x+0.22$; Rsq=1	$y=1*x+0.24$; Rsq=1
St Louis, MO	$y=1.04*x+0.73$; Rsq=0.99	$y=1.03*x+0.62$; Rsq=1	$y=1.08*x+4.01$; Rsq=0.99	$y=1.08*x+2.89$; Rsq=0.98	$y=1.03*x+0.95$; Rsq=0.97	$y=0.99*x+0.5$; Rsq=0.99	$y=1.03*x+1.96$; Rsq=0.99	$y=1.02*x+3.12$; Rsq=0.99	$y=1*x+0.23$; Rsq=0.99	$y=1*x+1.03$; Rsq=0.99	$y=0.93*x+4.74$; Rsq=0.98	$y=1.09*x+4.55$; Rsq=0.98
Tacoma, WA	$y=0.97*x+1.51$; Rsq=1	$y=1.08*x+3.92$; Rsq=0.73	$y=1.02*x+0.31$; Rsq=0.97	$y=1.03*x+0.72$; Rsq=0.99	$y=1.02*x+0.71$; Rsq=0.98	$y=1.02*x+0.21$; Rsq=0.99	$y=1.04*x+0.37$; Rsq=0.98	$y=1.03*x+0.58$; Rsq=0.99	$y=0.98*x+1.41$; Rsq=0.98	$y=0.97*x+0.92$; Rsq=0.99	$y=1.02*x+0.03$; Rsq=1	$y=0.99*x+2.77$; Rsq=0.97

PM_{10-2.5} Light Extinction: The source of the PM_{10-2.5} data used to calculate its contribution to PM₁₀ light extinction for the 14 cities varies by urban area. As described in the UFVA (section 3.2.1), for some urban areas hourly PM_{10-2.5} was estimated by the difference method. Birmingham, Detroit, Baltimore and Philadelphia had collocated continuous PM₁₀ and PM_{2.5} monitors, while Tacoma, Los Angeles, Phoenix, Atlanta, and New York had two separate sites with continuous PM₁₀ and PM_{2.5} monitors. The other urban areas used a regional ratio of PM_{10-2.5} to PM_{2.5} multiplied by hourly PM_{2.5} to estimate PM_{10-2.5}. While this approach provides some insight about the long-term contributions to light extinction by PM_{10-2.5}, the hourly estimates using a ratio times PM_{2.5} do not provide meaningful hourly information, so the results for those urban areas are not discussed in this assessment. Figure 4B-4 is a box plot of the percent difference between the estimate of hourly PM₁₀ light extinction estimated by approach C (includes coarse PM contributions) and the estimate of hourly PM_{2.5} light extinction by approach A (does not include coarse PM contributions). Urban areas that used the ratio method to estimate PM_{10-2.5} are labeled with an asterisk above the figure box. The other difference between calculated PM light extinction between the two approaches is the use of the SANDWICH model in approach A that is not used in Approach C. It is this difference that is responsible for the negative values in the box plots.

Figure 4B-4 Box and whisker plot of the percent difference between approaches C and approach A paired estimates of hourly PM₁₀ light extinction. Urban areas that used the ratio method to estimate PM_{10-2.5} are designated with an asterisk (*) above the figure box.



The relative importance of PM_{10-2.5} to light extinction is more clearly shown by comparing hourly calculated PM₁₀ light extinction values from approach C to paired hourly PM_{2.5} light extinction from approach B, which is identical except for the PM_{10-2.5} contributions. As can be seen in example scatter plots 4Figure B-5 (urban areas with collocated sampler) and Figure 4B-6 (urban areas using two different sites), PM₁₀ light extinction equals or exceeds PM_{2.5} when approaches C and B are compared. Not surprisingly these show that PM_{10-2.5} contributions to PM light extinction are not often important for most of the urban areas, especially those in the Eastern U.S., though they can on rare occasions contribute as much as 100 Mm⁻¹ to the PM_{2.5} light extinction at some sites (e.g., Birmingham, Detroit, Philadelphia, and Atlanta). Based on these scatter plots PM_{10-2.5} light extinction is a much more frequent and relatively large contributor to PM₁₀ light extinction at Phoenix (i.e., regression slope of 1.55) and an infrequent but large contributor to otherwise low PM₁₀ light extinction hours at Los Angeles. However, the Los Angeles PM_{10-2.5} data should be treated skeptically, since the PM₁₀ monitoring was done at a site outside the LA basin that may be unrepresentative of LA conditions. The PM_{10-2.5} contribution to PM light extinction at Tacoma is about 50 Mm⁻¹, contributing about 7% (i.e., regression slope is 1.07).

Figure 4B-5 Scatter plots comparing hourly PM light extinction for approaches C and B, for the four urban areas with collocated PM₁₀ and PM_{2.5} monitoring.

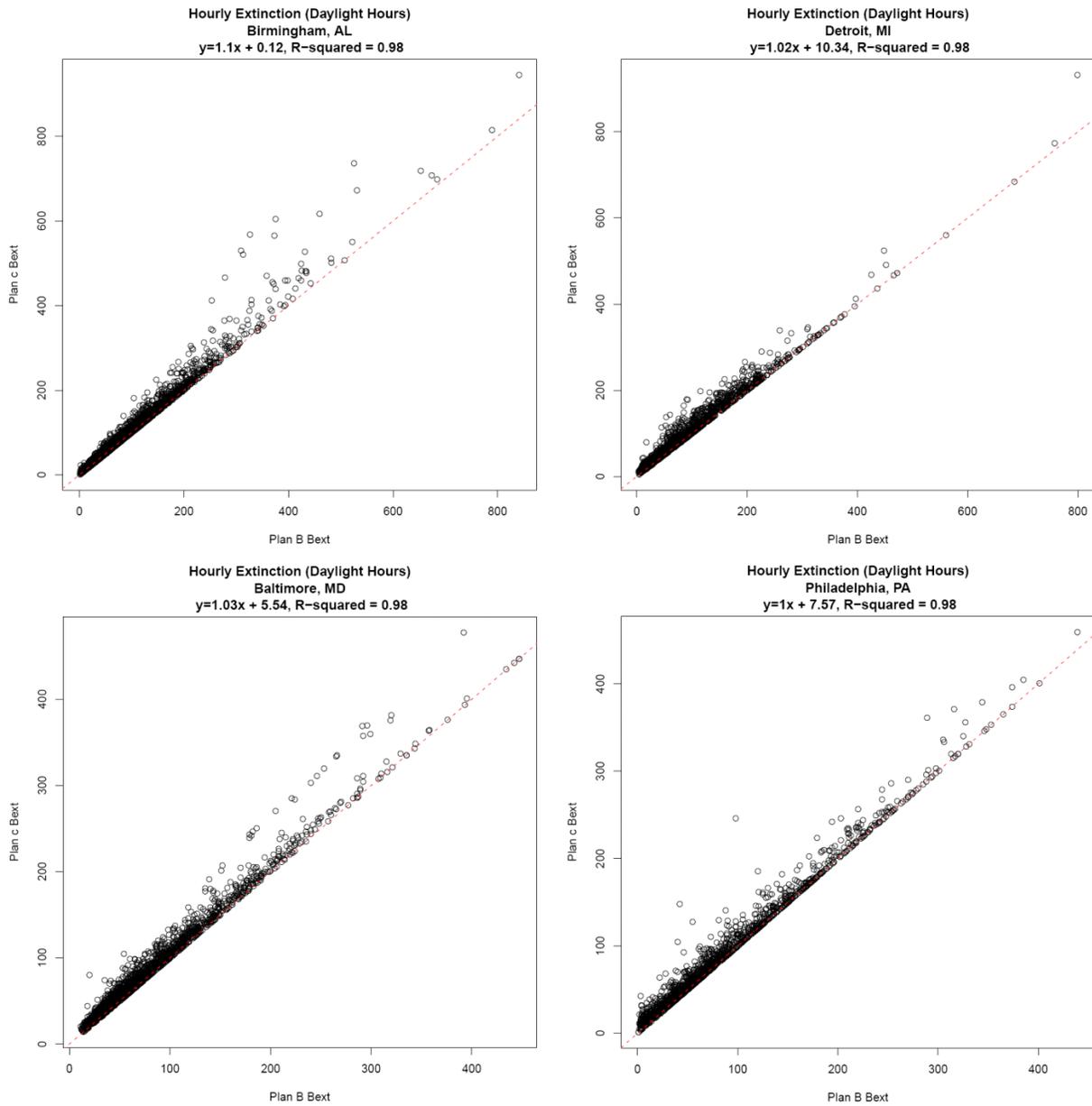
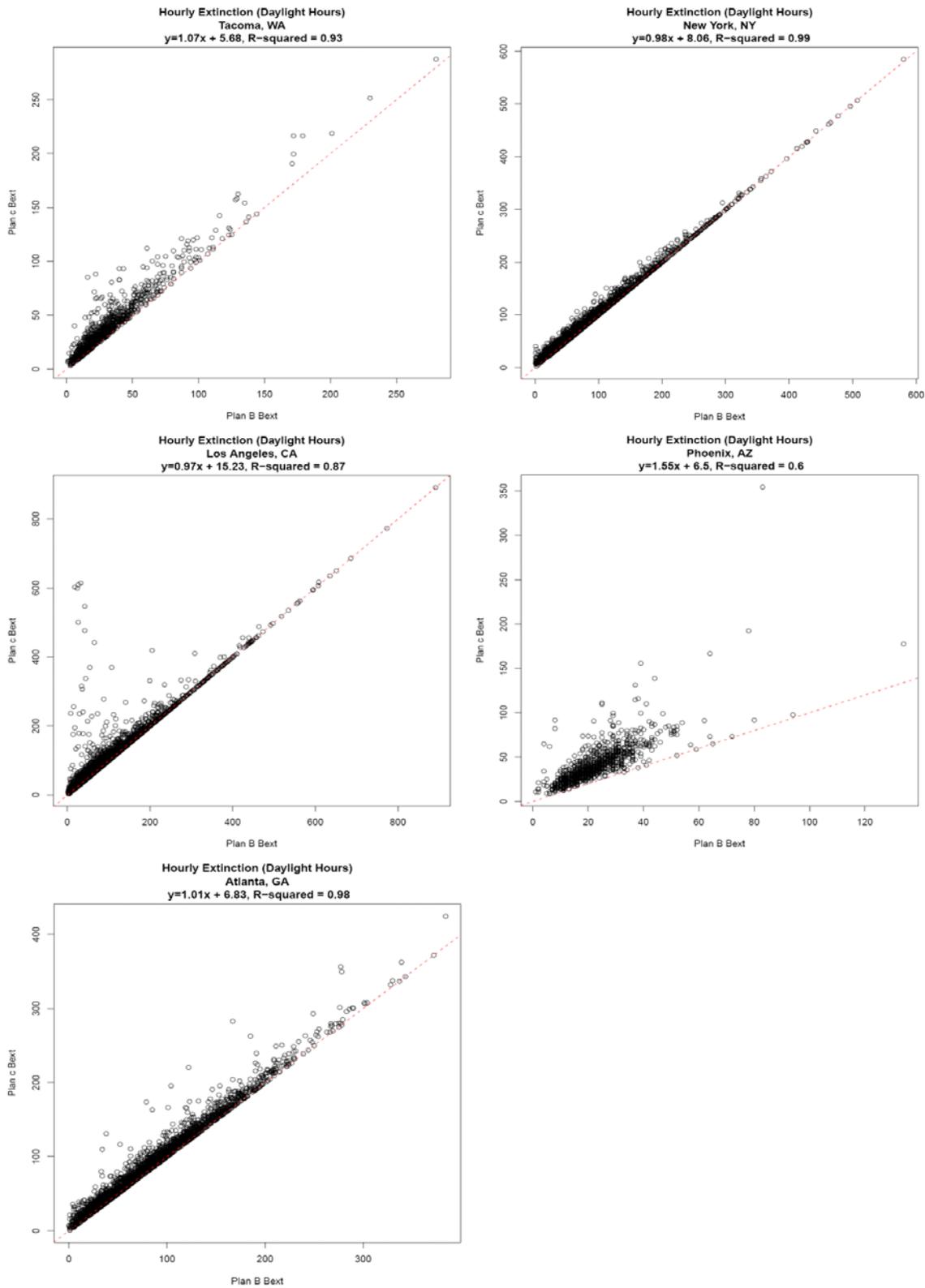


Figure 4B-6 Scatter plots comparing hourly PM light extinction for approaches C and B, for the five urban areas with different sites for PM₁₀ and PM_{2.5} monitoring.

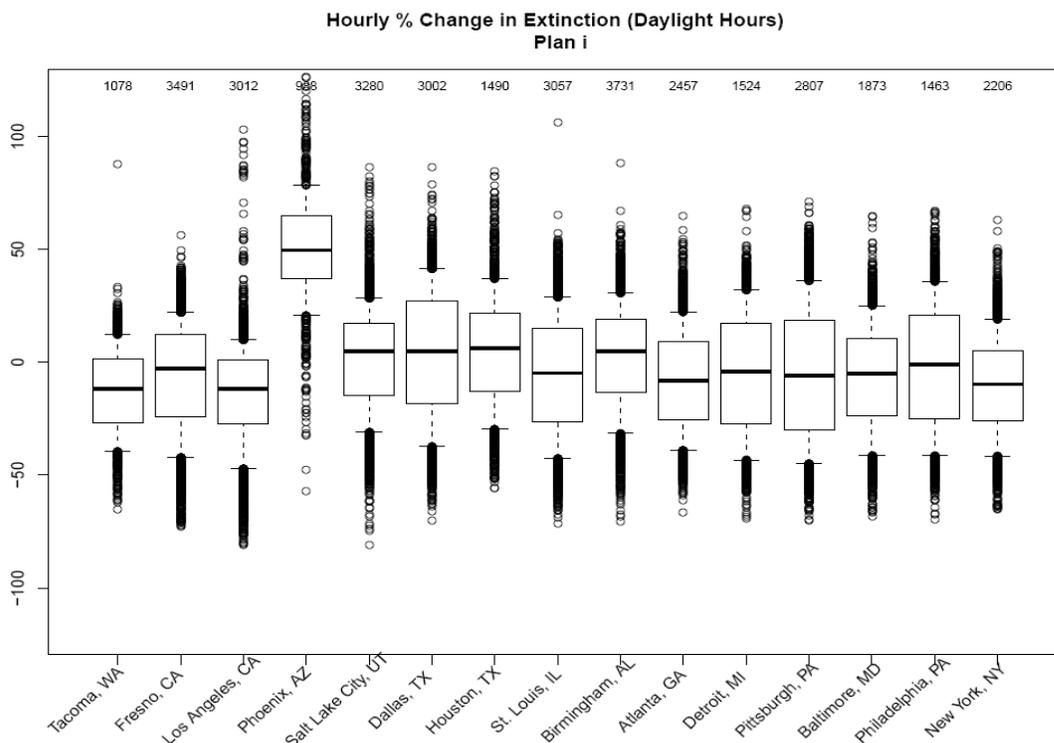


PM_{2.5} Mass Concentration Used Alone to Calculate PM_{2.5} Light Extinction: Approach I predicts PM_{2.5} light extinction by merely multiplying the same hourly PM_{2.5} values as used in approach A (continuous PM_{2.5} instrument normalized to match the FRM 24-hour concentration) by a constant (i.e., 4.35 m²/g).⁶ As shown in Figures 4B-7 and 4B-8, the results are not nearly as comparable as those calculated using approach F (or any of the methods that include composition and hourly relative humidity data). In Figures 4B-7 and 4B-8, the percentage difference is calculated as follows, which is different than for the earlier box and whisker plots:

$$\text{Percentage difference} = [(\text{“C” estimate}) - \text{“A” estimate}]/(\text{“C” estimate}) * 100\%$$

The variations in the relationships are even more evident in the monthly regression equations as shown in Table 4B-5. The slopes range from about one half to nearly two with R² values that are often below 0.9 and as low as 0.4.

Figure 4B-7 Box and whisker plot of the percent difference in calculated hourly PM_{2.5} light extinction between approaches I and A by urban area.



⁶ The value of 4.35 m²/g has no derivation of particular importance. It was chosen by EPA staff as an initial estimate of the value that would make the mean PM_{2.5} light extinction estimated by approach I be the same as in approach B in one particular city.

Figure 4B-8 Scatter plots of hourly calculated PM_{2.5} light extinction for approach I versus approach A for four selected urban areas.

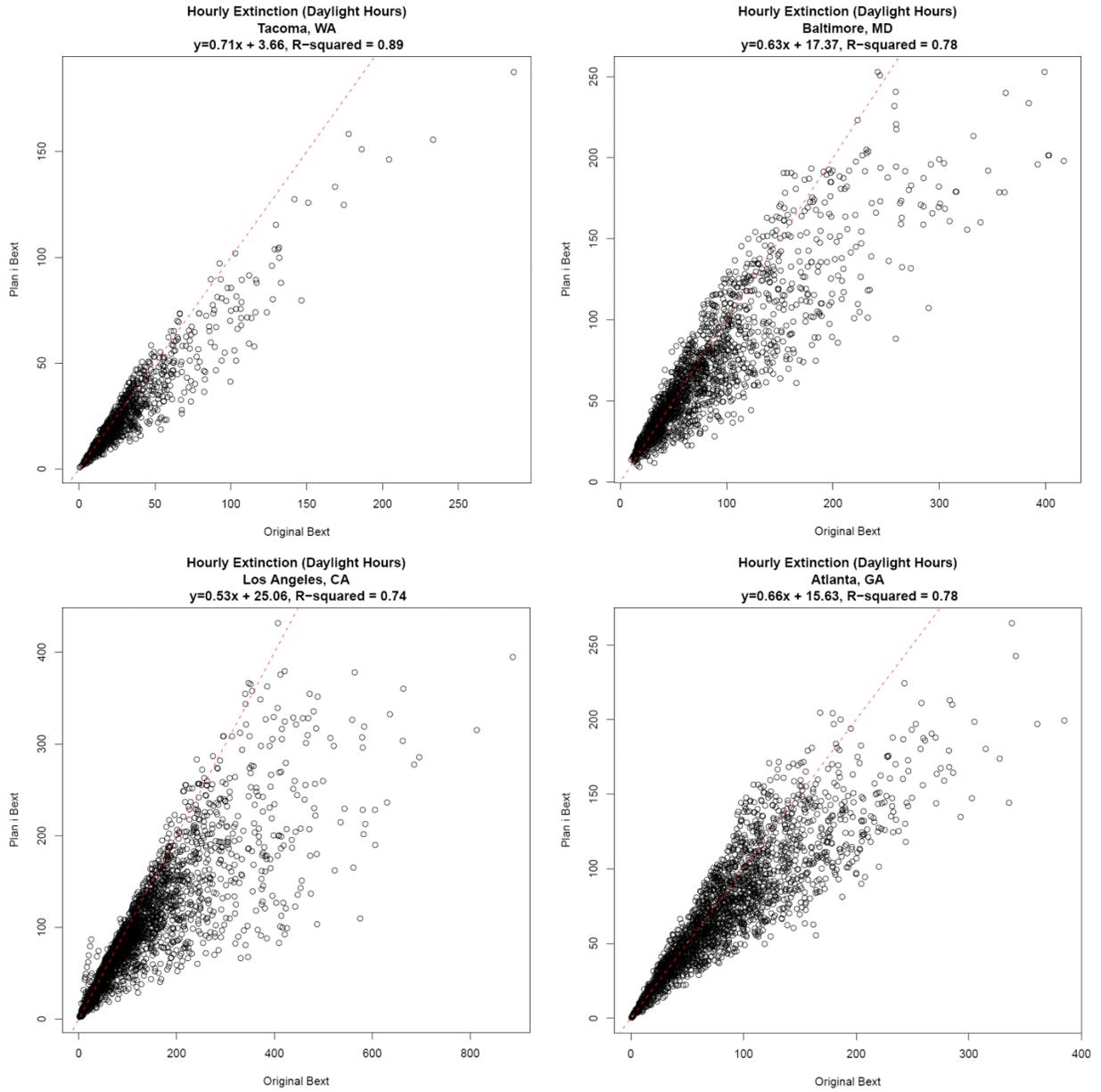


Table 4B-5 Linear regression equation and R² values for relating hourly PM_{2.5} light extinction values calculated using approach I (x in the equation) to those using approach A (y in the equation) by month for 15 urban areas.

Month	1	2	3	4	5	6	7	8	9	10	11	12
Atlanta, GA	y=1.49*x+-6.02; Rsq=0.91	y=1.46*x+-6.61; Rsq=0.85	y=1.25*x+-8.03; Rsq=0.85	y=1.33*x+-11.91; Rsq=0.85	y=1.2*x+-7.04; Rsq=0.77	y=1.06*x+5.99; Rsq=0.65	y=0.87*x+25.82; Rsq=0.55	y=1.06*x+8.1; Rsq=0.62	y=1.21*x+-2.64; Rsq=0.75	y=1.22*x+-2.26; Rsq=0.89	y=1.53*x+-12.49; Rsq=0.91	y=1.56*x+-13.07; Rsq=0.92
Baltimore, MD	y=1.28*x+-8.03; Rsq=0.88	y=1.65*x+-31.13; Rsq=0.84	y=1.47*x+-17.1; Rsq=0.79	y=1.03*x+0.11; Rsq=0.79	y=1.05*x+6.54; Rsq=0.62	y=1.23*x+-6.26; Rsq=0.7	y=1.1*x+-1.47; Rsq=0.74	y=1.09*x+7.77; Rsq=0.85	y=1.33*x+-8.8; Rsq=0.81	y=1.49*x+-7.7; Rsq=0.72	y=1.69*x+-15.94; Rsq=0.94	y=1.34*x+-5.56; Rsq=0.71
Birmingham, AL	y=1.29*x+-5.86; Rsq=0.8	y=1.31*x+-7.3; Rsq=0.85	y=1.11*x+-5.08; Rsq=0.89	y=1.05*x+-2.93; Rsq=0.85	y=1.06*x+-6.34; Rsq=0.9	y=1*x+-2.72; Rsq=0.82	y=1.09*x+-0.42; Rsq=0.77	y=0.95*x+2.44; Rsq=0.77	y=1.16*x+-13.85; Rsq=0.8	y=1.24*x+-7.38; Rsq=0.87	y=1.17*x+-5; Rsq=0.89	y=1.2*x+-0.63; Rsq=0.92
Dallas, TX	y=1.69*x+-8.95; Rsq=0.84	y=1.94*x+-21.97; Rsq=0.88	y=1.54*x+-13.47; Rsq=0.79	y=1.34*x+-10.87; Rsq=0.75	y=1.16*x+-2.83; Rsq=0.69	y=0.86*x+3.04; Rsq=0.74	y=0.86*x+3.72; Rsq=0.75	y=0.96*x+-2.36; Rsq=0.7	y=1.04*x+-2.03; Rsq=0.71	y=1.19*x+-2.94; Rsq=0.81	y=1.15*x+-2.83; Rsq=0.8	y=1.68*x+-10.83; Rsq=0.92
Detroit, MI	y=1.18*x+13.01; Rsq=0.7	y=1.26*x+5.26; Rsq=0.84	y=1.12*x+7.51; Rsq=0.66	y=1.17*x+5.53; Rsq=0.85	y=0.79*x+11.4; Rsq=0.82	y=0.78*x+14.2; Rsq=0.84	y=0.99*x+8.98; Rsq=0.73	y=1.22*x+1.21; Rsq=0.55	y=1.22*x+11.05; Rsq=0.72	y=1.42*x+6.44; Rsq=0.79	y=1.7*x+24.85; Rsq=0.91	y=1.14*x+18.66; Rsq=0.76
Fresno, CA	y=1.2*x+2.75; Rsq=0.82	y=1.28*x+1.8; Rsq=0.84	y=1.46*x+0.46; Rsq=0.83	y=1.29*x+2.15; Rsq=0.66	y=0.98*x+4.24; Rsq=0.54	y=1.01*x+1.51; Rsq=0.68	y=0.94*x+1.1; Rsq=0.93	y=1.13*x+5.1; Rsq=0.73	y=1.24*x+4.61; Rsq=0.73	y=1.63*x+8.37; Rsq=0.7	y=1.23*x+9.65; Rsq=0.81	y=1.31*x+3.3; Rsq=0.87
Houston, TX	y=1.29*x+-2.32; Rsq=0.87	y=1.35*x+-6.35; Rsq=0.81	y=1.23*x+-7.63; Rsq=0.85	y=1.38*x+-18.36; Rsq=0.82	y=1.33*x+-19.81; Rsq=0.76	y=1.07*x+-2.13; Rsq=0.77	y=0.54*x+20.69; Rsq=0.57	y=0.75*x+11.34; Rsq=0.43	y=1.21*x+-12.05; Rsq=0.82	y=1.3*x+-10.7; Rsq=0.95	y=1.21*x+-7.66; Rsq=0.86	y=1.32*x+-4.95; Rsq=0.88
Los Angeles, CA	y=1.45*x+-6.19; Rsq=0.87	y=1.22*x+3.21; Rsq=0.85	y=1.26*x+6.69; Rsq=0.79	y=1.43*x+-6.52; Rsq=0.78	y=1.41*x+-3.21; Rsq=0.78	y=1.72*x+-25.83; Rsq=0.56	y=1.24*x+4.44; Rsq=0.57	y=1.59*x+-21.91; Rsq=0.49	y=1.42*x+-7.87; Rsq=0.71	y=1.46*x+-14.3; Rsq=0.81	y=1.49*x+-12.55; Rsq=0.86	y=1.18*x+6.72; Rsq=0.85
New York, NY	y=1.5*x+-14.28; Rsq=0.89	y=1.16*x+-5.06; Rsq=0.92	y=1.39*x+-5.96; Rsq=0.87	y=1.22*x+-1.7; Rsq=0.89	y=1.2*x+-2.33; Rsq=0.8	y=1.36*x+-9.36; Rsq=0.78	y=1.17*x+0.71; Rsq=0.65	y=1.29*x+-13.03; Rsq=0.79	y=1.37*x+-5.7; Rsq=0.89	y=1.89*x+-21.18; Rsq=0.83	y=1.42*x+-0.51; Rsq=0.92	y=1.3*x+0.85; Rsq=0.86

Philadelp hia, PA	y=1.38*x+- 9.12; Rsq=0.76	y=1.1*x+2. 37; Rsq=0.78	y=1.03*x+ 1.61; Rsq=0.83	y=0.81*x+ 7.81; Rsq=0.68	y=0.88*x+ 4.55; Rsq=0.85	y=0.97*x+3 .07; Rsq=0.78	y=1.03*x+1 .63; Rsq=0.82	y=0.79*x+3 4.86; Rsq=0.69	y=0.8*x+1 0.76; Rsq=0.64	y=1.01*x+ 8.02; Rsq=0.86	y=1.3*x+- 0.08; Rsq=0.85	y=1.06*x+1 0.34; Rsq=0.74
Phoenix, AZ	y=0.83*x+- 2.5; Rsq=0.89	y=0.76*x+- 1.53; Rsq=0.97	y=0.65*x+ 1.46; Rsq=0.96	y=0.56*x+ 2.12; Rsq=0.92	y=0.6*x+1. 7; Rsq=0.88	y=0.62*x+1 .11; Rsq=0.96	y=0.5*x+5. 73; Rsq=0.61	y=0.69*x+0 .51; Rsq=0.97	y=0.6*x+0. 85; Rsq=0.95	y=0.6*x+2. 35; Rsq=0.58	y=0.76*x+- 0.91; Rsq=0.91	y=0.97*x+- 2.04; Rsq=0.89
Pittsburg h, PA	y=1.6*x+- 7.31; Rsq=0.75	y=1.22*x+ 0.13; Rsq=0.76	y=1.19*x+- 2.09; Rsq=0.76	y=1.06*x+ 0.19; Rsq=0.74	y=1.06*x+ 0.15; Rsq=0.56	y=0.97*x+1 1.89; Rsq=0.67	y=1.06*x+1 0.46; Rsq=0.51	y=1.28*x+- 9.71; Rsq=0.74	y=1.18*x+- 2.72; Rsq=0.62	y=1.16*x+ 2.65; Rsq=0.72	y=1.4*x+- 9.05; Rsq=0.82	y=1.47*x+- 6.84; Rsq=0.83
Salt Lake City, UT	y=1.38*x+- 4.78; Rsq=0.93	y=1.25*x+- 0.98; Rsq=0.91	y=1.23*x+- 0.88; Rsq=0.87	y=0.92*x+ 2.06; Rsq=0.75	y=0.73*x+ 4.3; Rsq=0.69	y=0.83*x+2 .89; Rsq=0.79	y=0.9*x+0. 11; Rsq=0.98	y=0.88*x+- 0.16; Rsq=0.96	y=0.79*x+ 5.83; Rsq=0.87	y=1.1*x+2. 23; Rsq=0.82	y=1.06*x+ 2.81; Rsq=0.92	y=1.12*x+9 .19; Rsq=0.94
St Louis,, MO	y=1.5*x+- 0.2; Rsq=0.9	y=1.7*x+- 23.42; Rsq=0.87	y=1.79*x+- 23.76; Rsq=0.83	y=1.29*x+- 5.93; Rsq=0.69	y=1.24*x+- 7.67; Rsq=0.73	y=0.88*x+8 .95; Rsq=0.74	y=1.12*x+- 5.02; Rsq=0.75	y=1.07*x+- 2.69; Rsq=0.72	y=1.03*x+ 2.61; Rsq=0.74	y=1.71*x+- 15.49; Rsq=0.79	y=1.67*x+- 14.71; Rsq=0.94	y=1.75*x+- 15.12; Rsq=0.81
Tacoma, WA	y=1.11*x+2 .25; Rsq=0.98	y=1.51*x+- 2.18; Rsq=0.94	y=1.28*x+ 0.12; Rsq=0.92	y=1.37*x+- 3.6; Rsq=0.86	y=1.15*x+- 1.92; Rsq=0.71	y=1.1*x+2. 17; Rsq=0.63	y=1.03*x+2 .64; Rsq=0.81	y=1.3*x+- 2.74; Rsq=0.68	y=1.07*x+ 2.74; Rsq=0.84	y=1.35*x+- 0.62; Rsq=0.92	y=1.37*x+- 2.65; Rsq=0.98	y=1.17*x+2 .34; Rsq=0.96

Appendix 4C

Assessment of the PM Components Responsible for the Largest Hourly PM₁₀ Light Extinction and PM_{2.5} Mass Hours Selected as Maximum Daily and Using All Hours for 14 Urban Areas

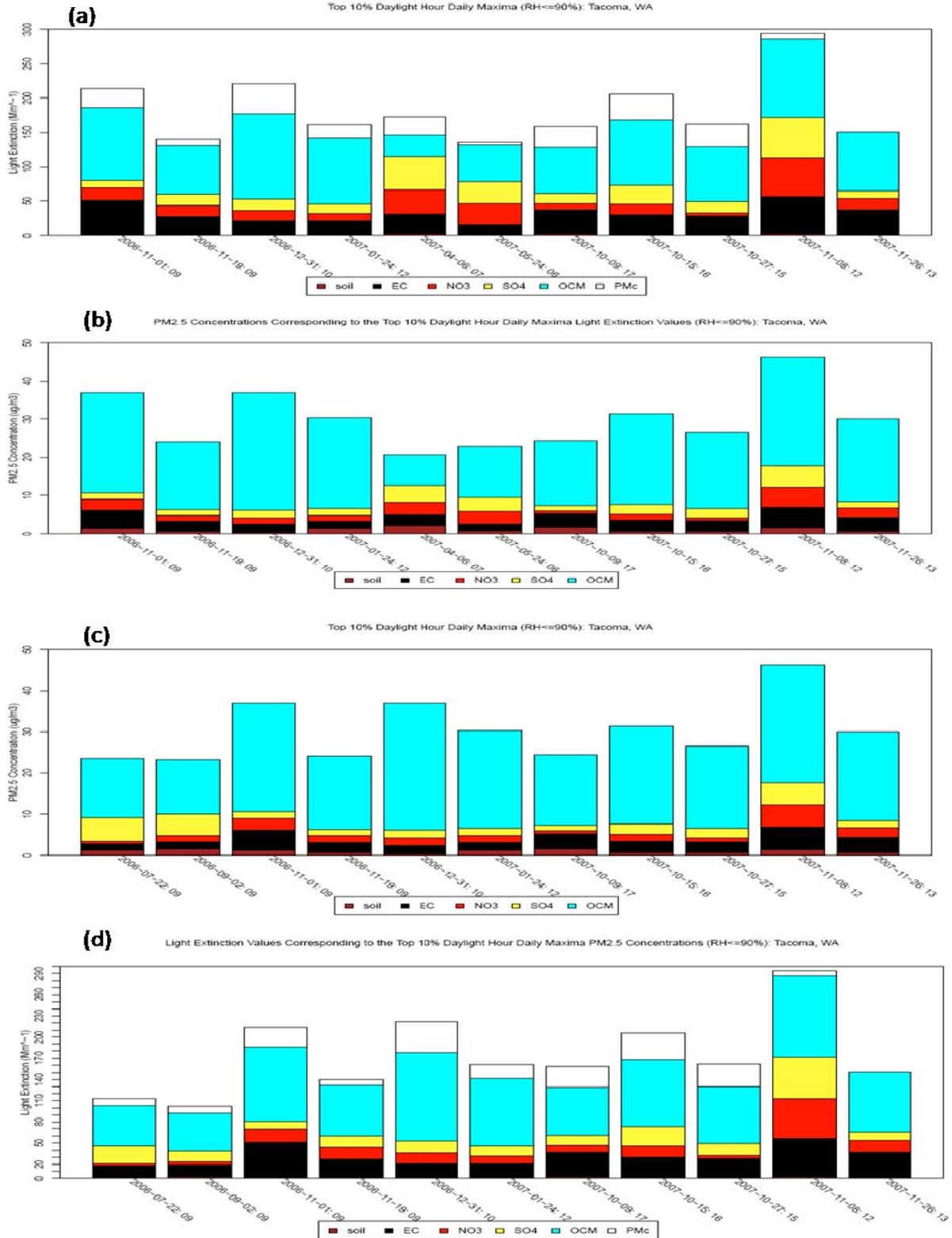
As discussed in the UFVA, light extinction is considered a better indicator of visibility impacts than PM_{2.5} mass concentration. The UFVA also considers two different forms: the maximum daily and all daylight hours for possible use in a secondary PM NAAQS to protect visibility. One way to explore the ramifications of the selection of indicator and form is to assess how much difference these choices make to the types of sources that might need to be controlled to improve visibility conditions. The purpose of this assessment is to provide a better understanding of the dependency of the mix of PM components (i.e., a reflection of the types of sources) responsible for the high haze values on the choice of form and indicator used. This was done using the hourly PM₁₀ light extinction and PM_{2.5} speciated mass composition data set as developed for the UFVA for the 14 urban areas¹. As was demonstrated in the UFVA (section 4.1.4) for PM₁₀ light extinction, the 90th percentile maximum daily daylight hour has design values that are nearly identical to the 98th percentile of all daylight hours for each of the urban areas. In order to reduce the numbers of cases for comparison, only these two forms, which give comparable design values, are included in this assessment.

Stacked bar plots that show the PM species contributions to PM₁₀ light extinction for each hour that is in the top 10% of daily maximum daylight hours of PM₁₀ light extinction and the top 2% of all daylight hours of PM₁₀ light extinction for each of the urban areas are displayed in Figure 3-13 of the UFVA. These show the components that contribute to light extinction for the hours that would need to be reduced in order to meet the 90th percentile maximum daily and 98th percentile of all daylight hours PM₁₀ light extinction. Similar figures were generated for this comparison to display the mass components of each hour that is in the top 10% daily maximum daylight hours and top 2% of all daylight hours for PM_{2.5} mass. To aid making the comparisons, the plots already available in the UFVA are repeated in this appendix and each set of plots is shown both as components contributing to light extinction and those contributing to PM_{2.5} speciated mass concentration. This leads to eight separate stacked bar plots (i.e., 2 indicators x 2 forms x 2 component displays) for each of the 14 urban areas (see Figures 4C1 to 4C14).

¹ Due to the inappropriate coarse mass values developed for the St. Louis dataset, it was excluded in this assessment.

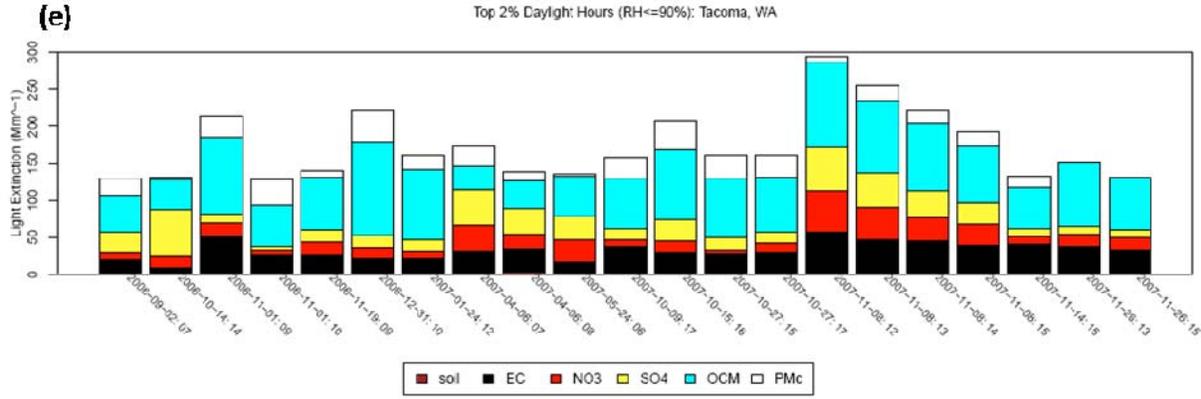
1
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Figure 4C-1 Tacoma PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

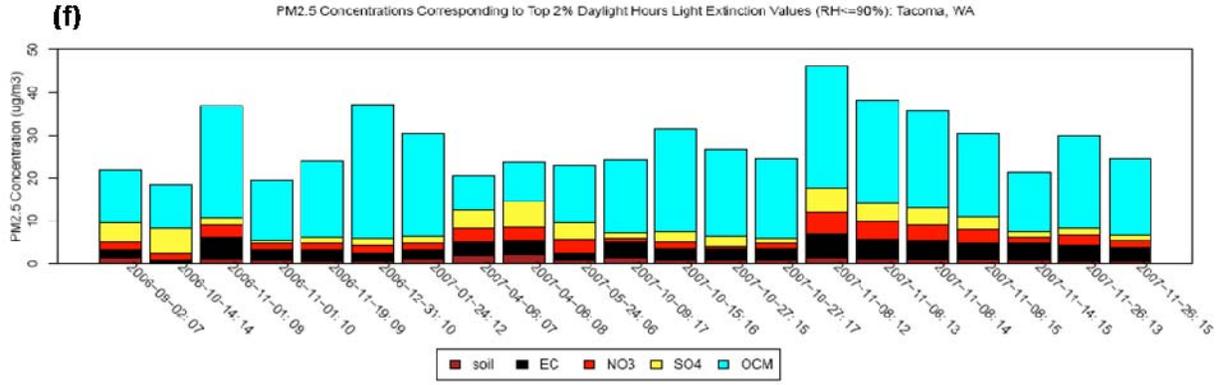


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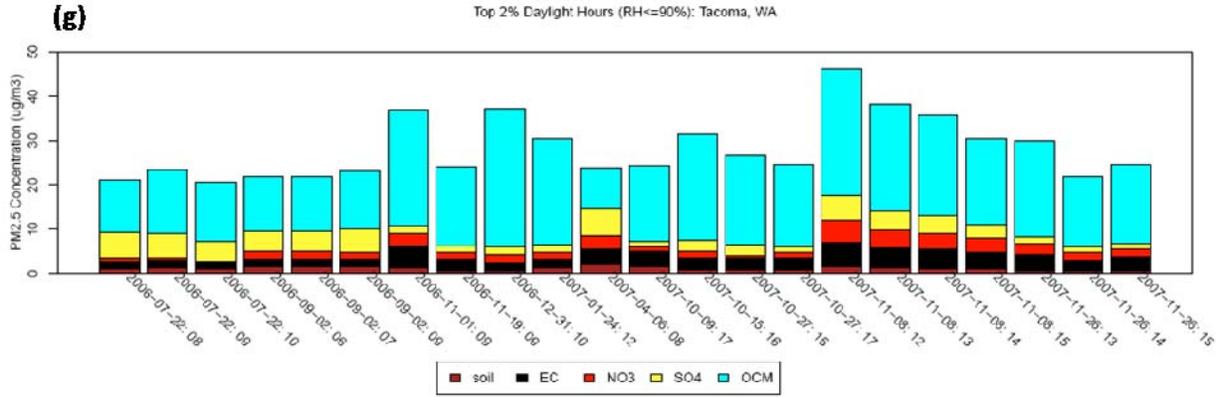
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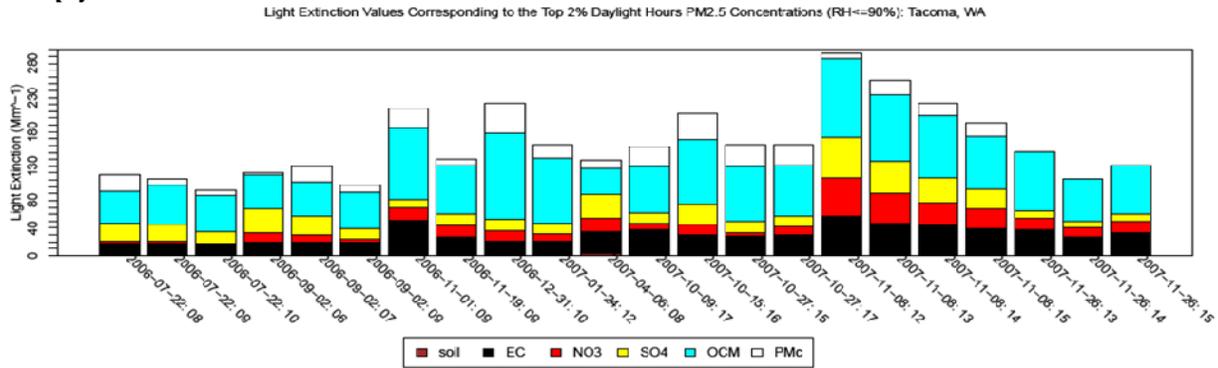
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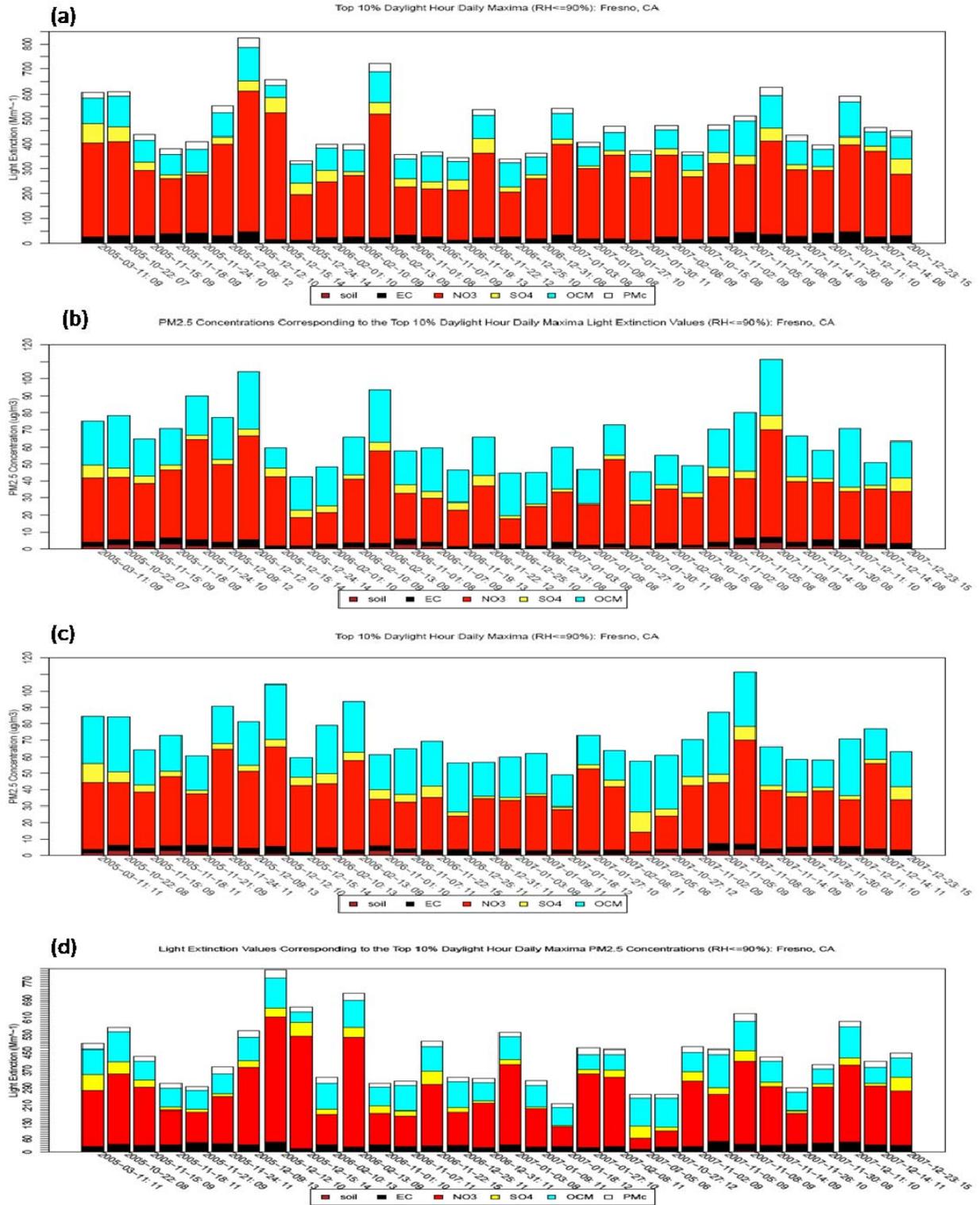
(g)



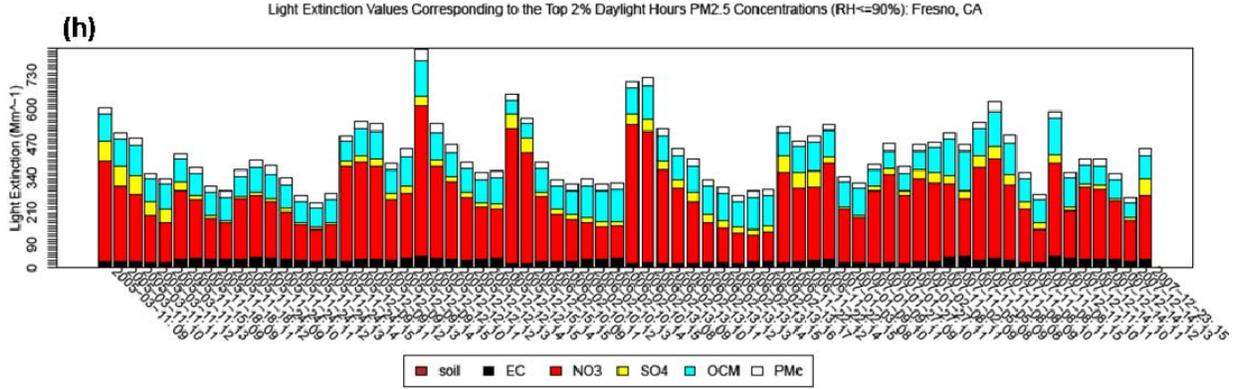
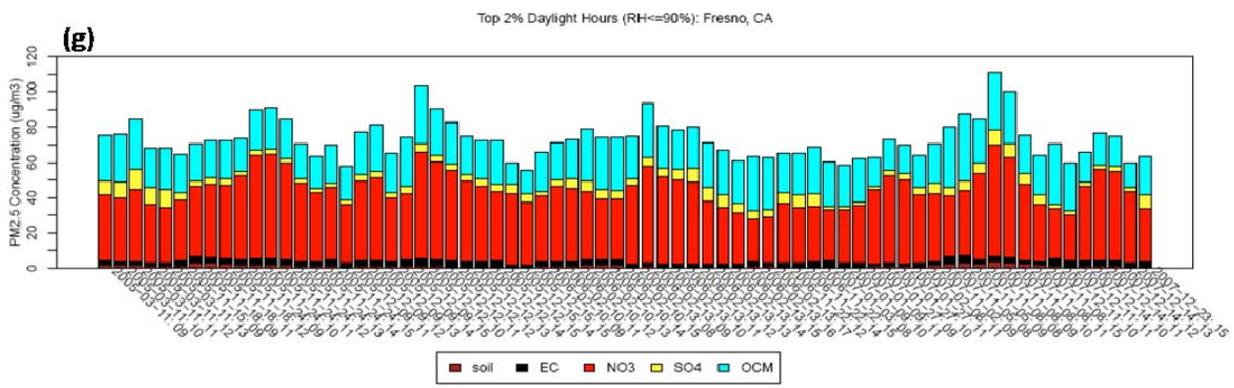
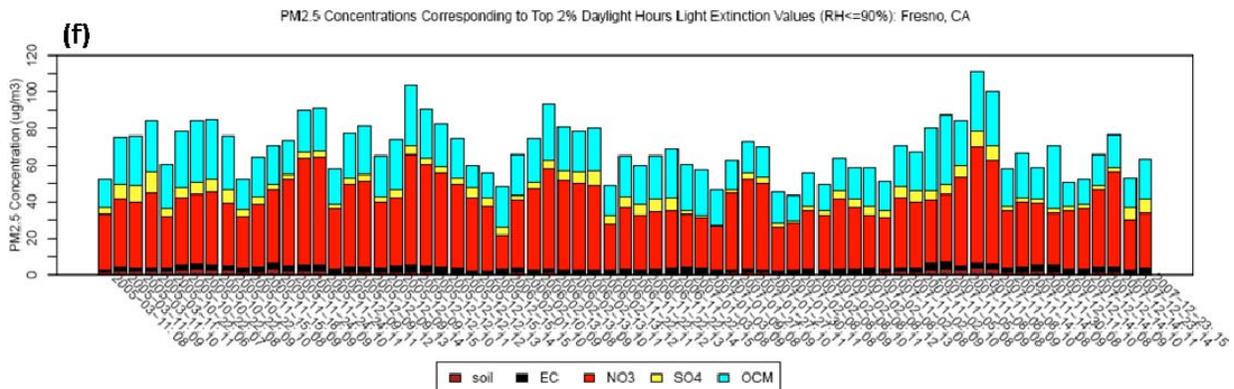
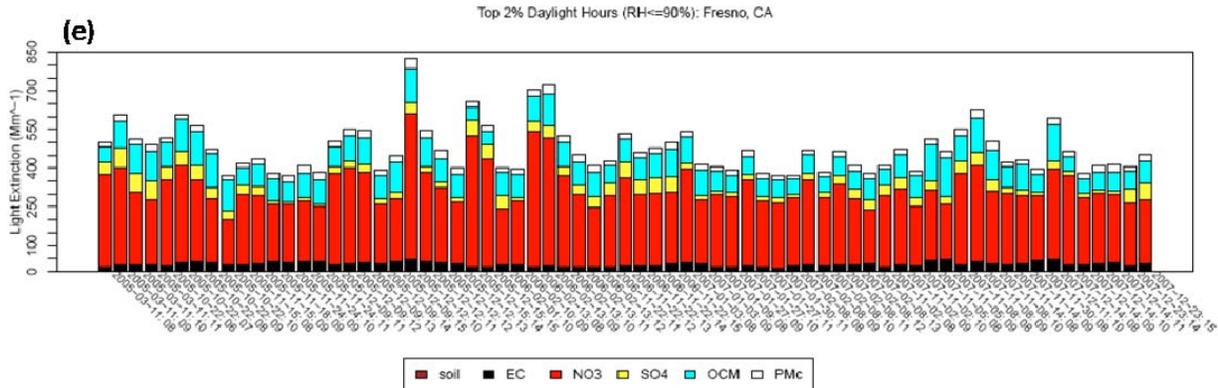
(h)



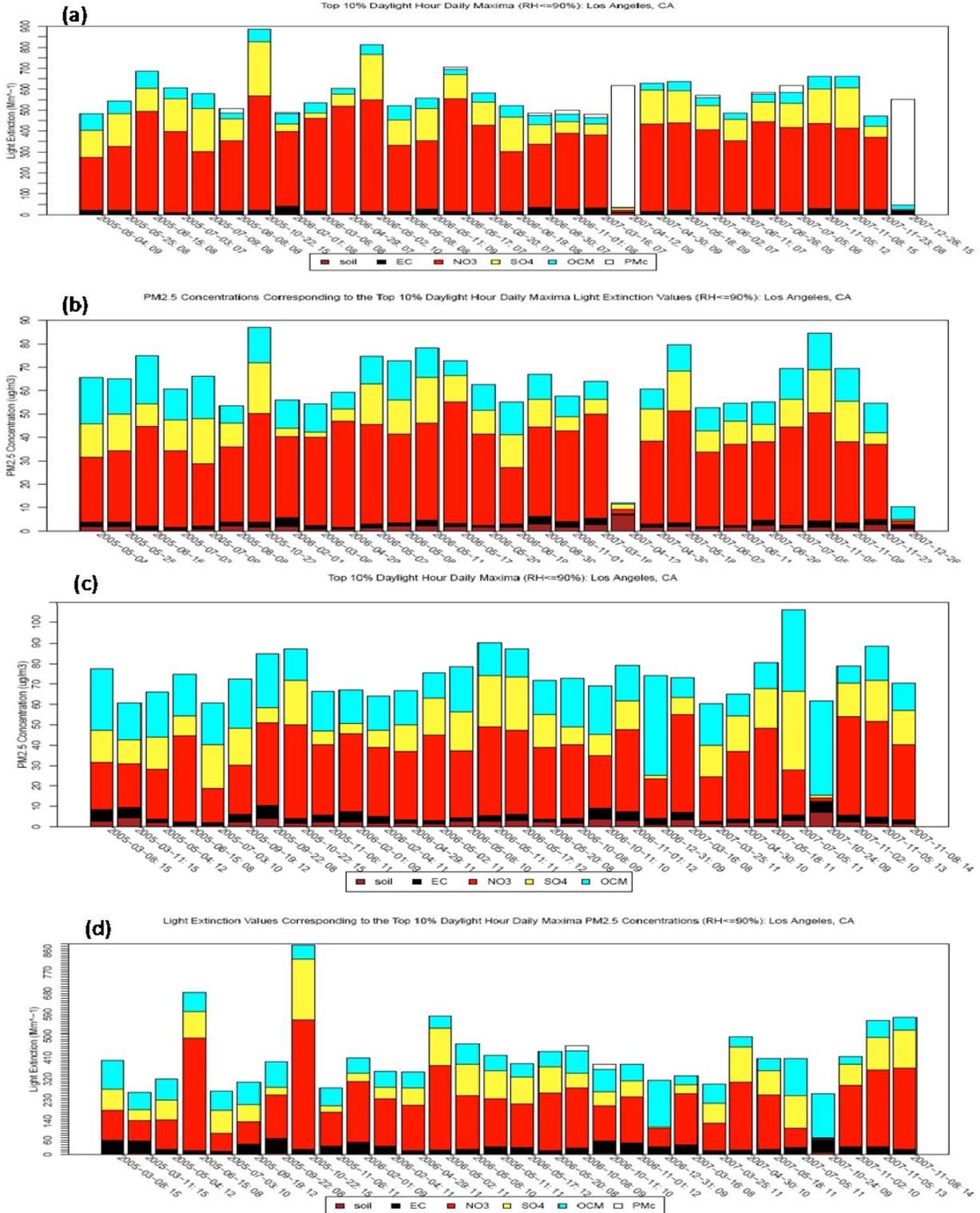
1 **Figure 4C-2 Fresno PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light**
 2 **extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e**
 3 **& f) and for PM_{2.5} mass (g & h).**



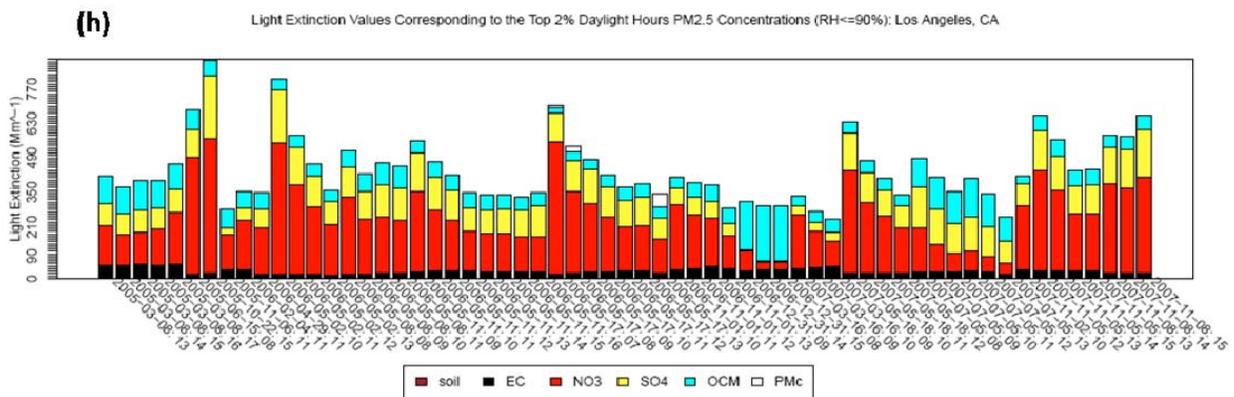
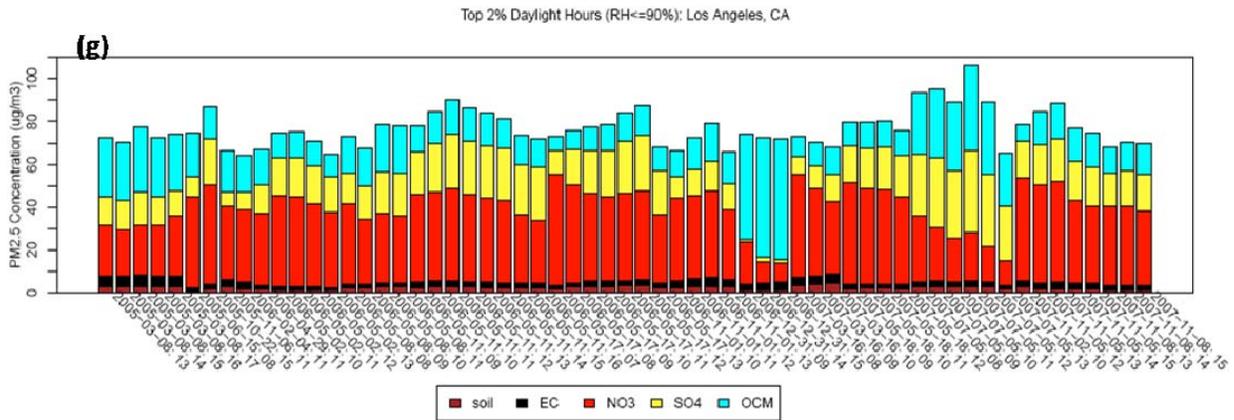
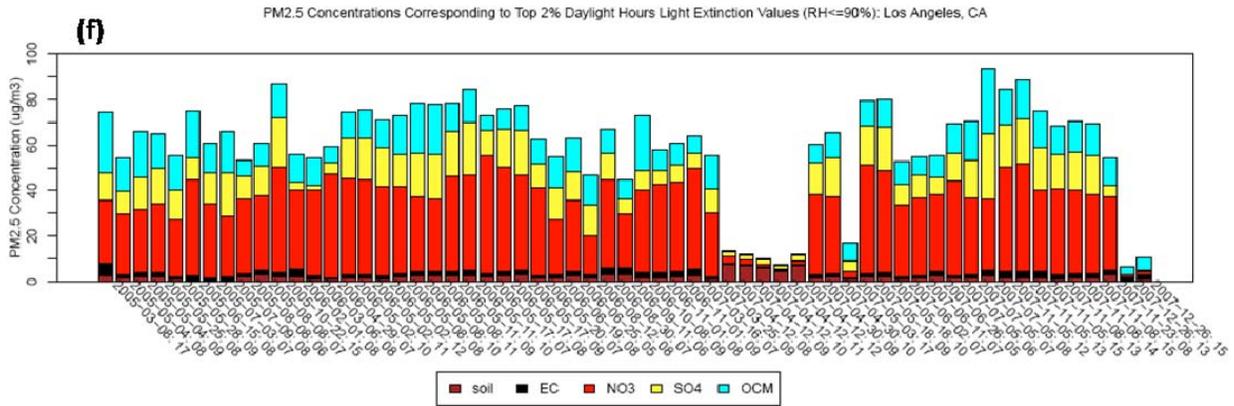
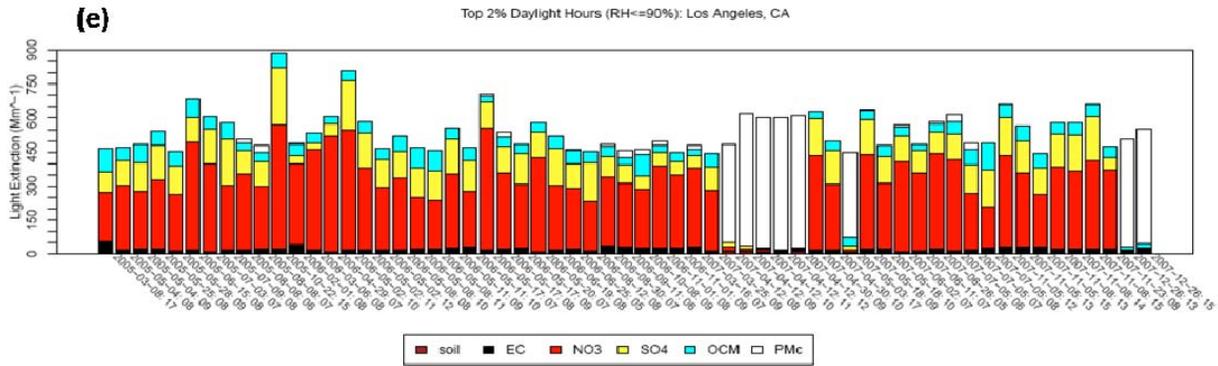
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1 **Figure 4C-3 Los Angeles PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light**
 2 **extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e**
 3 **& f) and for PM_{2.5} mass (g & h).**

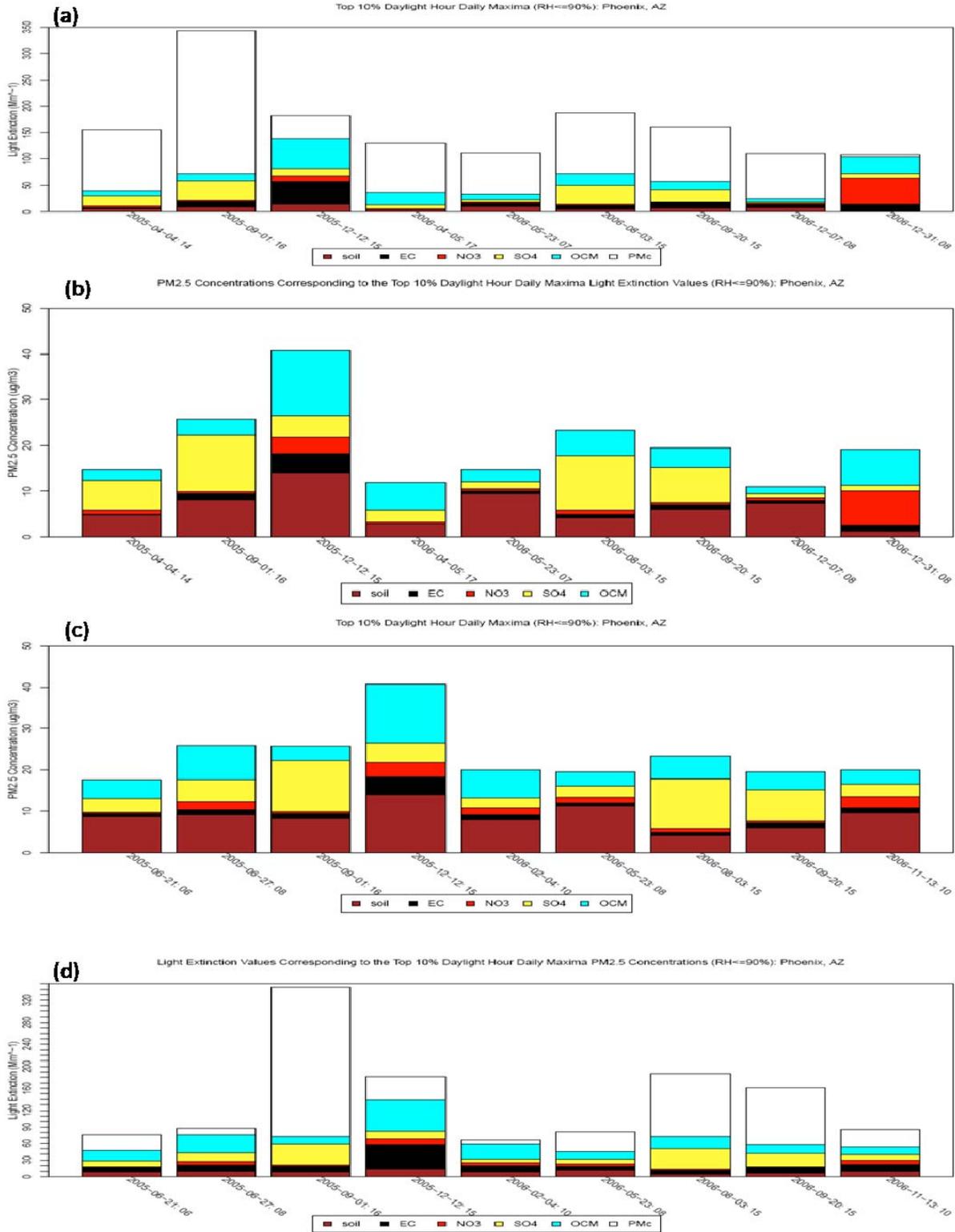


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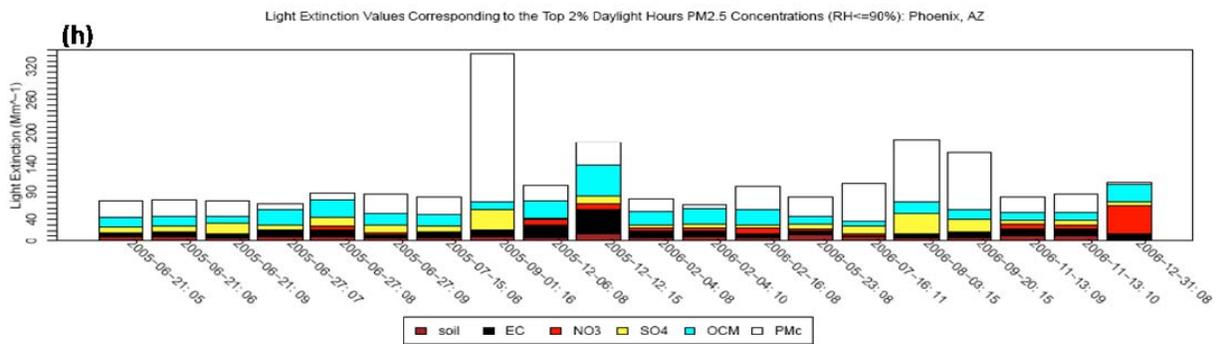
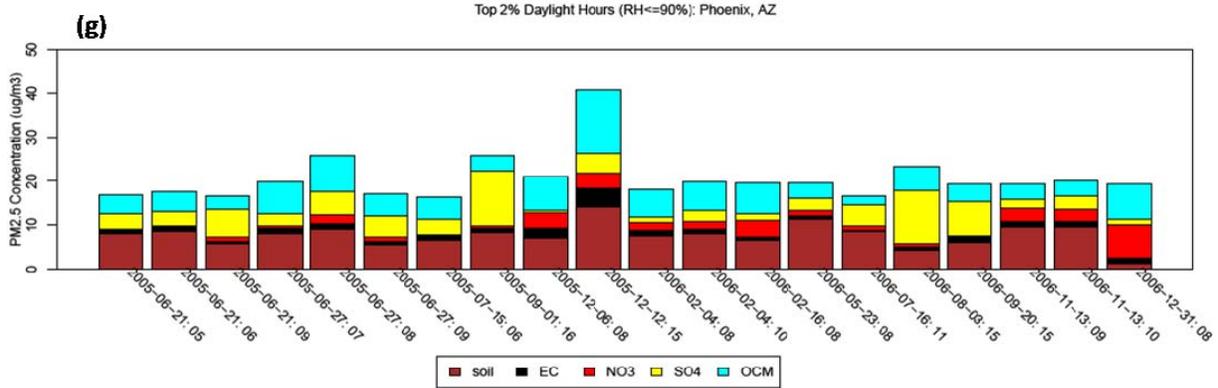
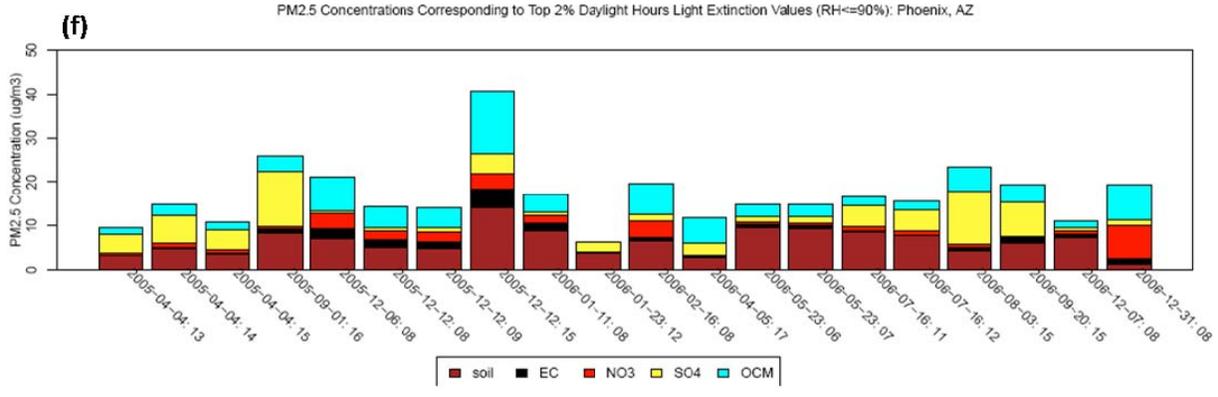
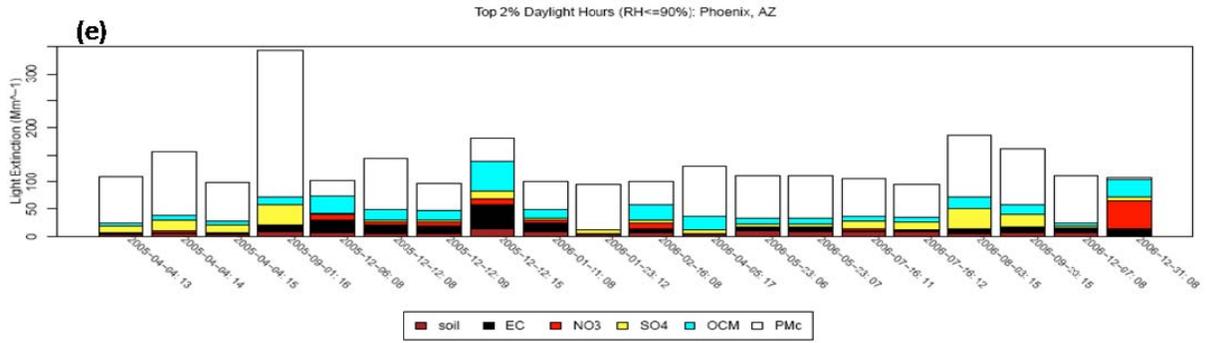


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Figure 4C-4 Phoenix PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

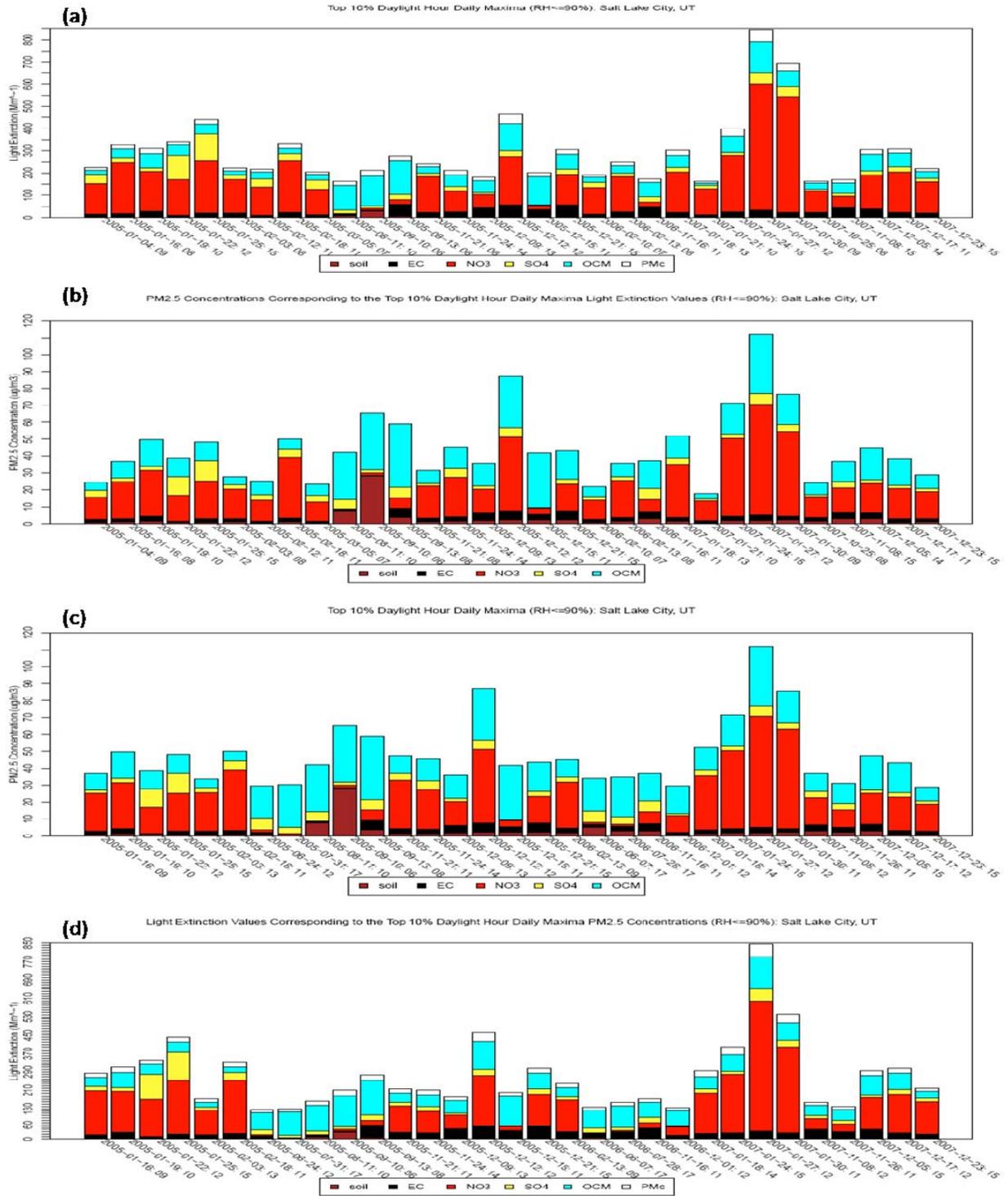


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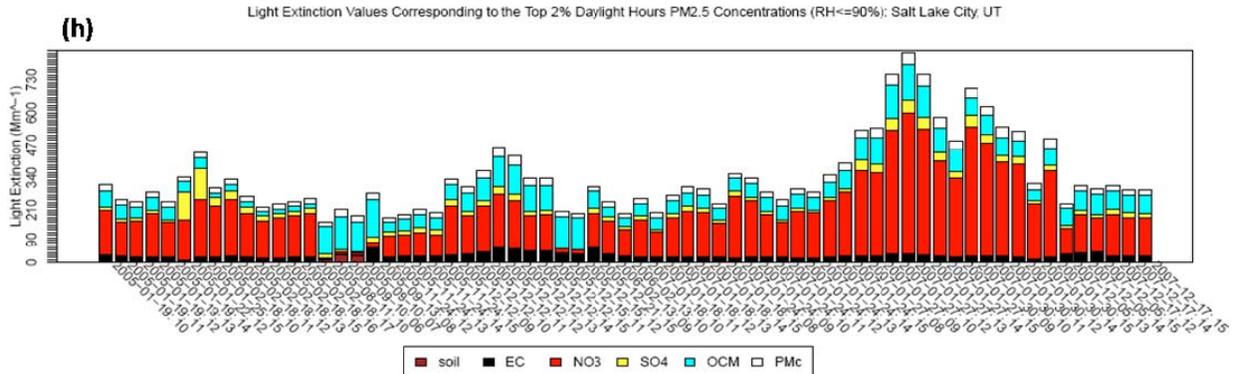
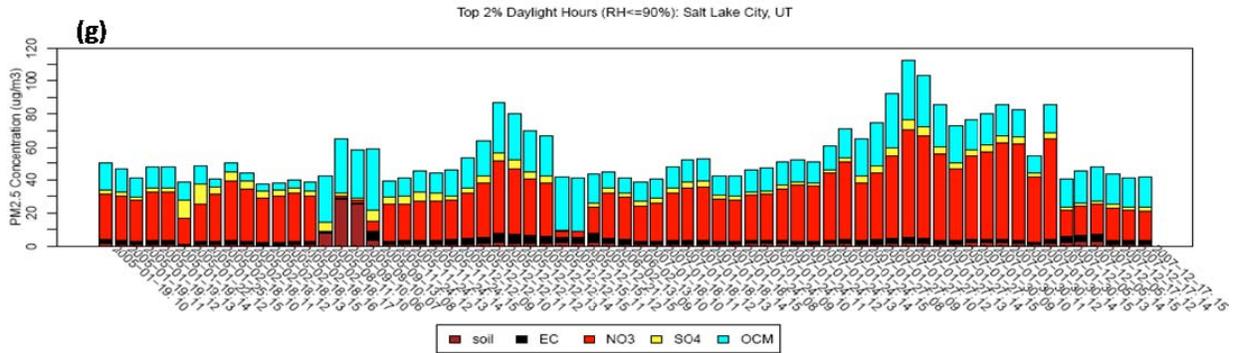
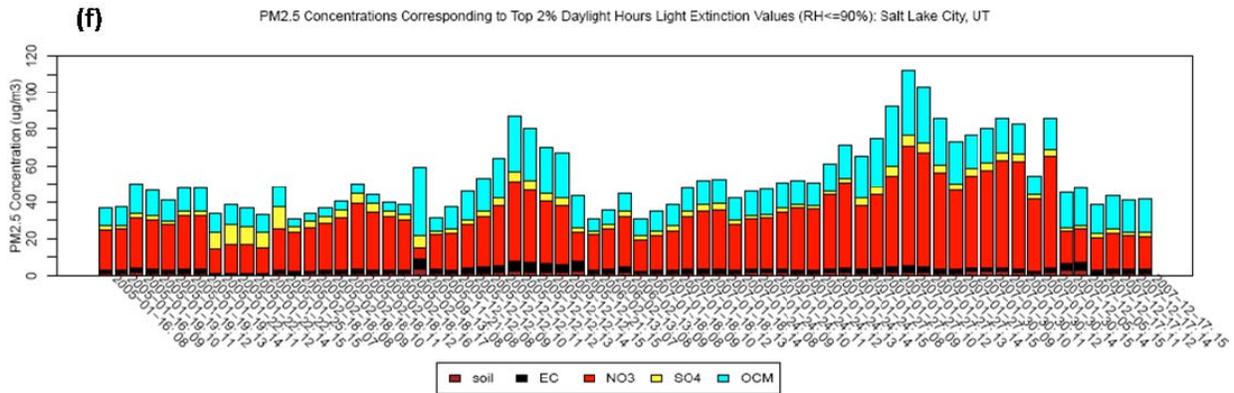
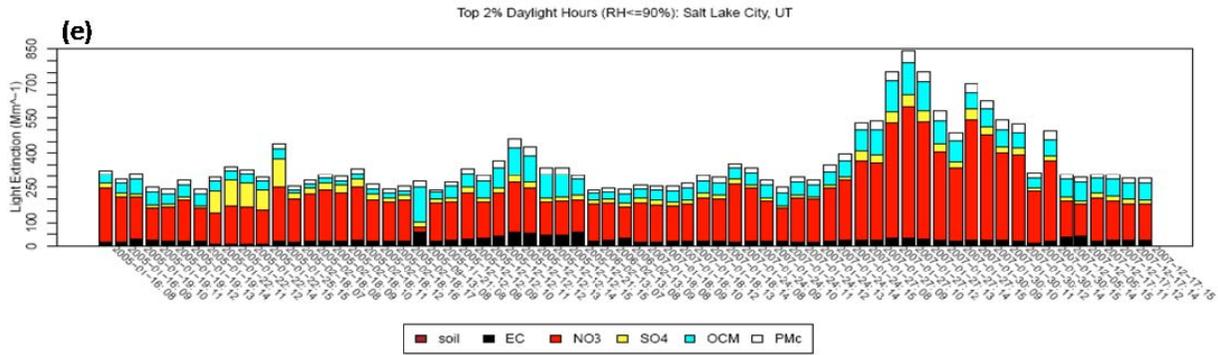


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Figure 4C-5 Salt Lake City PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

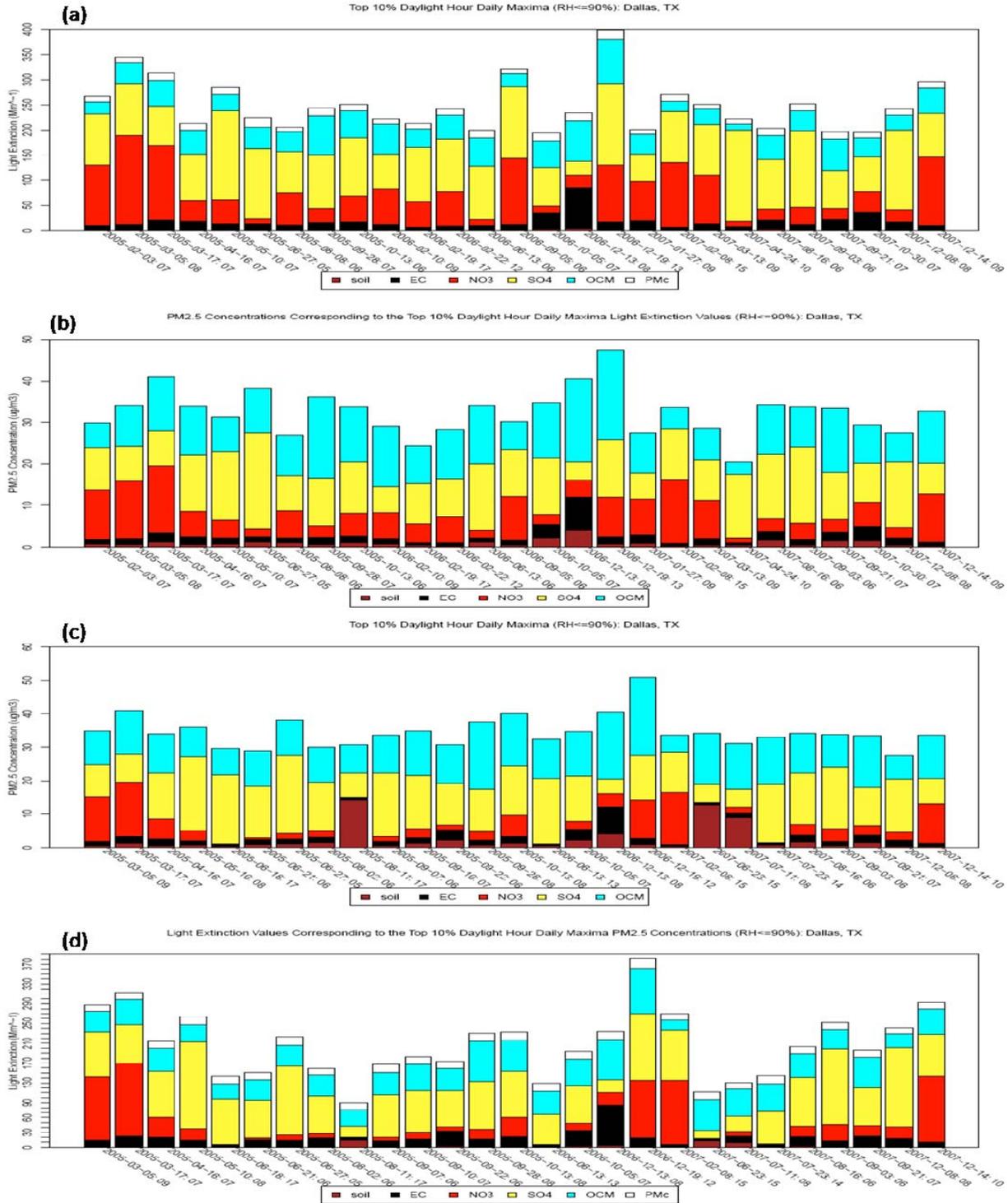


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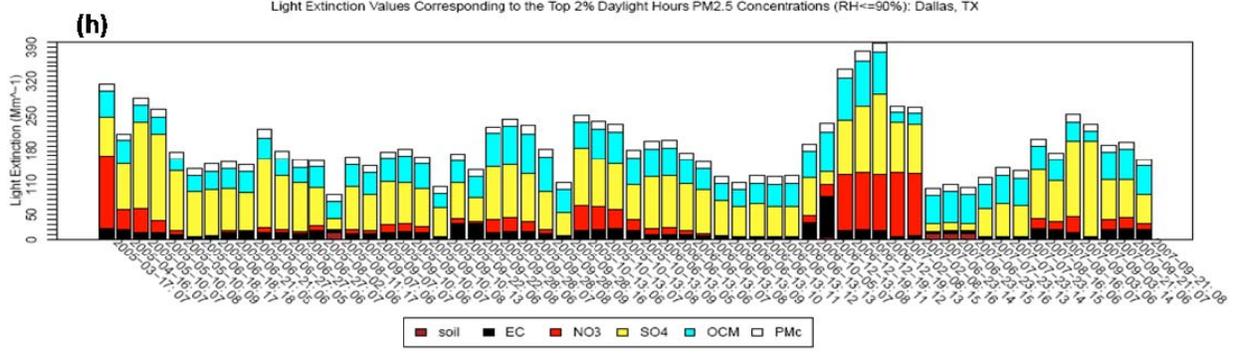
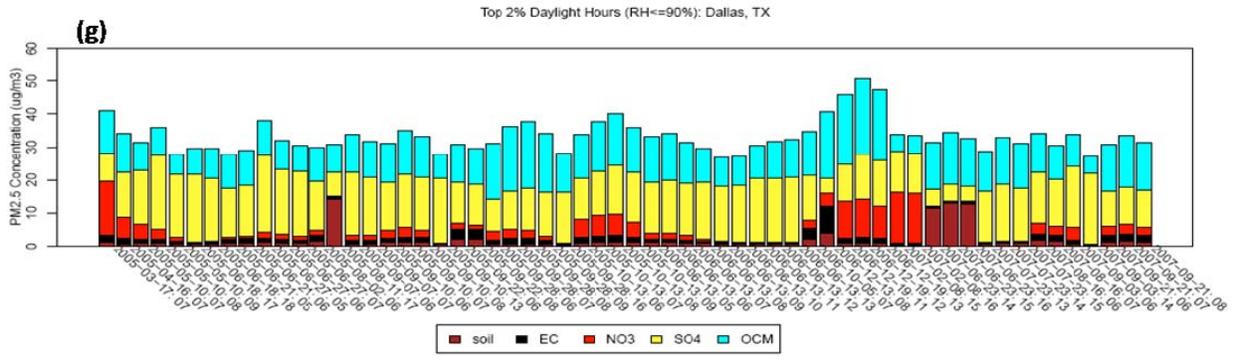
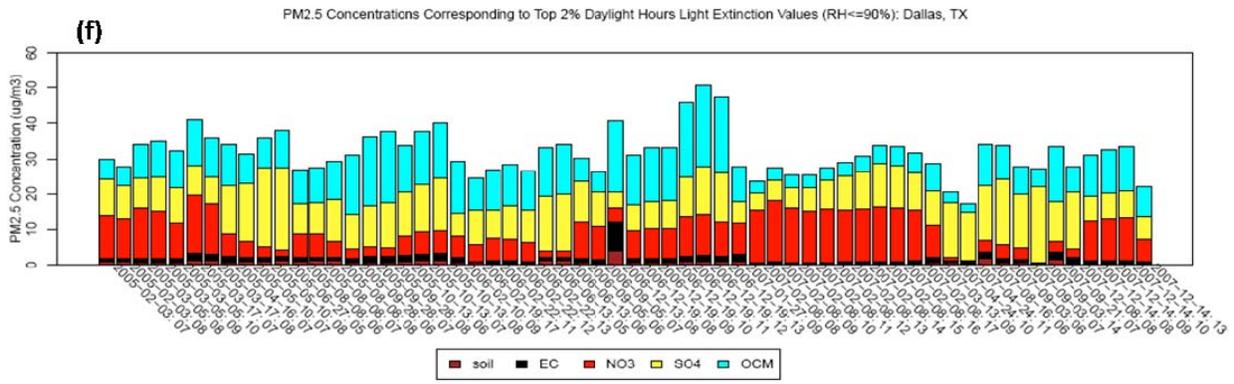
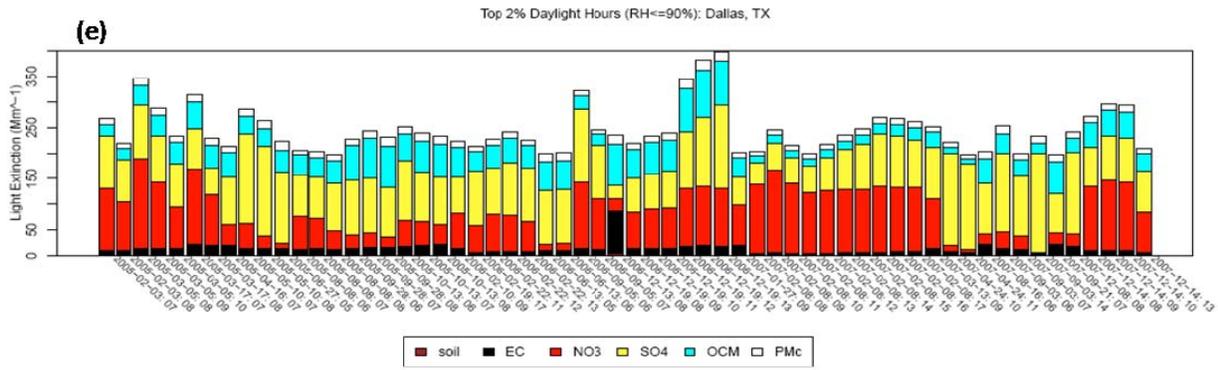


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Figure 4C-6 Dallas PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

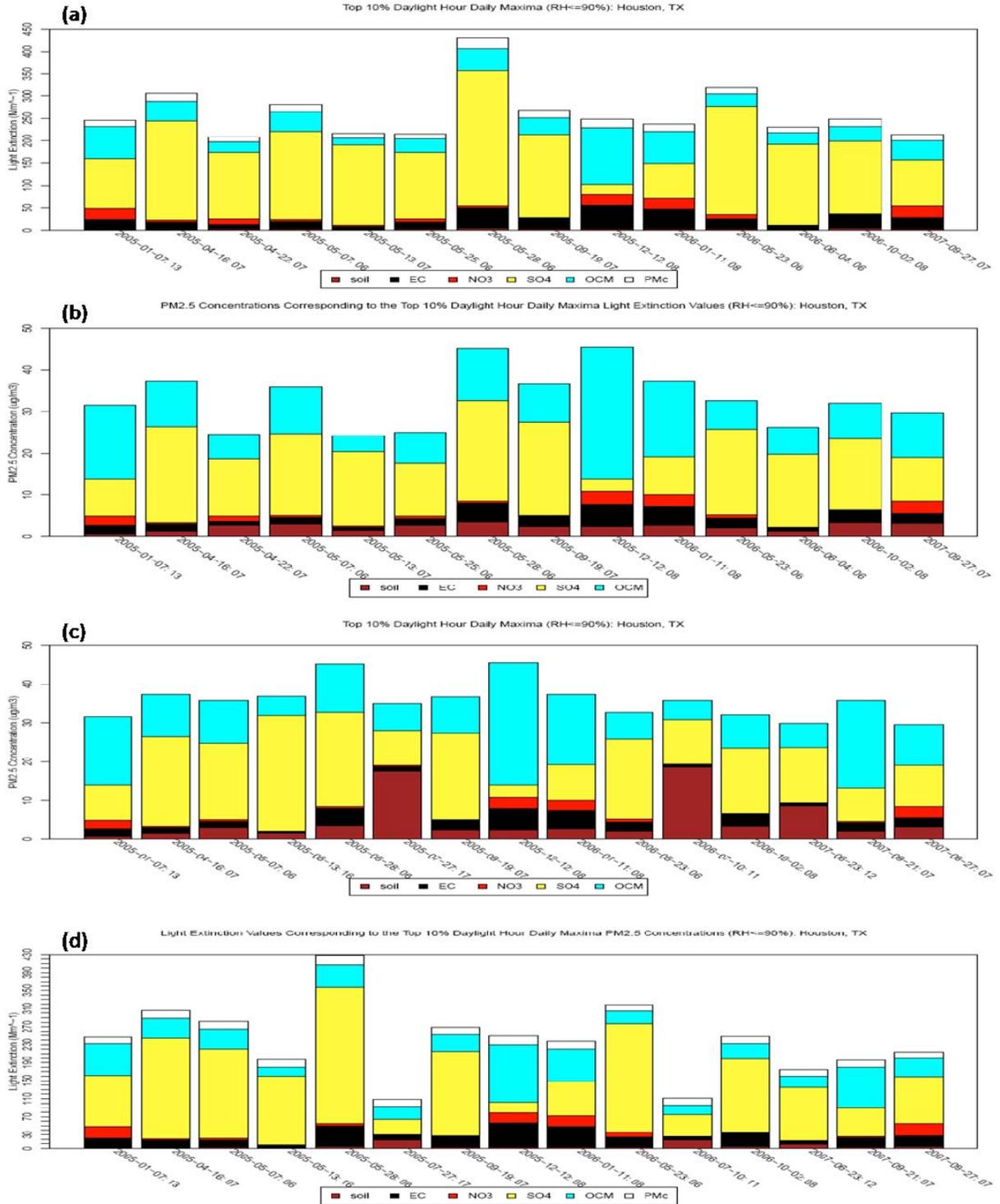


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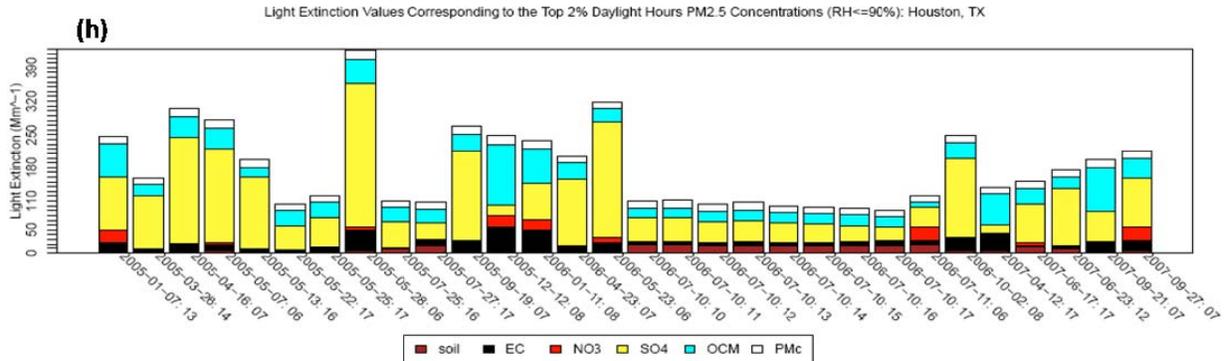
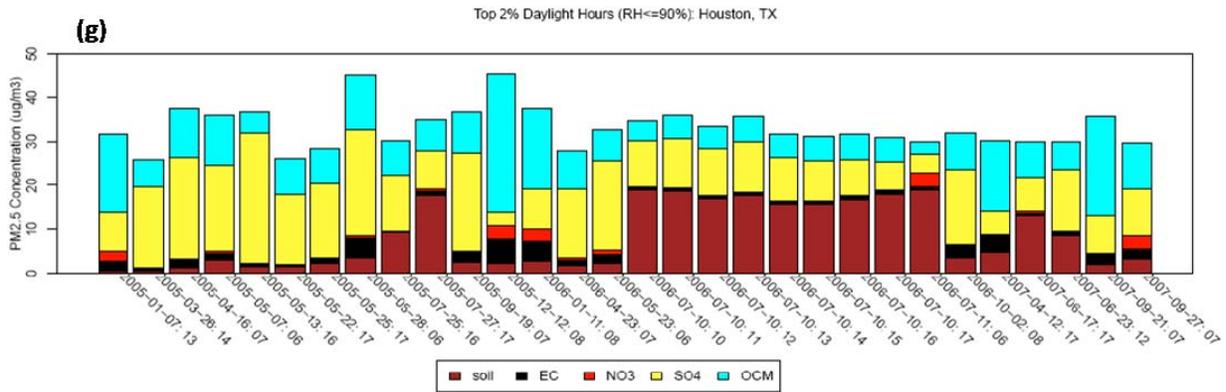
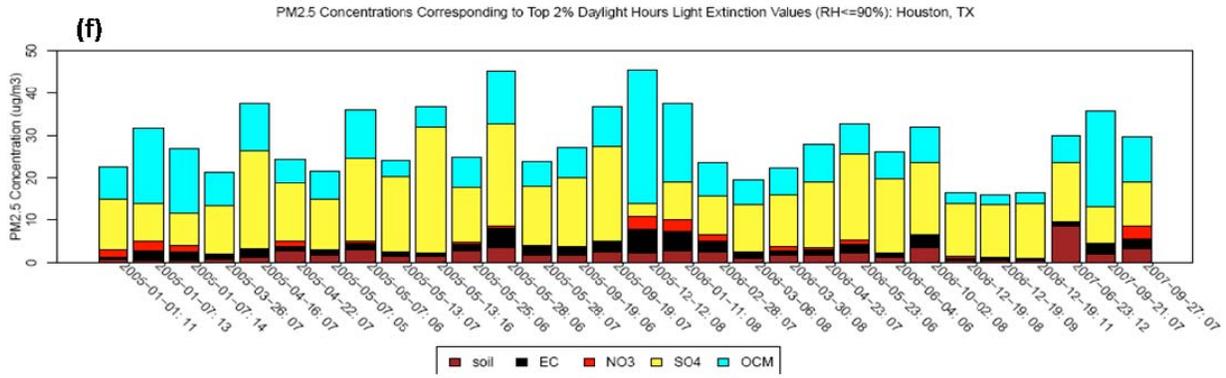
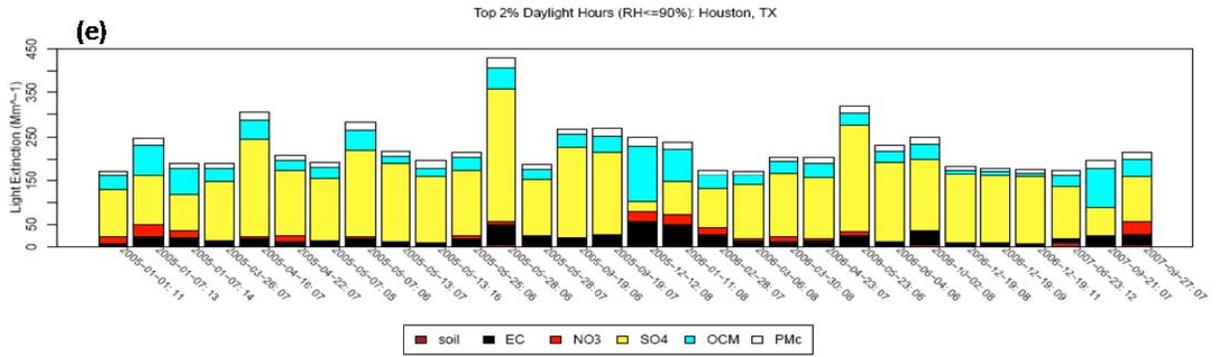


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Figure 4C-7 Houston PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).



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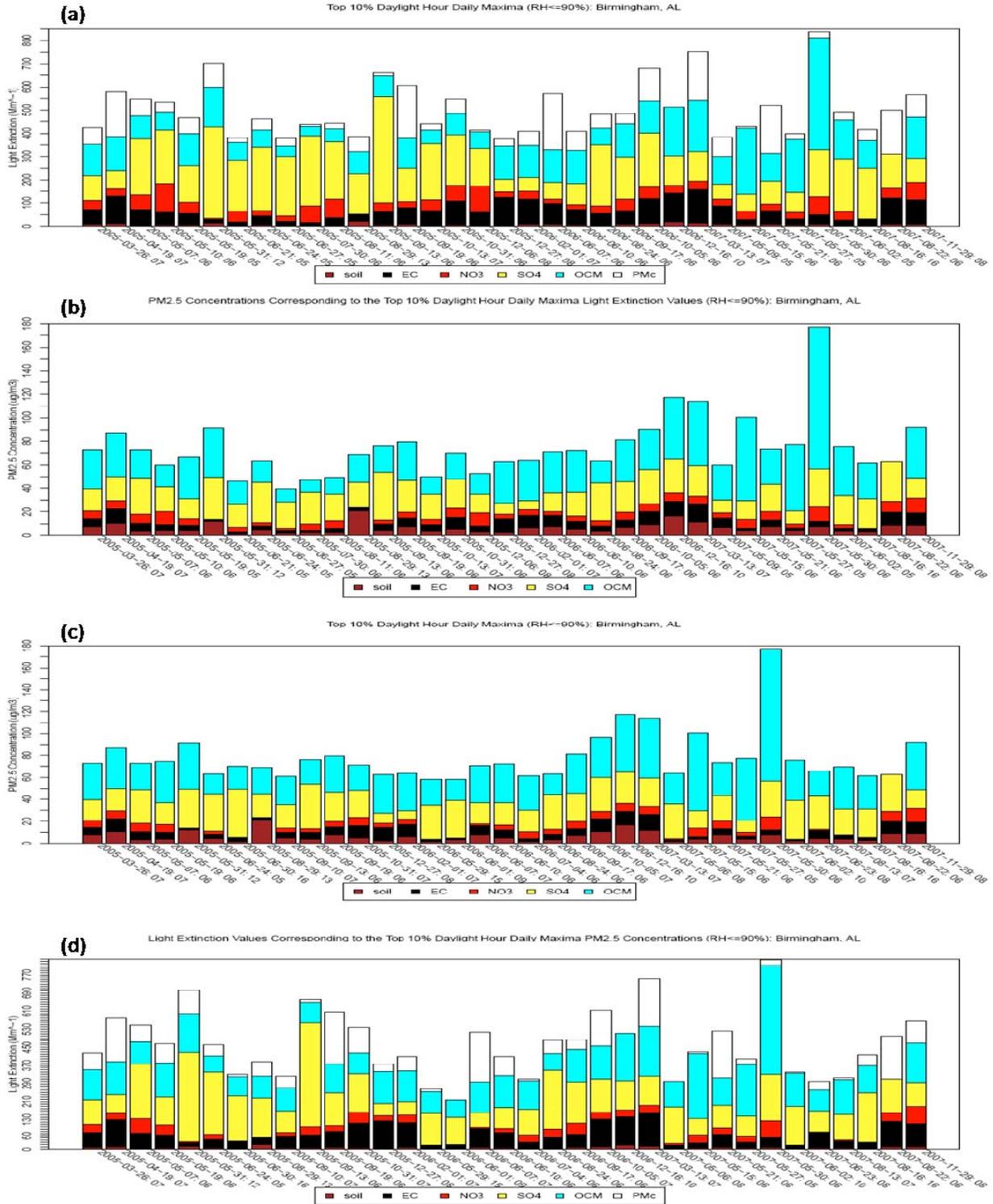
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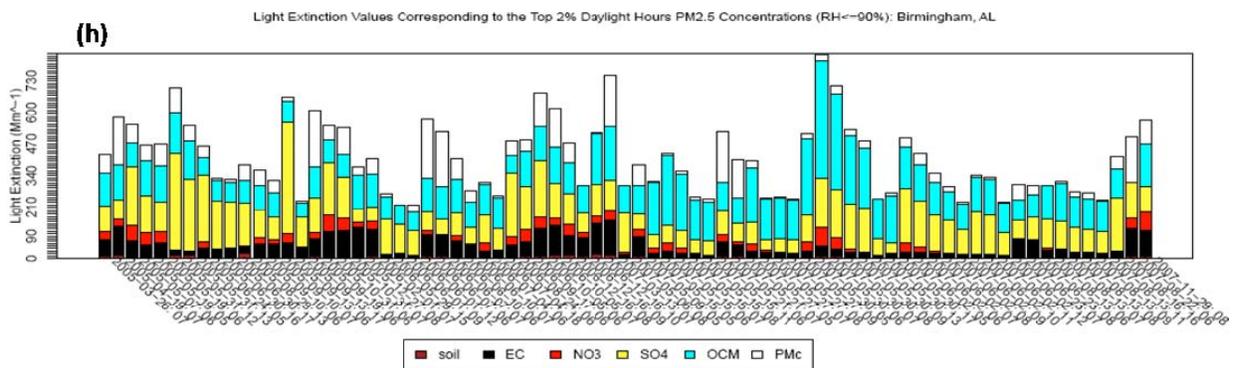
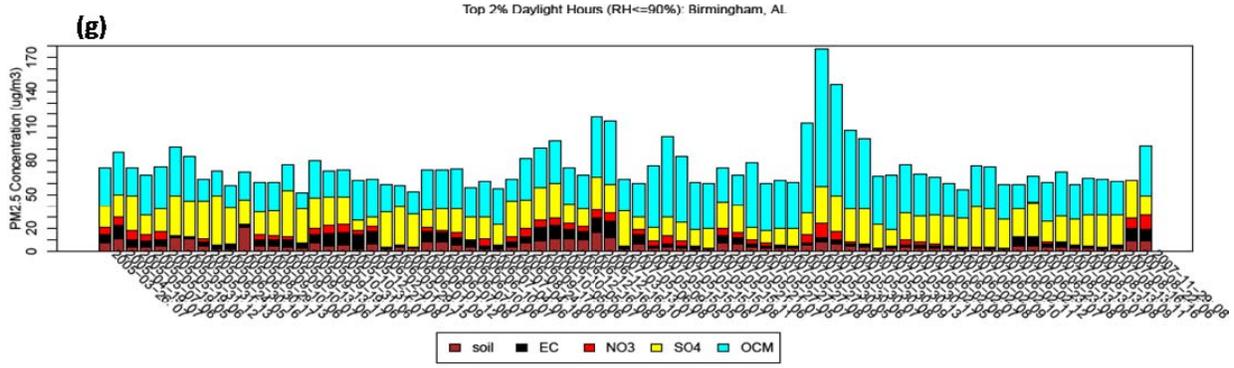
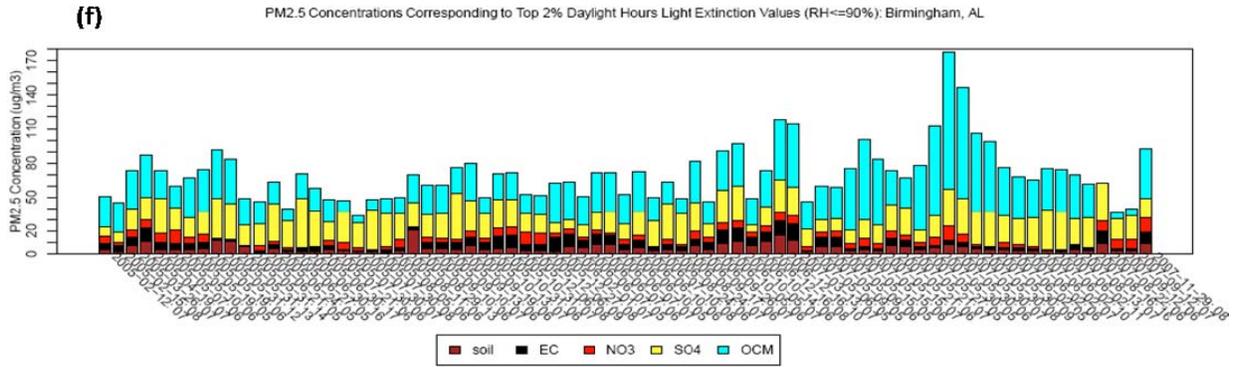
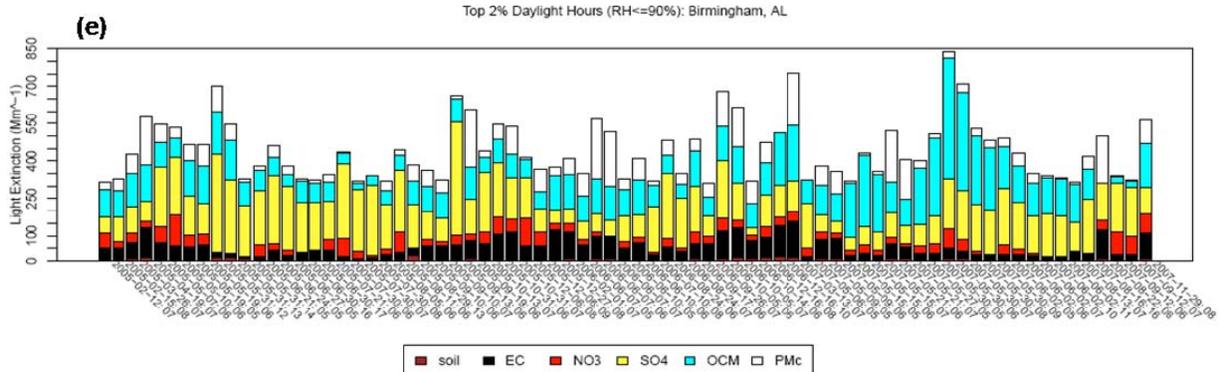
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Figure 4C-8 Birmingham PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).



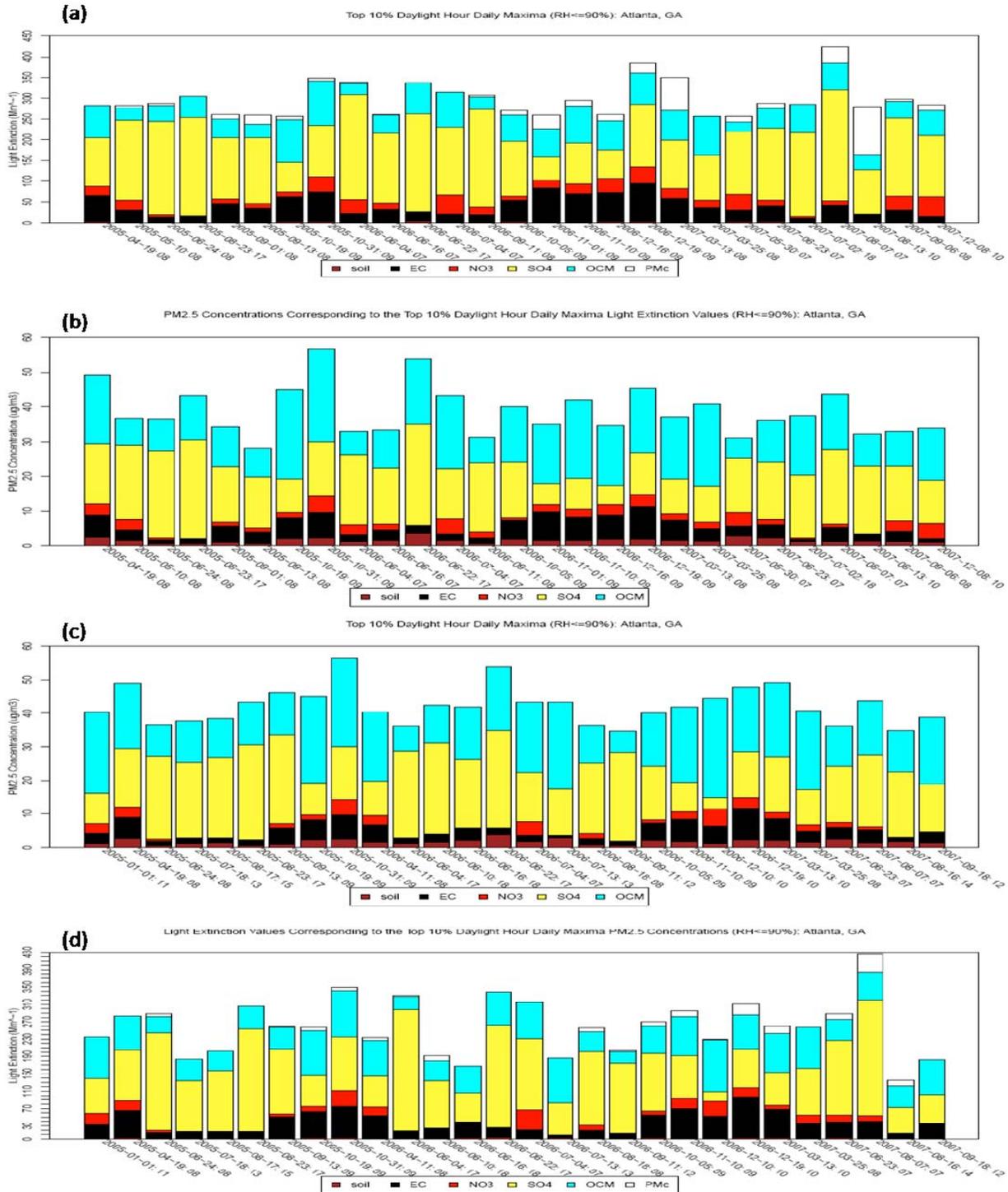
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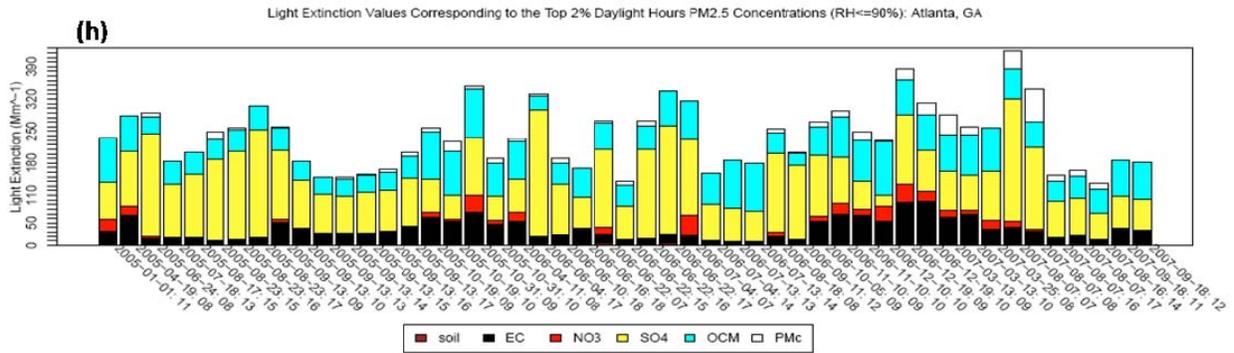
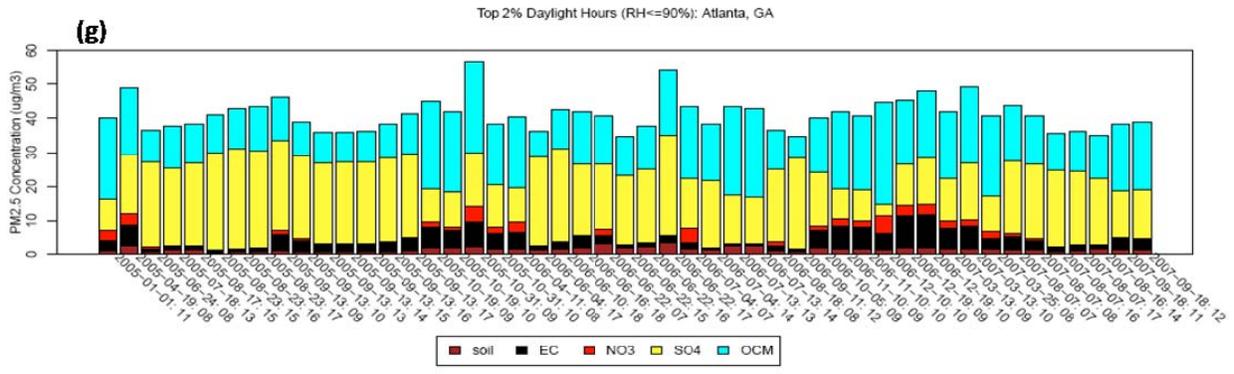
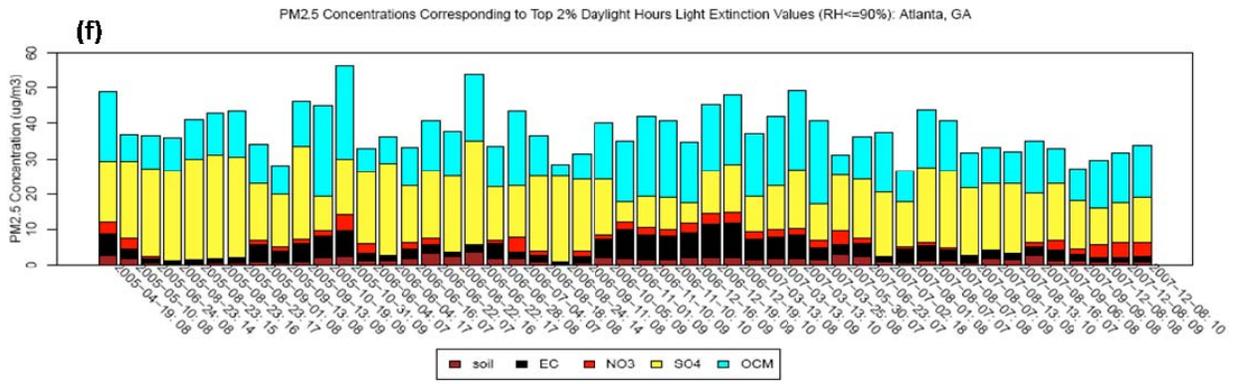
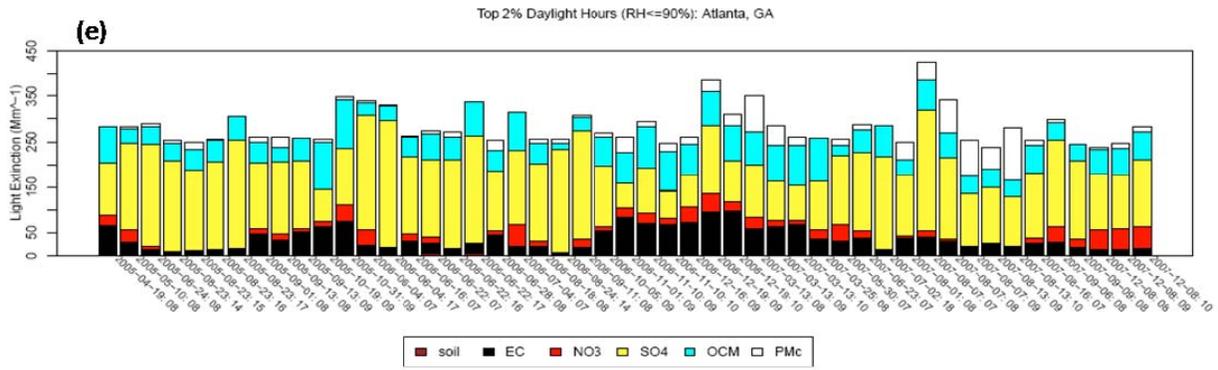
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Figure 4C-9 Atlanta PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

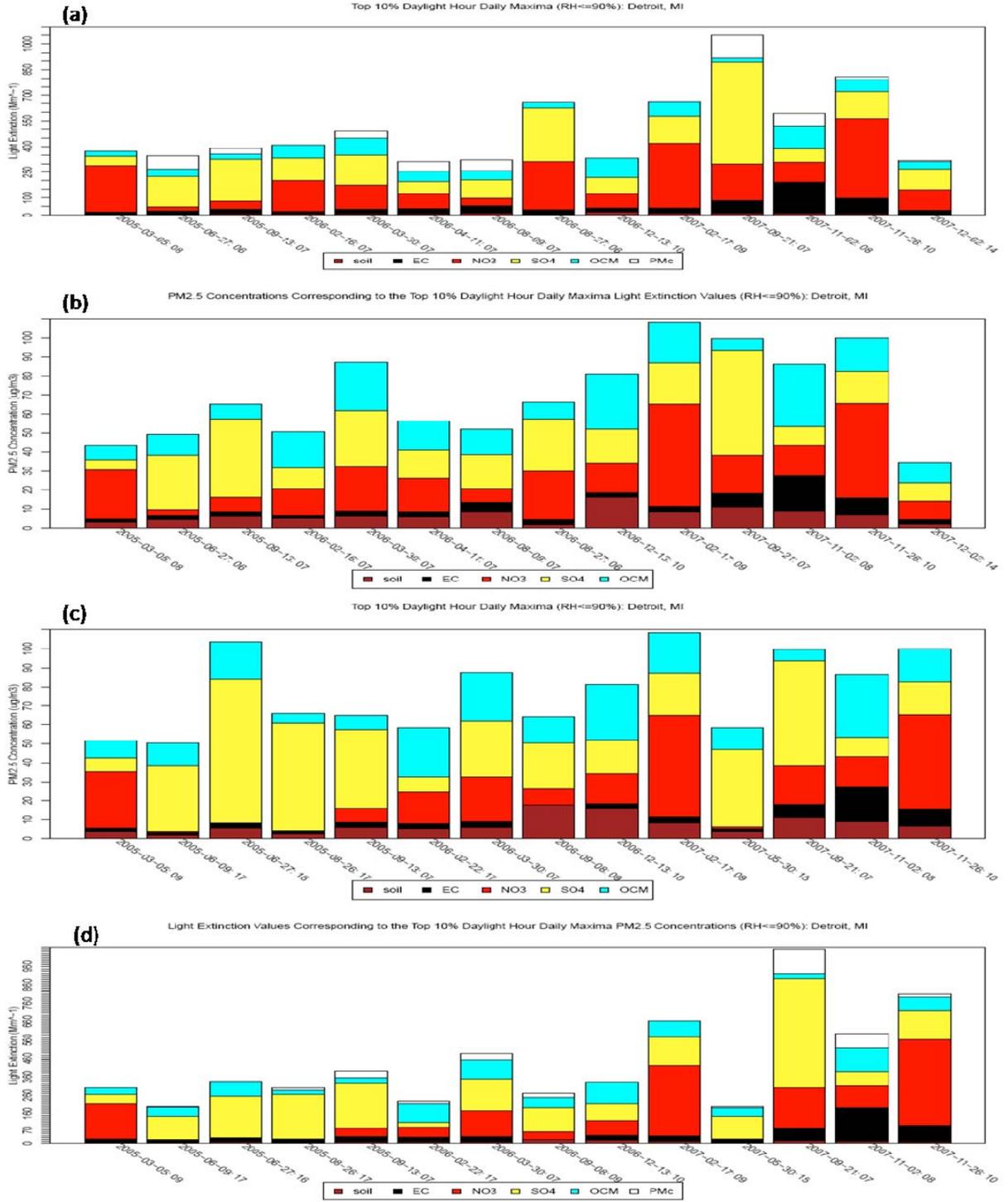


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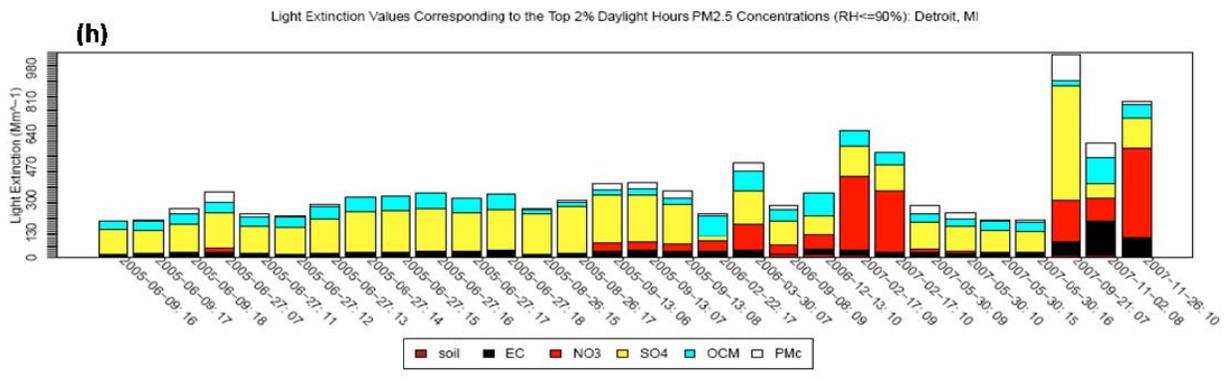
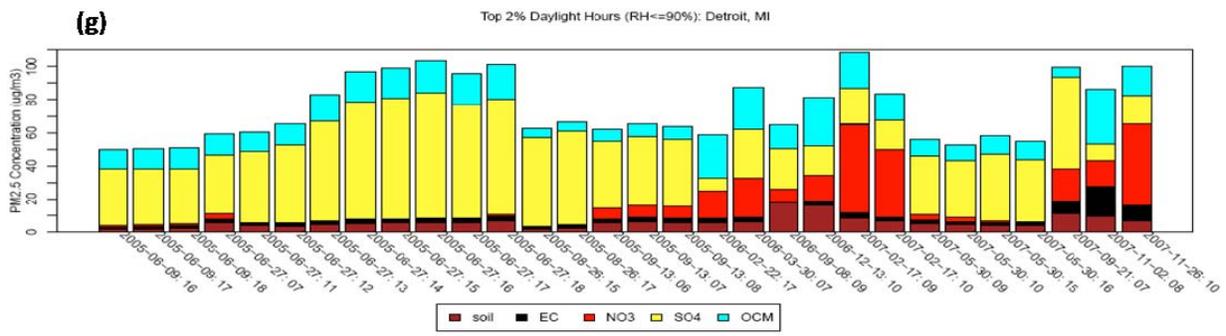
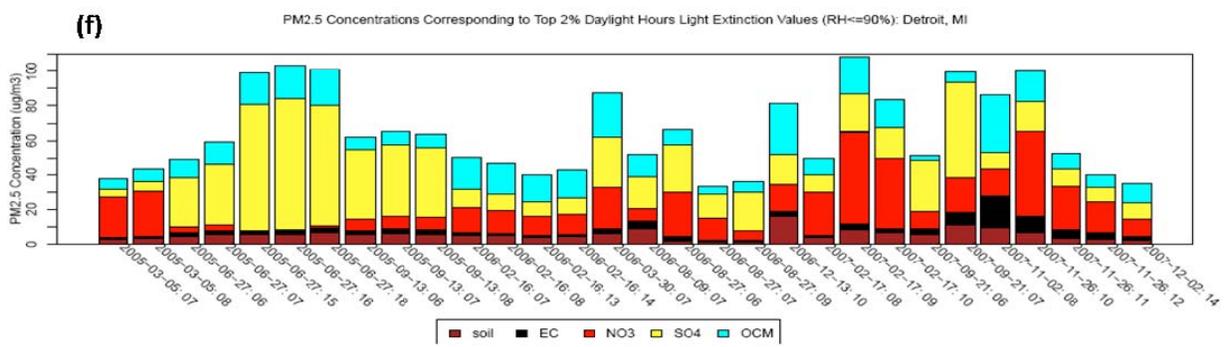
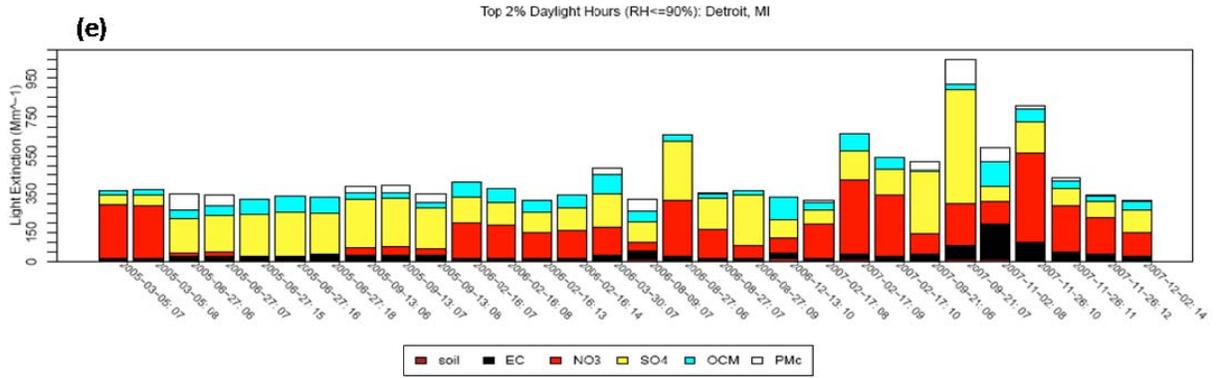


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Figure 4C-10 Detroit PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).



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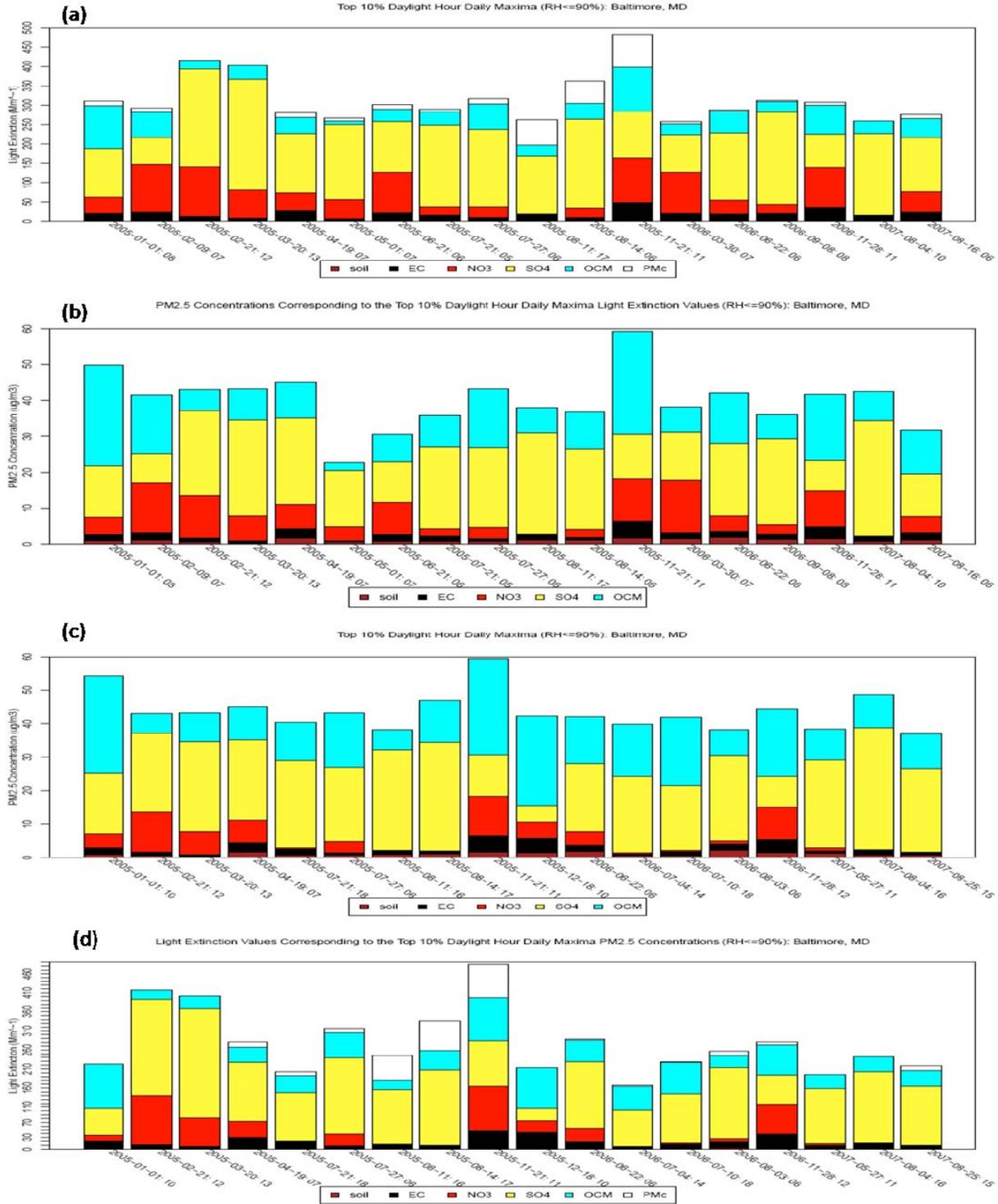
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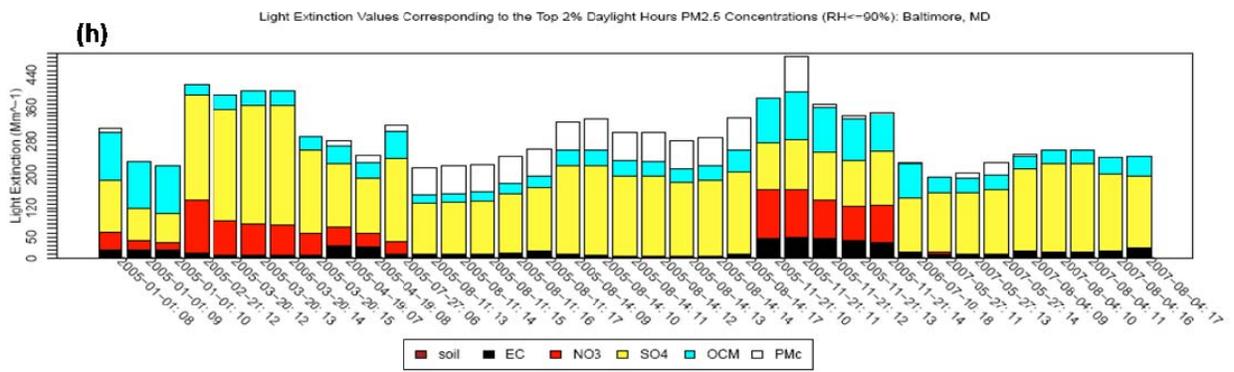
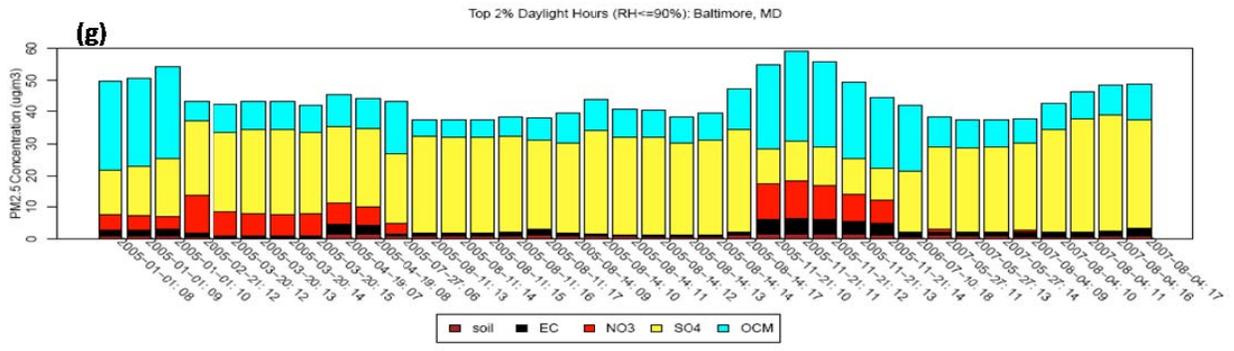
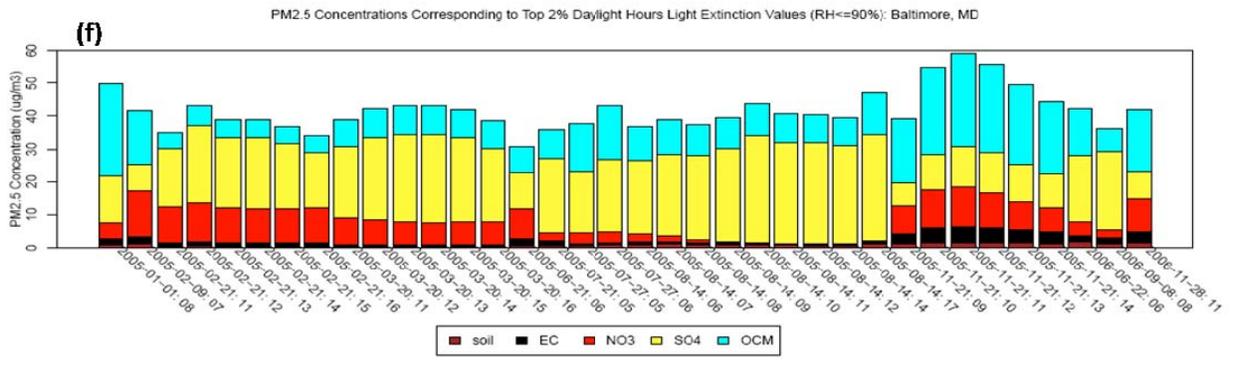
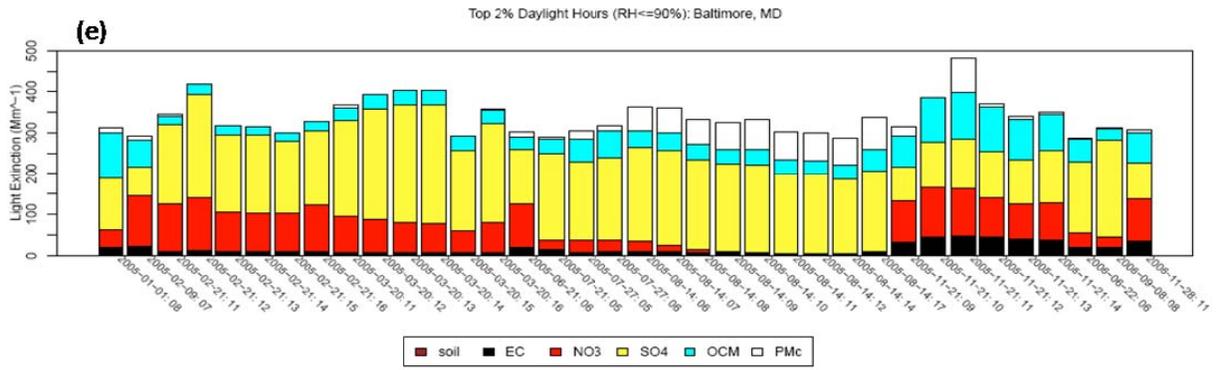
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Figure 4C-11 Baltimore PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

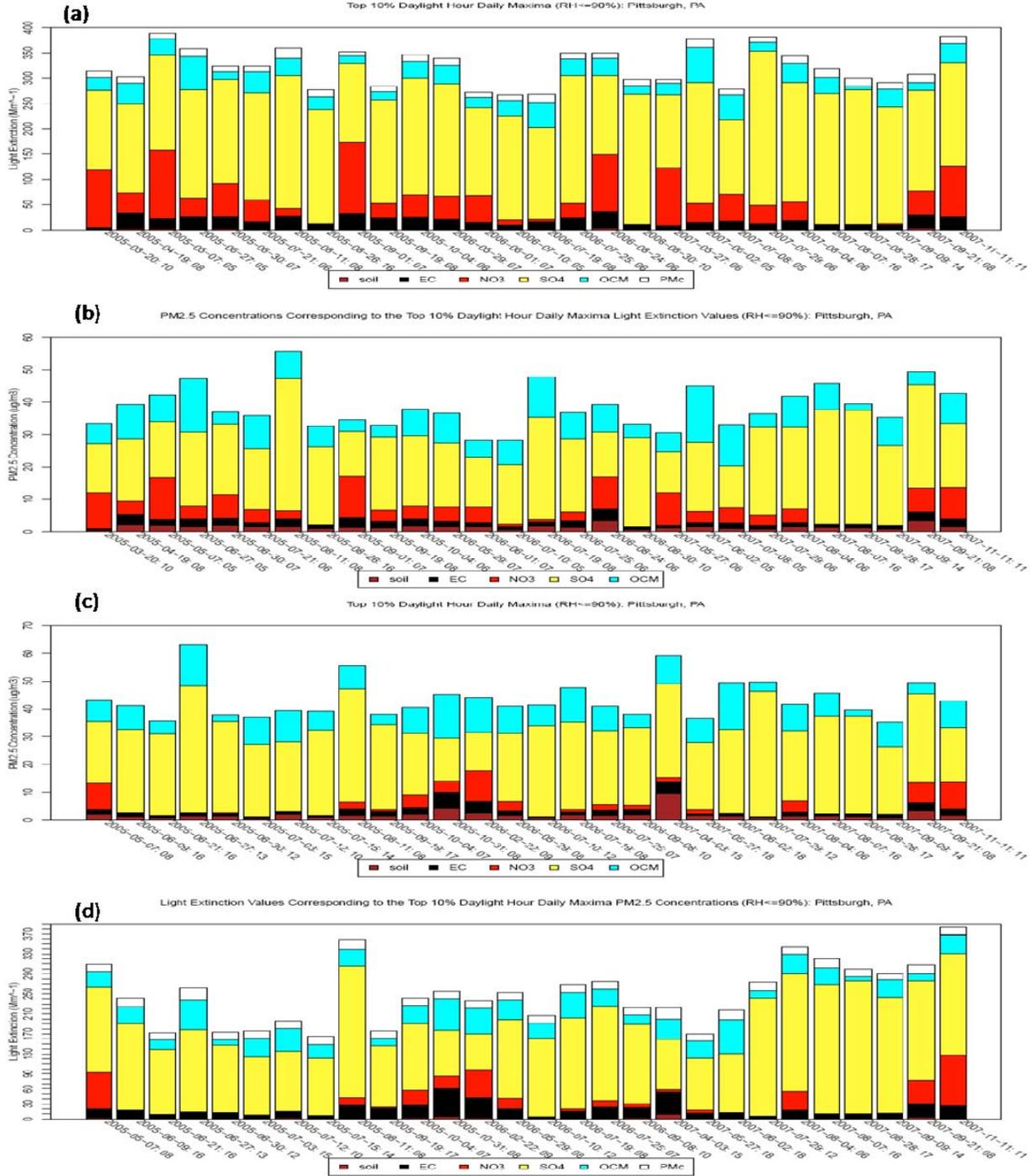


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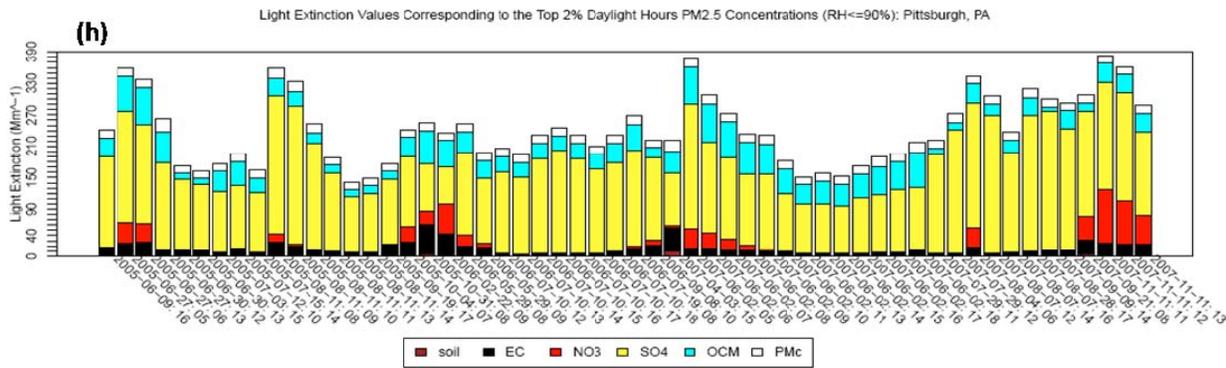
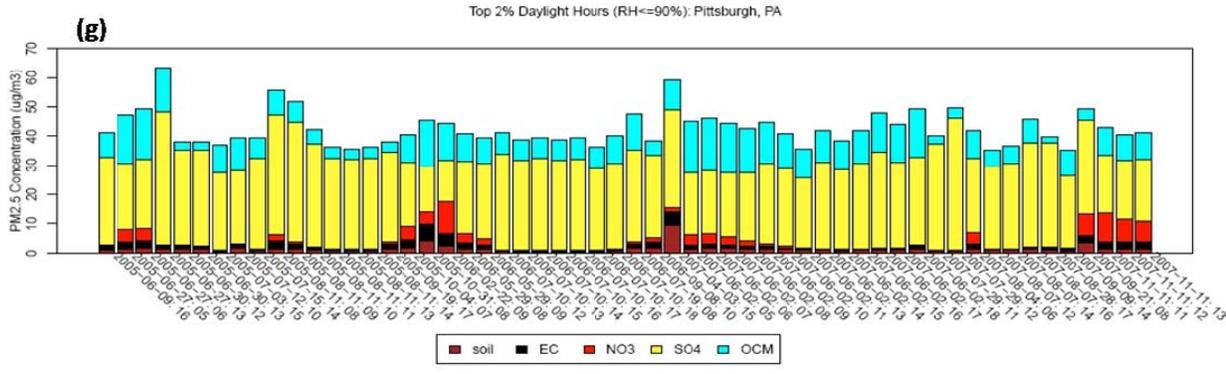
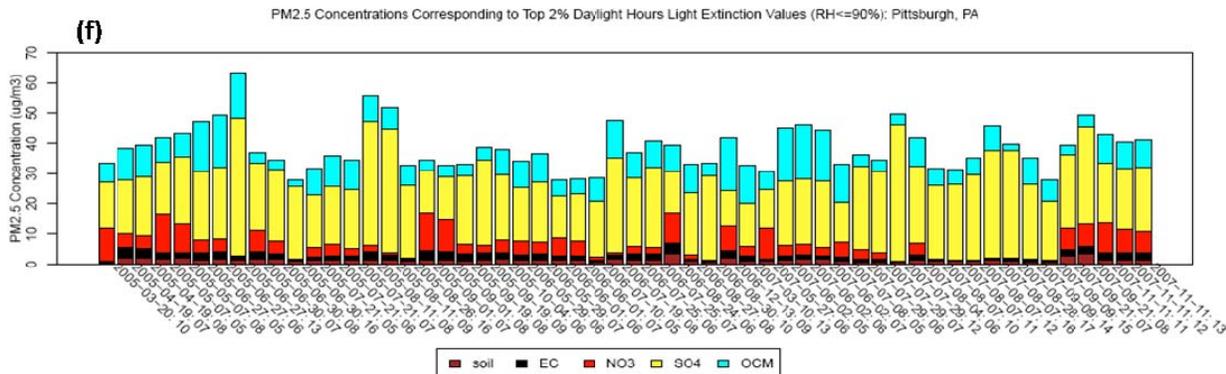
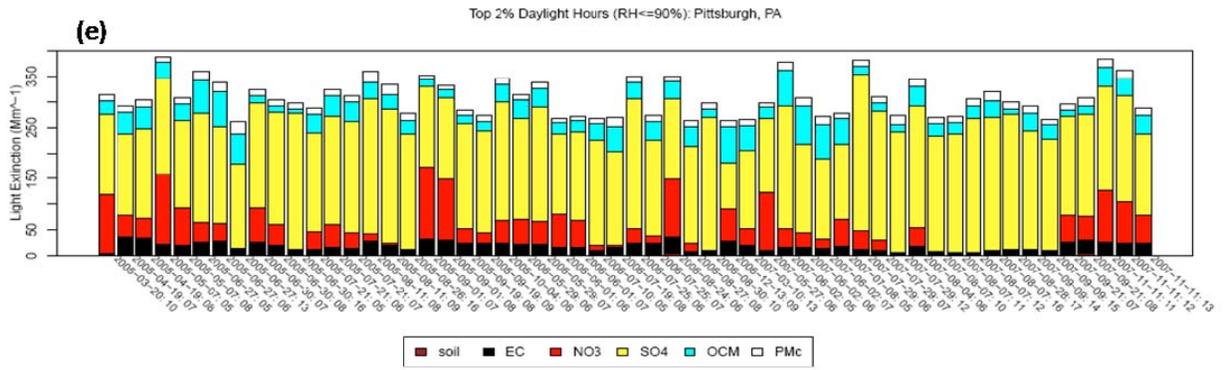


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Figure 4C-12 Pittsburgh PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

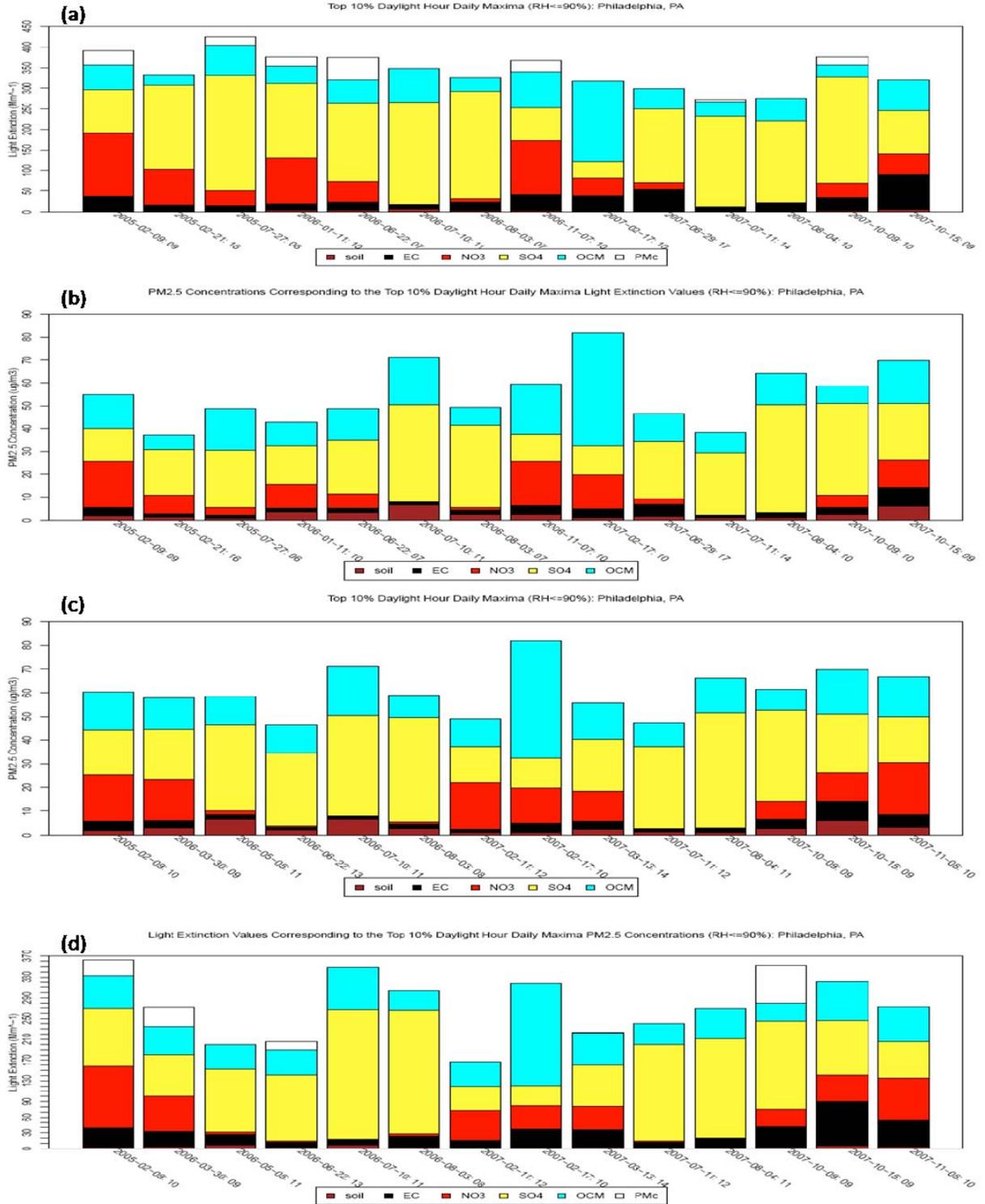


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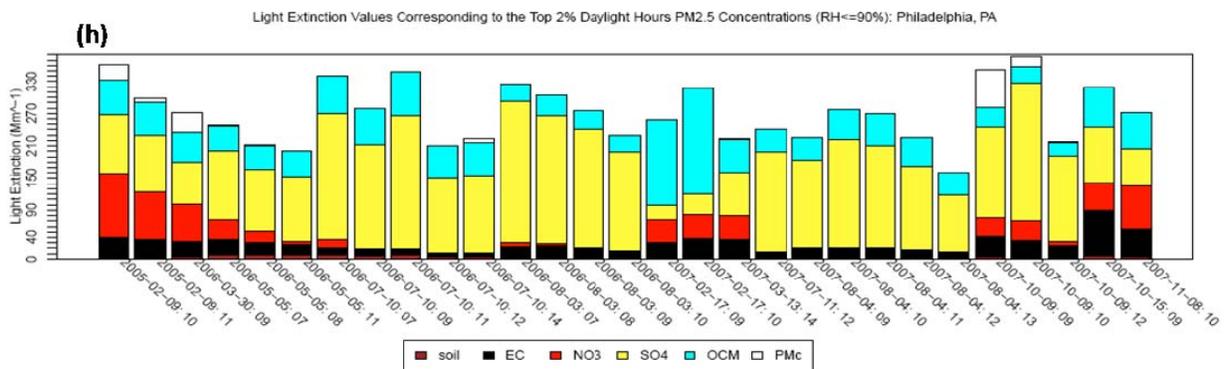
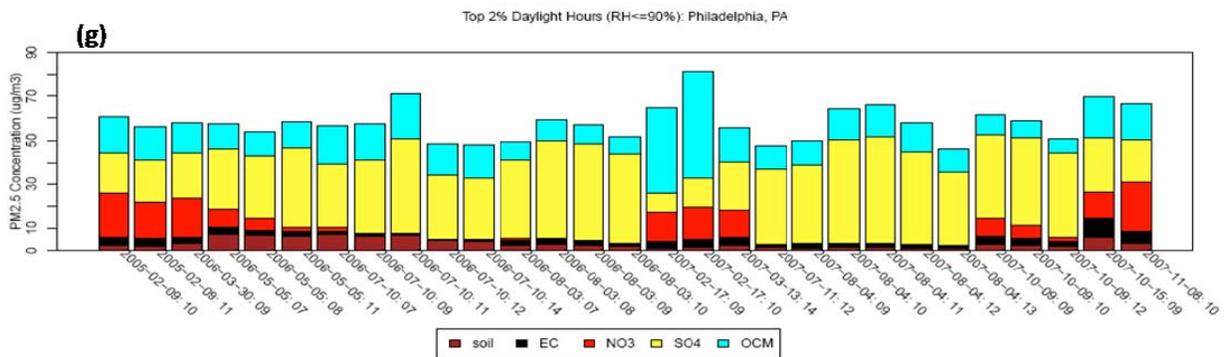
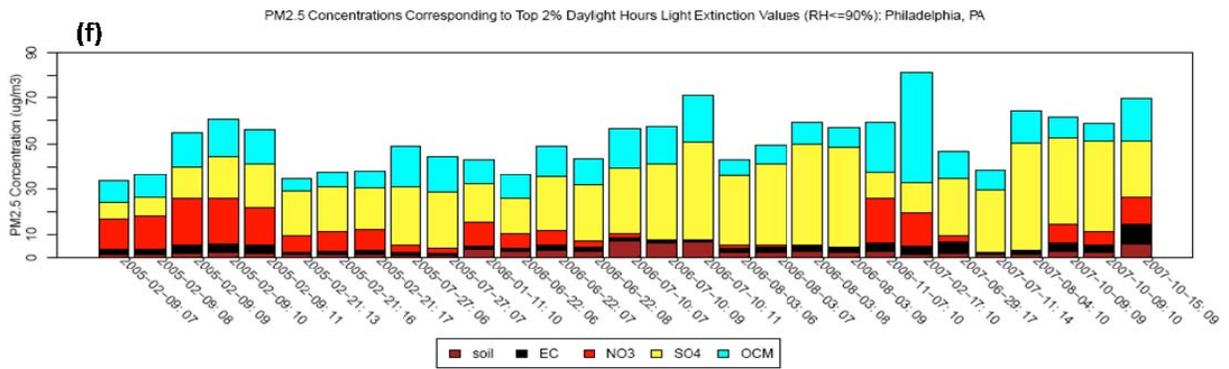
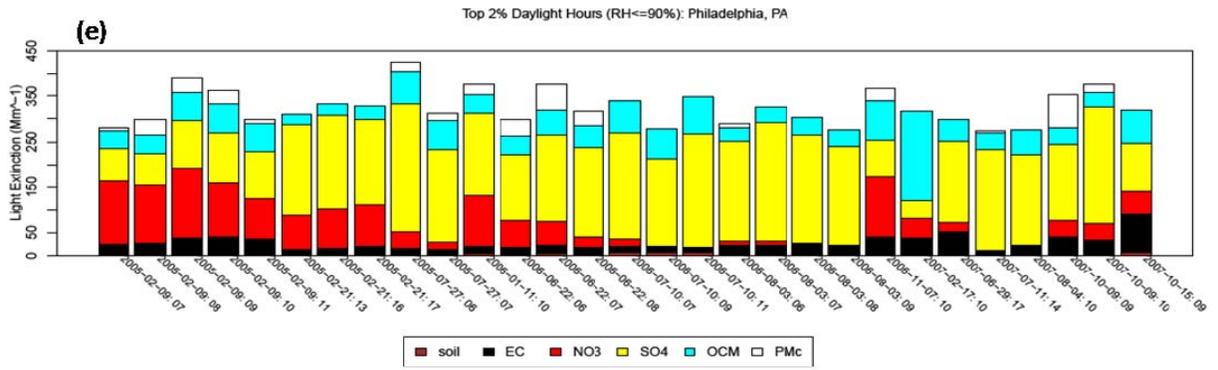


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Figure 4C-13 Philadelphia PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).

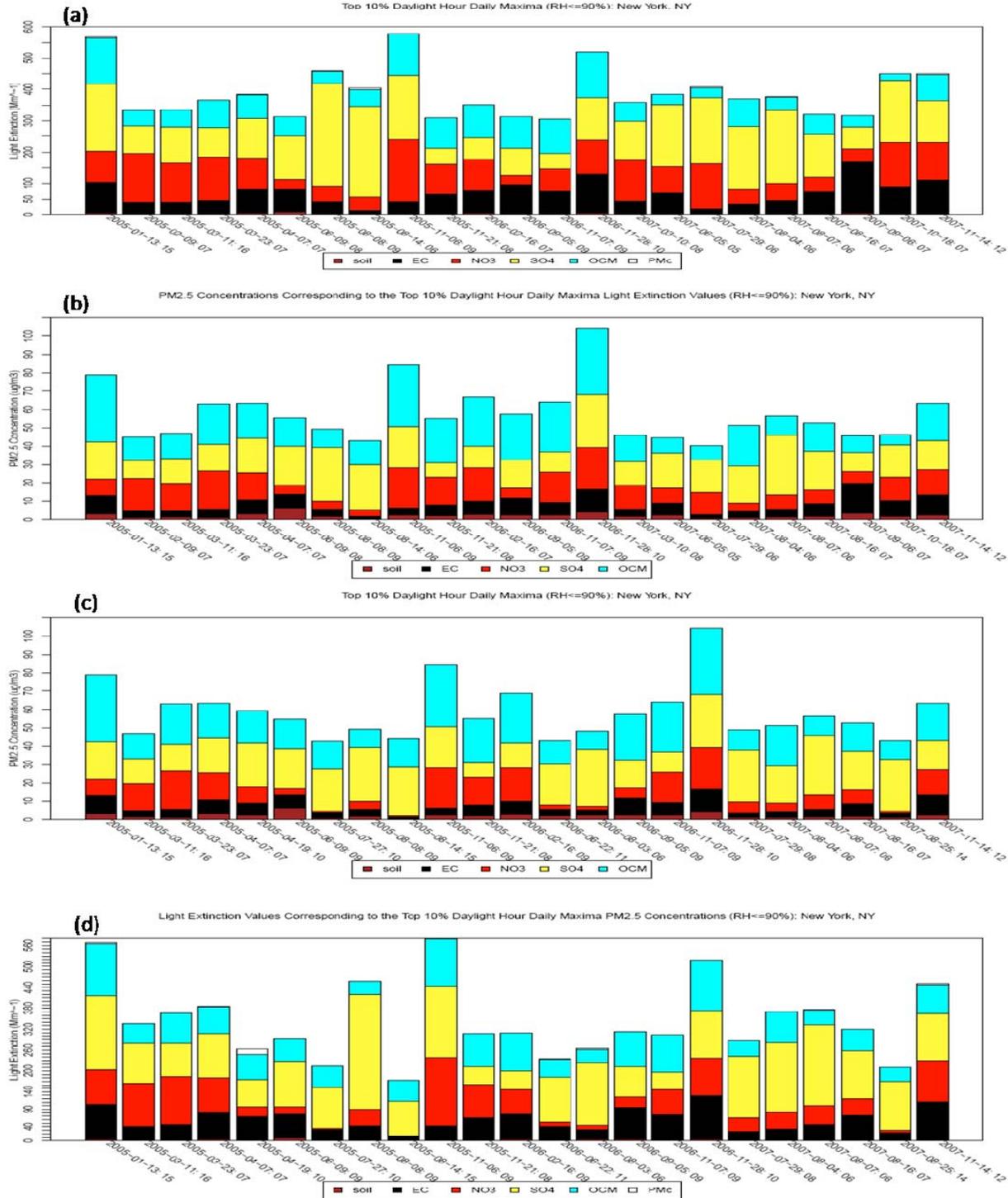


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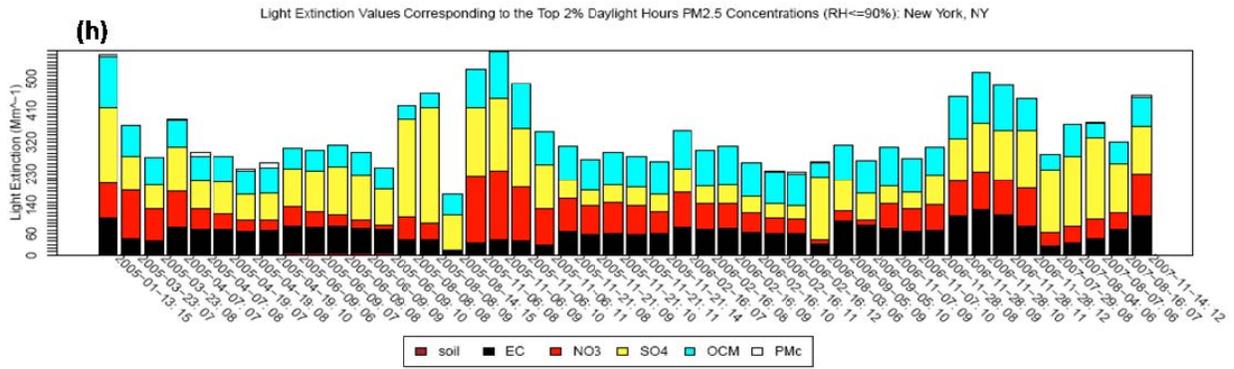
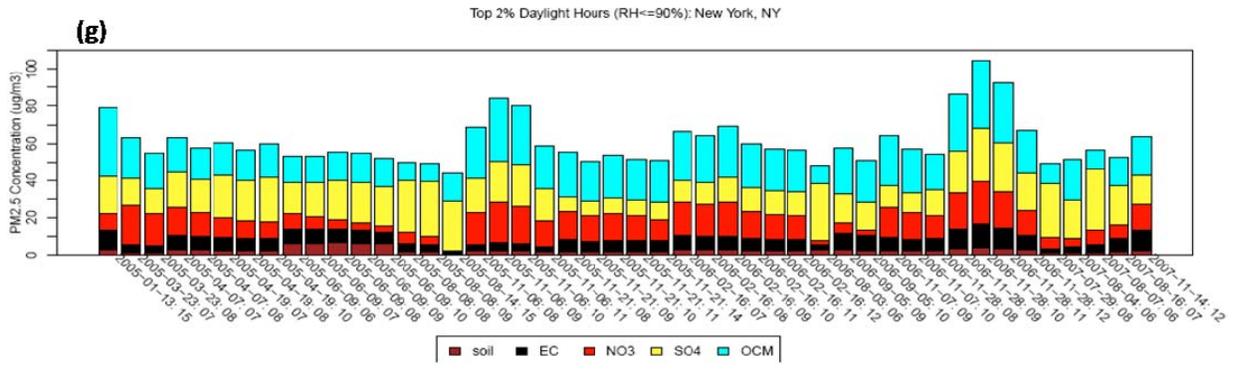
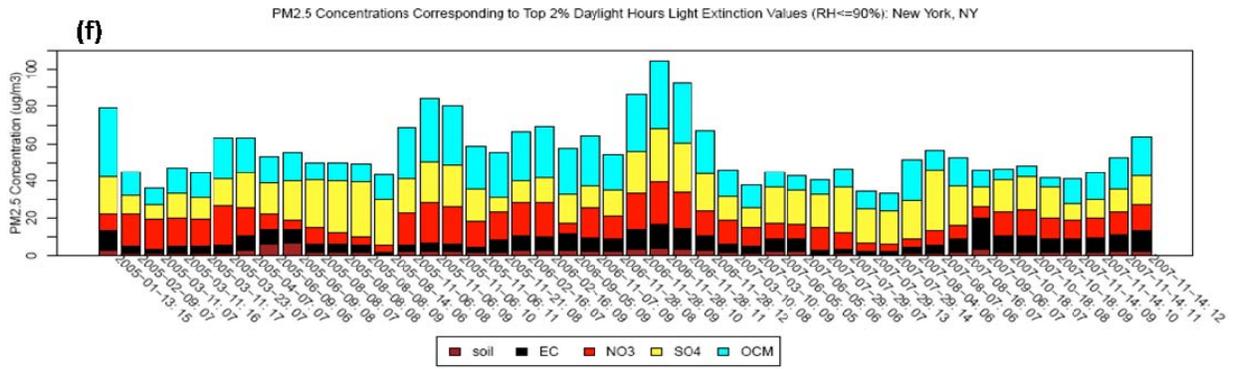
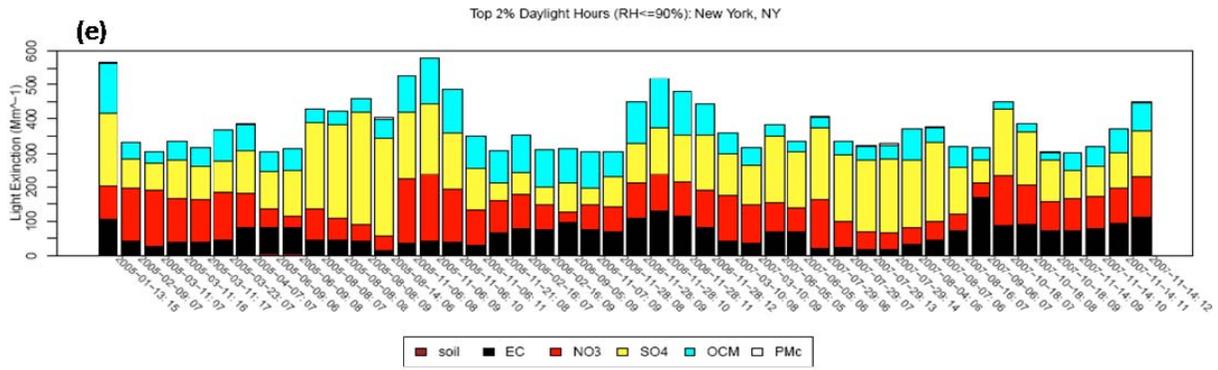


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Figure 4C-14 New York City PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).



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1 In each set of the 8 stacked bar plots, the plots labeled (a) and (b) are the PM₁₀ light
2 extinction and PM_{2.5} mass components for hours selected as the top 10 percent of the maximum
3 daily PM₁₀ light extinction; and the plots labeled (c) and (d) are the PM_{2.5} mass and PM₁₀ light
4 extinction components for the hours selected as the top 10 percent of the maximum daily PM_{2.5}
5 mass concentration. The plots labeled (e) through (h) are in the same order as (a) through (d)
6 except these hours were selected based on their being in the top 2% of all daylight hours for
7 PM₁₀ light extinction (e) and (f), and the top 2% of all daylight hours for PM_{2.5} mass (g) and (h).
8 In other words, there are four unique selections of hours based on two different indicators (i.e.,
9 PM₁₀ light extinction and PM_{2.5} mass) and two different forms (i.e. top 10% of maximum hourly
10 and top 2% of all daylight hours). Each of these four selections of hours is displayed first as the
11 components of the indicator used to make the selection, then in the following plot as the
12 components of the indicator not used in the selection of the hours.

13 Three sets of comparisons are possible using the plots in Figures 4C1 to 4C14. The first
14 two are the comparisons of the PM composition for hours selected for the two PM indicators
15 (i.e., light extinction and mass) and the two forms (i.e., 90th percentile maximum daily and 98th
16 percentile of all daylight hours). The key difference for these first two types of comparisons has
17 to do with the PM components for the hours selected by these four combinations of indicator and
18 form. The third set of comparisons is the apportionments of components using PM₁₀ light
19 extinction versus using PM_{2.5} mass where the hours selected are identical.

20 For any selected hour, the relative contributions of components to PM mass is not the
21 same as the relative contributions of the components to PM light extinction because of the
22 differing light extinction efficiencies of the components. Similarly, the relative importance of
23 the hygroscopic PM components (i.e., PM sulfates and nitrates) compared to that of the non-
24 hygroscopic components (i.e. primarily organic and elemental carbon) to PM mass is generally
25 different than it is to PM light extinction and that difference changes as a function of relative
26 humidity.

27 Indicator Comparisons: To assess the similarity of light extinction components selected
28 by the two indicators compare plots (a) and (d) for the daily maximum form, as well as plots (e)
29 and (h) for the all hours form for each urban area. A corresponding set of comparisons of the
30 PM components can be done by comparing plots (b) and (c), as well as plots (f) and (g) for each
31 urban area. For each of the urban areas there are multiple examples of identical or very nearly
32 identical contributions among the top contributors for PM₁₀ light extinction and PM_{2.5} mass. A
33 more careful examination shows that the two indicators have selected the same hour or another
34 hour during the same day as top contributors (note that the dates and times are listed below each
35 stacked bar).

36 To more efficiently identify the degree to which common and unique periods are selected
37 by the two indicators, Table 4C-1 lists the numbers of common and unique days selected by the

1 two. The greater the number of common days compared to the number of unique days, the more
2 similar are the components of PM selected by the two indicators. For example, Tacoma, Fresno,
3 Salt Lake City, Birmingham and New York each have almost twice as many common days as
4 unique day for the maximum daily form, though only Tacoma and Fresno have this characteristic
5 for the all hours form. Most of the other urban areas have nearly comparable number of unique
6 and common days for the 1-hour maximum daily form, and comparable to as much as three
7 times (Dallas) the number of unique days compared to common days for the all hours form.
8 Locations with relatively large numbers of unique days selected are more likely to have the
9 greatest differences of the mix of PM components. For example, Los Angeles and Dallas have
10 greater contributions by organics when $PM_{2.5}$ mass is the indicator than when PM_{10} light
11 extinction is used. In addition, many of the eastern urban areas have higher contributions of
12 organics and less contributions by PM nitrate when $PM_{2.5}$ mass is the indicator than when PM_{10}
13 light extinction is used.

14 Even if the unique hours selected by $PM_{2.5}$ mass have similar composition they
15 necessarily have lower light extinction than those selected based on being in the top percentage
16 of light extinction. This can be seen by comparing the magnitude of the lowest selected light
17 extinction hour in plots (a) or (e) with the light extinction values for the hours selected using
18 $PM_{2.5}$ mass in corresponding plots (d) or (h) for Figures 4C-1 to 4C-14. For example, notice that
19 for Birmingham (Figure 4C-8) there are many hours selected by $PM_{2.5}$ mass with light extinction
20 values well below the minimum PM_{10} light extinction value for hours selected based on PM_{10}
21 light extinction (i.e. compare (d) to (a) and (h) to (e)).

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Table 4C-1 The numbers of common and unique days selected for each urban area comparing PM₁₀ light extinction versus PM_{2.5} mass for the top 10% of daily maximum and for the top 2% of all hours.

	10% maximum		2% all hours	
	common	unique	common	unique
Tacoma	9	4	11	6
Fresno	27	10	20	10
Los Angeles	18	24	14	38
Phoenix	5	8	9	18
Salt Lake City	25	12	14	12
Dallas	17	20	13	43
Houston	11	7	16	23
Birmingham	27	16	32	26
Atlanta	18	19	19	27
Detroit	9	10	8	22
Baltimore	12	12	6	22
Pittsburgh	19	18	16	30
Philadelphia	9	10	8	17
New York City	18	10	17	16

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6 Form Comparisons: A similar set of comparisons of PM component mixtures can be
7 made between hours selected based on their being in the top 10% of daily maximum or the top
8 2% of all daylight hours. For Figures 4C-1 to 4C-14 composition displayed in plots labeled (a)
9 and (e) for the PM₁₀ light extinction indicator and (c) and (g) for the PM_{2.5} indicator can be
10 compared. Some of the same hours are selected by both forms. By definition the top 10%
11 maximum daily hour form never has more than 1 hour selected per day. There are very nearly
12 twice as many hours in the top 2% of all hours compared to the top 10% of maximum daily

1 hours. However, it is not unusual to have multiple hours selected in the same day by the top 2%
2 of all hours form. These hours selected in the same day tend to have nearly identical
3 composition (e.g., Figure 4C-1 (e) has 4 consecutive hours on 11/08/2007, which is also selected
4 with a 10% maximum daily form (a)). The top 2% of all hours may select hours on a greater
5 number of days compared with the top 10% of maximum daily hours, or if there are enough
6 instances of multiple hours per day, it could select hours on a smaller number of days compared
7 with the top 10% of maximum daily hours. A list of the numbers of common and unique days
8 selected comparing the two forms for both indicators and the number of days selected by each
9 combination of indicator and form is shown in Table 4C-2.

10 In addition to affecting the component mixture for the top hours selected, the number of
11 multiple hours per day selected among the top 2% of all hours affects whether its design values
12 are greater than or less than the corresponding design values for the top 10% of maximum daily
13 hours. If the number of multiple hours per day is small enough that the 2% of all hours form has
14 more days selected than the top 10 percent of all hours, then the design value for the 2% form is
15 smaller than that of the 10% daily maximum form. An example of this is light extinction for
16 Birmingham where the 2% of all hours form selects hours on 47 days and has a PM light
17 extinction design value of 309 Mm^{-1} compared to the 35 days that have hours in the top 10% of
18 daily maximum light extinction with a design value of 357 Mm^{-1} . Among the 14 urban areas,
19 Salt Lake City light extinction is the most extreme example of the opposite situation with only 16
20 days having the top 2% of all hours and a design value of 225 Mm^{-1} compared with 31 days
21 selected by the top 10% maximum daily light extinction and a design value of 164 Mm^{-1} .

22 For most of the urban areas the number of common days is much larger than the number
23 of unique days. Generally, the PM component mix is less similar for sites with a lower number
24 of common and a greater number of unique days. For example, the 2% of all hours form for Salt
25 Lake City selected only one hour of one of the days where the PM carbonaceous components
26 was the major contributor compared with the 4 hours with high PM carbonaceous components
27 selected in the top 10% of the maximum daily form (compare Figure 4C-5 (a) with (e)). By
28 comparison for Salt Lake City most of the multiple hours in single days had high contributions
29 from PM nitrate. Composition of the hours selected for the two forms by either indicator are
30 generally very similar. To facilitate comparisons among the PM component contributions for the
31 two forms and indicators, Figure 4C-15 shows stacked bar plots of the average PM component
32 contributions by city.

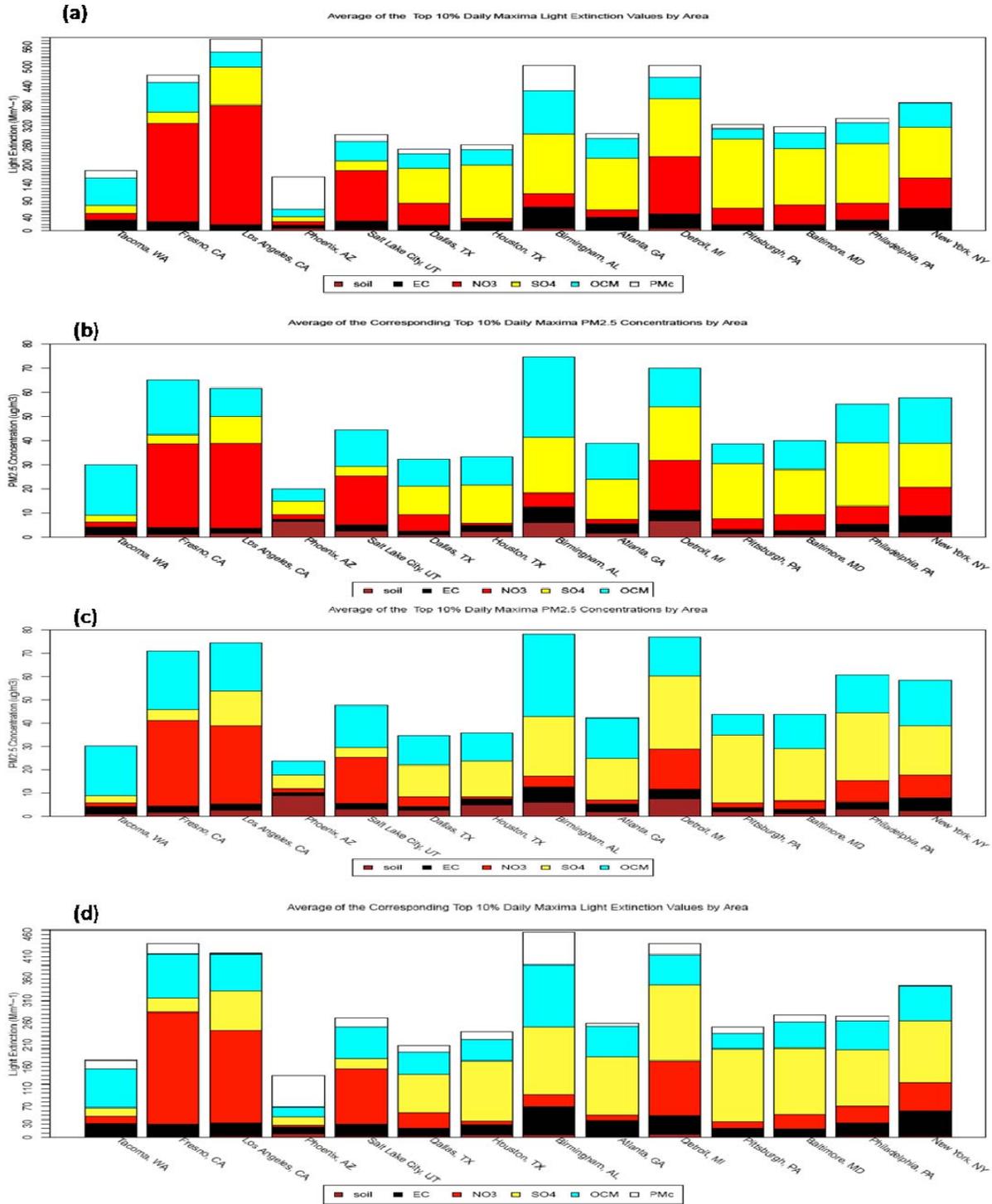
1 **Table 4C-2 The numbers of common and unique days selected for each urban area comparing the top 10%**
 2 **of 1-hr daily maximum to the top 2% of all hours for PM₁₀ light extinction and PM_{2.5} mass indicators. Also**
 3 **shown are the number of days selected for each form and indicator.**

	PM ₁₀ Light Extinction				PM _{2.5} Mass			
	10% vs. 2%		Number of days		10% vs. 2%		Number of days	
	common	unique	10%	2%	common	unique	10%	2%
Tacoma	11	3	11	14	11	1	11	12
Fresno	25	7	32	25	20	12	32	20
Los Angeles	30	8	30	38	18	12	30	18
Phoenix	9	6	9	14	9	5	9	14
Salt Lake City	16	15	31	16	18	13	31	18
Dallas	25	2	27	25	23	4	27	23
Houston	14	11	14	23	15	8	15	23
Birmingham	35	14	35	47	35	1	35	46
Atlanta	27	6	27	33	27	1	28	27
Detroit	13	1	14	13	13	1	14	13
Baltimore	12	6	18	12	11	7	18	11
Pittsburgh	28	3	28	31	24	4	28	24
Philadelphia	14	0	14	14	12	2	14	12
New York City	23	0	23	23	19	4	23	19

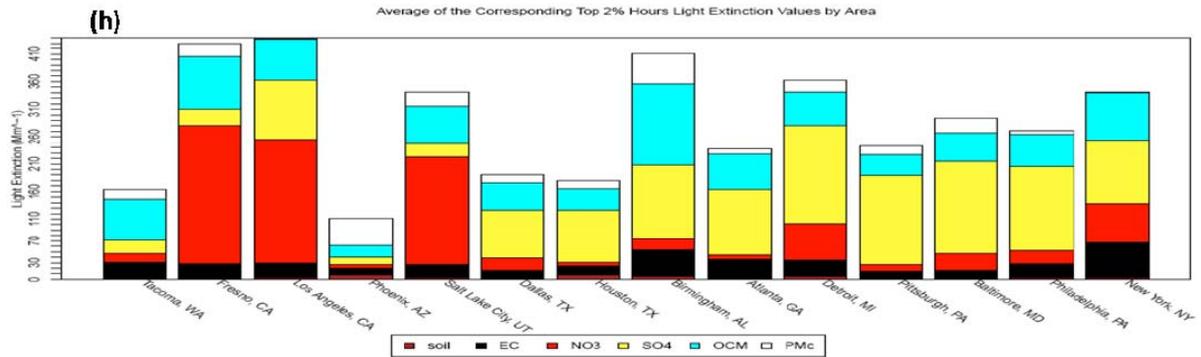
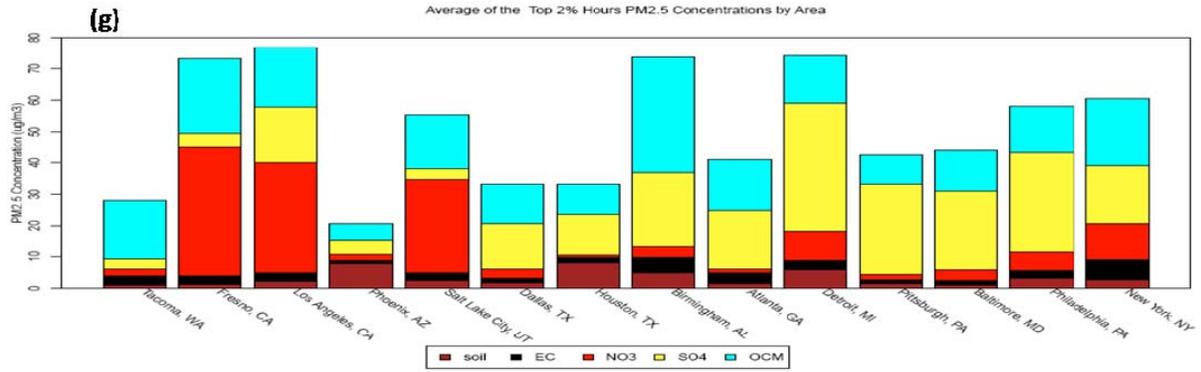
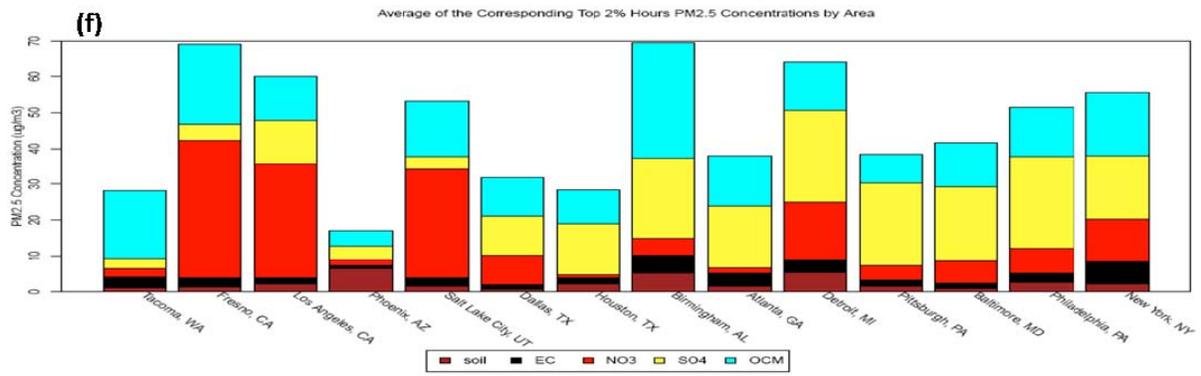
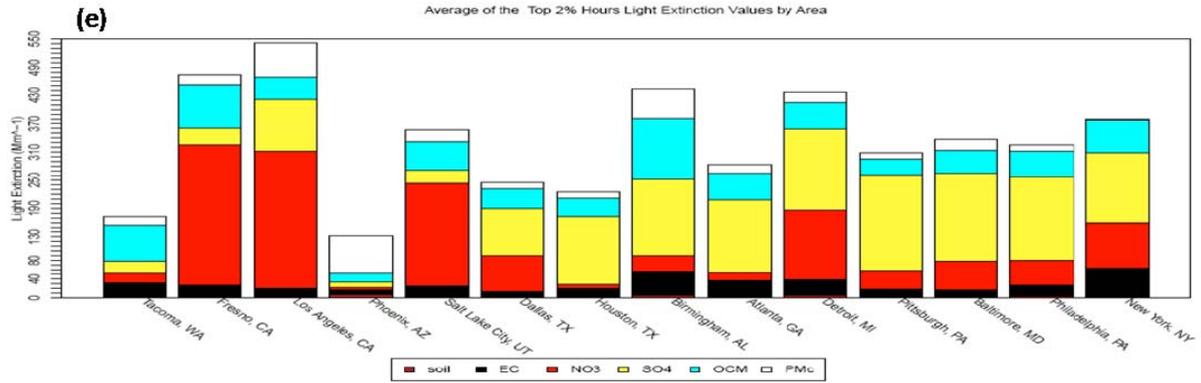
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Figure 4C-15 Average by urban areas of the PM components contributing to the top 10% of maximum daily 1-hour PM₁₀ light extinction (a & b) and PM_{2.5} mass (c & d), and to the top 2% of all daylight hours for PM₁₀ light extinction (e & f) and for PM_{2.5} mass (g & h).



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1 As discussed above the greatest differences between the two forms is for Salt Lake City.
2 For most of the 14 urban areas, the differences in results comparing the two forms are minor
3 (compare (a) to (e) and (c) to (g)). Figure 4C-15 also facilitates comparisons of the average
4 effects on PM component mixtures depending on which of the two indicators is used (compare
5 (a) to (d) and (e) to (h)). It shows that the difference in PM component contributions differ more
6 depending on the choice of indicator than the choice of form.

7 Component Apportionment Comparisons: This is the simplest of the three types of
8 comparisons because the exact same hours are compared and the only difference is in the relative
9 contributions of the components to PM₁₀ light extinction compared to the relative component
10 contributions to PM_{2.5} mass. To see these differences for individual hours or averages by urban
11 areas compare plots labeled (a) and (b), (c) and (d), (e) and (f), or (g) and (h) in Figures 4C-1 to
12 4C-15. The differences directly result from the application of the IMPROVE algorithm which
13 weights the various PM mass components differently when expressing them as components of
14 light extinction. For example, notice that the soil component of PM₁₀ light extinction is barely
15 discernable for Birmingham and Detroit, but is easily discerned as a component of PM_{2.5} mass.
16 In many of the urban areas the relative contributions by soil and organic mass PM components
17 are greater and the relative contributions by elemental carbon, sulfate and nitrate components are
18 less when expressed as contributions to PM_{2.5} mass than when expressed as PM₁₀ light
19 extinction.

20 The differences in the relative apportionment of the PM components are of concern when
21 emission control efforts are designed to reduce visibility impacts. Use of the relative
22 contribution by PM₁₀ light extinction components recognizes that some PM components cause
23 more visibility impacts per unit of mass concentration, whereas the use of relative contributions
24 to PM_{2.5} mass treat the most and least effective components as though they contributed equally
25 on a per unit mass contribution basis.

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