Review of the National Ambient Air Quality Standards for Ozone:

Policy Assessment of Scientific and Technical Information

OAQPS Staff Paper

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REVIEW OF THE NATIONAL AMBIENT AIR QUALITY STANDARDS FOR OZONE:

POLICY ASSESSMENT OF SCIENTIFIC AND TECHNICAL INFORMATION

OAQPS STAFF PAPER

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DISCLAIMER

This document has been reviewed by the Office of Air Quality Planning and Standards (OAQPS), U.S. Environmental Protection Agency (EPA), and approved for publication. This final OAQPS Staff Paper contains the conclusions and recommendations of staff of OAQPS and does not necessarily represent those of the EPA. Mention of trade names or commercial products is not intended to constitute endorsement or recommendation for use.

PREFACE TO JULY 2007 EDITION

The purpose of this July 2007 edition is to include the March 26, 2007 letter from the Clean Air Scientific Advisory Committee (CASAC) to the Administrator providing its final comments and advice on the January 2007 edition of the OAQPS Staff Paper. The CASAC letter has been included as an Attachment to this edition (Attachment B). In addition, this edition of the Staff Paper includes technical corrections to the exposure model resulting in revised exposure estimates and revised lung function risk estimates. As noted in chapters 4 and 5 of this edition of the Staff Paper, a small error was detected in the exposure model in January 2007 that resulted in small increases in the exposure estimates. This error has been corrected and the model runs have been redone, generally resulting in small increases in the exposure estimates. The corrected results are presented in this edition of the Staff Paper and in the revised Exposure Analysis technical support document. The lung function risk estimates also have been redone based on the corrected exposure estimates and are generally slightly higher than the original estimates presented in the January 2007 edition of the Staff Paper. The corrected lung function risk estimates are presented in this edition of the Staff Paper and in the revised Risk Assessment technical support document. Also, Figure 3-4 on page 3-56 has been modified to include a legend of the studies used to create the figure. Additional minor edits to Chapters 4 and 5 are listed in the errata.

ACKNOWLEDGMENTS

This Staff Paper is the product of the Office of Air Quality Planning and Standards (OAQPS). For the chapters on ozone-related health effects, exposure, risk, and primary standards, the principal authors include Karen Martin, David McKee, Harvey Richmond, Susan Lyon Stone, and John Langstaff. For the chapters on ozone-related welfare effects and secondary standards, the principal authors include Vicki Sandiford and Jeffrey Herrick. The principal authors of the chapter on air quality characterization include Lance McCluney and Michael Rizzo. Staff from other EPA offices, including the Office of Research and Development, the Office of General Counsel, and the Office of Transportation and Air Quality, also provided valuable comments.

Earlier drafts of this document were formally reviewed by the Clean Air Scientific Advisory Committee (CASAC) and made available for public comment. This document has been informed by the expert advice and comments received from CASAC, as well as by public comments submitted by a number of independent scientists, officials from State and local air pollution organizations, environmental groups, and industrial groups and companies.

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ERRATA

Corrections made to document:

Title page and back page, publication number has been changed to "EPA-452/07-007."

For Chapter 3:
On page 3-56, a list of epidemiology studies referenced in Figure 3-4 was added.

For Chapter 4:
The following tables and figures were revised and the text was revised to reflect the corrected estimates presented in these tables and figures:
Tables 4-8, 4-9,
Figures 4-2, 4-3
Figures 4-9, 4-10, 4-11
Exhibits 1 through 9

In the last paragraph on page 4-1, continuing onto page 4-2, a redundant sentence was removed.

The footnote on page 4-1 was revised to clarify why the results for active children were not presented in the Staff Paper. The corrected exposure modeling, conducted after the Staff Paper was published, fixed the error in how children are characterized as active.

Figures 4-2 and 4-3 (comparison of modeled and measured weekly average personal exposures) were changed to present the corrected exposure estimates. The discussion of these results on pages 4-32 and 4-34 was revised accordingly, and now concludes that the exposures predicted by APEX are generally in good agreement with the personal exposure measurements.

Tables 4-8 and 4-9 (comparison of APEX model results based on all CHAD with model results based on the NHAPS part of CHAD) were revised to reflect the corrected exposure estimates. The correction results in larger differences between these simulations, and the text on page 4-35 was revised accordingly.

The references for Chapter 4 were edited to make the style consistent and to remove references not cited in the text.

ERRATA

For Chapter 5:
The following tables and figures were revised and the text was revised to reflect the corrected estimates presented in these tables and figures:
Tables 5-8, 5-9, 5-10, 5-11
Figures 5-7, 5-8, 5-9, 5-10, 5-17a, b, 5-18a,b

The references to the Exposure Assessment TSD and Risk Assessment TSD were updated to reflect the corrected final reports.

For Chapter 6:
The following tables and figures were revised and the text was revised to reflect the corrected estimates presented in these tables and figures:
Tables 6-1a,b,c, 6-2, 6-3, 6-7, 6-8
Figures 6-1, 6-2, 6-3, 6-4

For Appendix 4A:
All of the tables in this appendix were revised:

For Appendix 5C:
The following tables and figures were revised:
Tables 5C-1 through 5C-6
Table 5C-17
Figures 5C-1 through 5C-4

Attachment B was added, which is a copy of the March 26, 2007 CASAC letter to the Administrator of EPA regarding comments on Chapters 6 and 8 of the final Staff Paper.

1. INTRODUCTION

1.1 PURPOSE

This Staff Paper, prepared by staff in the U.S. Environmental Protection Agency's (EPA) Office of Air Quality Planning and Standards (OAQPS), evaluates the policy implications of the key studies and scientific information contained in the document, *Air Quality Criteria for Ozone and Related Photochemical Oxidants* (USEPA, 2006; henceforth referred to as the CD), prepared by EPA's National Center for Environmental Assessment (NCEA). This document also presents and interprets results from several quantitative analyses (e.g., air quality analyses, human exposure analyses, human health risk assessments, and an environmental assessment of vegetation-related impacts) that we believe should also be considered in EPA's current review of the national ambient air quality standards (NAAQS) for ozone (O_3), and presents factors relevant to the evaluation of current primary and secondary 0_3 standards. Finally, this document presents staff conclusions and recommendations on a range of policy options that we believe are appropriate for the Administrator to consider concerning whether, and if so how, to revise the primary (health-based) and secondary (welfare-based) O_3 NAAQS.

The policy assessment presented in this Staff Paper is intended to help "bridge the gap" between the scientific assessment contained in the CD and the judgments required of the EPA Administrator in determining whether it is appropriate to retain or revise the NAAQS for O_3. This policy assessment considers the available scientific evidence and quantitative risk-based analyses, together with related limitations and uncertainties, and focuses on the basic elements of air quality standards: indicator, averaging times, forms,[1] and levels. These elements, which serve to define each standard, must be considered collectively in evaluating the health and welfare protection afforded by the O_3 standards. Our conclusions and policy recommendations on whether, and if so how, to revise these standard elements are based on the assessment and integrative synthesis of information presented in the CD and on staff analyses and evaluations presented in this document, and are further informed by comments and advice received from an independent scientific review committee, the Clean Air Scientific Advisory Committee (CASAC),[2] in their review of earlier drafts of this document, as well as comments on earlier drafts submitted by public commenters.

[1] 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.

[2] The statutory requirements for CASAC are discussed below in the next section.

While this Staff Paper should be of use to all parties interested in the O_3 NAAQS review, it is written with an expectation that the reader has some familiarity with the technical discussions contained in the CD.

1.2 BACKGROUND

1.2.1 Legislative Requirements

Two sections of the Clean Air Act (Act) govern the establishment and revision of the NAAQS. Section 108 (42 U.S.C. 7408) directs the Administrator to identify and list "air pollutants" that "in his judgment, may reasonably be anticipated to endanger public health and welfare" and whose "presence . . . in the ambient air results from numerous or diverse mobile or stationary sources" and, if listed, to issue air quality criteria for them. These air quality criteria are intended to "accurately reflect the latest scientific knowledge useful in indicating the kind and extent of identifiable effects on public health or welfare which may be expected from the presence of [a] pollutant in ambient air . . . ."

Section 109 (42 U.S.C. 7409) directs the Administrator to propose and promulgate "primary" and "secondary" NAAQS for pollutants identified under section 108. Section 109(b)(1) defines a primary standard as one "the attainment and maintenance of which in the judgment of the Administrator, based on such criteria and allowing an adequate margin of safety, are requisite to protect the public health."[3] A secondary standard, as defined in Section 109(b)(2), must "specify a level of air quality the attainment and maintenance of which, in the judgment of the Administrator, based on such criteria, is requisite to protect the public welfare from any known or anticipated adverse effects associated with the presence of [the] pollutant in the ambient air."[4]

The requirement that primary standards include an adequate margin of safety was intended to address uncertainties associated with inconclusive scientific and technical information available at the time of standard setting. It was also intended to provide a reasonable degree of protection against hazards that research has not yet identified. *Lead Industries Association v. EPA*, 647 F.2d 1130, 1154 (D.C. Cir 1980), cert. denied, 101 S. Ct. 621 (1980);

[3] 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)].

[4] Welfare effects as defined in section 302(h) [42 U.S.C. 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."

American Petroleum Institute v. Costle, 665 F.2d 1176, 1186 (D.C. Cir. 1981), cert. denied, 102 S.Ct. 1737 (1982). Both kinds of uncertainties are components of the risk associated with pollution at levels below those at which human health effects can be said to occur with reasonable scientific certainty. Thus, in selecting primary standards that include an adequate margin of safety, the Administrator is seeking not only to prevent pollution levels that have been demonstrated to be harmful but also to prevent lower pollutant levels that may pose an unacceptable risk of harm, even if the risk is not precisely identified as to nature or degree.

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

In setting standards that are "requisite" to protect public health and welfare, as provided in section 109(b), EPA's task is to establish standards that are neither more nor less stringent than necessary for these purposes. In so doing, EPA may not consider the costs of implementing the standards. See generally *Whitman v. American Trucking Associations*, 531 U.S. 457, 464, 475-76 (2001).

Section 109(d)(1) of the Act requires that "not later than December 31, 1980, and at 5-year intervals thereafter, the Administrator shall complete a thorough review of the criteria published under section 108 and the national ambient air quality standards . . . and shall make such revisions in such criteria and standards and promulgate such new standards as may be appropriate . . . ." Section 109(d)(2) requires that an independent scientific review committee "shall complete a review of the criteria . . . and the national primary and secondary ambient air quality standards . . . and shall recommend to the Administrator any new . . . standards and revisions of existing criteria and standards as may be appropriate . . . ." Since the early 1980's, this independent review function has been performed by the Clean Air Scientific Advisory Committee (CASAC), a standing committee of EPA's Science Advisory Board.

1.2.2 History of Ozone NAAQS Reviews

Tropospheric (ground-level) O_3 is formed from biogenic precursor emissions and as a result of anthropogenic precursor emissions. Naturally occurring O_3 in the troposphere can result from biogenic organic precursors reacting with naturally occurring nitrogen oxides (NO_x) and by stratospheric O_3 intrusion into the troposphere. Anthropogenic precursors of O_3, specifically NO_x and volatile organic compounds (VOC), originate from a wide variety of stationary and mobile sources. Ambient O_3 concentrations produced by these emissions are directly affected by temperature, solar radiation, wind speed and other meteorological factors.

The EPA initially established primary and secondary NAAQS for photochemical oxidants on April 30, 1971 (36 FR 8186). Both primary and secondary standards were set at an hourly average of 0.08 parts per million (ppm), total photochemical oxidants, not to be exceeded more than one hour per year.

On February 8, 1979, EPA completed its first periodic review of the criteria and standards for O_3 and other photochemical oxidants (44 FR 8202). In that action, EPA made significant revisions to the original standard: the level of the primary and secondary NAAQS was changed to 0.12 ppm; the indicator was changed to O_3; and the form of the standards was changed to be based on the expected number of days per calendar year with a maximum hourly average concentration above 0.12 ppm (i.e., attainment of the standard occurs when that number is equal to or less than one).

On March 9, 1993, EPA concluded its second periodic review of the criteria and standards for O_3 by deciding that revisions to the O_3 NAAQS were not warranted at that time (58 FR 13008). The timing of this decision was required by a court order issued to resolve a lawsuit filed to compel EPA to complete its review of the criteria and standards for O_3 in accordance with the Act. This decision reflected EPA's review of relevant scientific and other information assembled since the last review, as contained in the 1986 O_3 CD (U.S. EPA, 1986), its Supplement (U.S. EPA, 1992), and the 1989 O_3 Staff Paper (U.S. EPA, 1989), although it did not take into consideration a large number of studies on the health and welfare effects of O_3 published since the literature was last assessed in the O_3 Supplement. The final decision emphasized the Administrator's intention to proceed as rapidly as possible with the next periodic review of the air quality criteria and standards to consider the more recent information.

Under a court-ordered schedule and a highly accelerated review process, EPA completed its third review of the O_3 NAAQS on July 18, 1997, based on the 1996 O_3 CD (U.S. EPA, 1996a) and 1996 O_3 Staff Paper (U.S. EPA, 1996b). EPA revised the primary and secondary O_3 standards on the basis of the then latest scientific evidence linking exposures to ambient O_3 to adverse health and welfare effects at levels allowed by the 1-hr average standards (62 FR 38856). The O_3 standards were revised by replacing the existing primary 1-hr average standard with an 8-hr average O_3 standard set at a level of 0.08 ppm. The form of the primary standard was changed to the annual fourth-highest daily maximum 8-hr average concentration, averaged over three years. The secondary O_3 standard was changed by making it identical in all respects to the revised primary standard.

1.2.3 Litigation Related to the 1997 Ozone Standards

Following promulgation of the revised O_3 NAAQS, petitions for review were filed addressing a broad range of issues. On May 14, 1999, in response to those challenges, the U.S.

Court of Appeals for the District of Columbia Circuit (D.C. Circuit Court) remanded the O_3 NAAQS to EPA, finding that section 109 of the Act, as interpreted by EPA, effected an unconstitutional delegation of legislative authority.[5] In addition, the D.C. Circuit Court directed that EPA should consider the potential beneficial health effects of O_3 pollution in shielding the public from the effects of solar ultraviolet (UV) radiation, as well as the adverse health effects.

EPA petitioned the U.S. Supreme Court for certiorari on the constitutional issue but did not request review of the D.C. Circuit Court ruling regarding its obligation to consider the potential beneficial health effects of O_3. On February 27, 2001, the U.S. Supreme Court unanimously reversed the judgment of the D.C. Circuit Court on the constitutional issue, holding that section 109 of the CAA does not delegate legislative power to the EPA in contravention of the Constitution, and remanded the case to the D.C. Circuit Court to consider those challenges to the O_3 NAAQS that had not been addressed by that Court's earlier decisions.[6] On March 26, 2002, the D.C. Circuit Court issued its final decision, finding the 1997 O_3 NAAQS to be "neither arbitrary nor capricious," and denying the remaining petitions for review.[7]

In response to the D.C. Circuit Court's remand to consider the potential beneficial health effects of O_3 pollution in shielding the public from the effects of solar (UV) radiation, on November 14, 2001, EPA proposed to leave the 1997 8-hr NAAQS unchanged (66 FR 52768). After considering public comment on the proposed decision, EPA reaffirmed the 8-hr O_3 NAAQS set in 1997 (68 FR 614). Finally, on April 30, 2004, EPA announced the decision to make the 1-hr O_3 NAAQS no longer applicable to areas one year after the effective date of the designation of those areas for the 8-hr NAAQS (69 FR 23966). For most areas, the date that the 1-hr NAAQS no longer applied was June 15, 2005. (See 40 CFR 50.9 for details.)

1.2.4 Current Ozone NAAQS Review

EPA initiated the current NAAQS review in September 2000 with a call for information (65 FR 57810). A project work plan (U.S. EPA, 2002) for the preparation of the CD was released in November 2002 for CASAC and public review. EPA held a series of workshops in mid-2003 on several draft chapters of the CD to obtain broad input from the relevant scientific communities. These workshops helped to inform the preparation of the first draft CD (U.S. EPA, 2005a), which was released for CASAC and public review on January 31, 2005.

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[5] *American Trucking Associations v. EPA*, 175 F.3d 1027 (D.C. Cir., 1999).

[6] *Whitman v. American Trucking Associations*, 531 U.S. 457 (2001).

[7] *Whitman v. American Trucking Associations*, 283 F.3d 355 (D.C. Cir. 2002).

During the process of preparing the first draft CD, NCEA revised the planned format of the CD described in the 2002 work plan. These revisions were made as part of a collaborative effort with OAQPS staff to modify the review process so as to enhance the Agency's ability to meet this and future NAAQS review schedules. As described in Chapter 1 of the first draft CD, emphasis is placed on interpretative evaluation and integration of evidence in the main body of the document, with more detailed descriptions of individual studies being provided in a series of accompanying annexes. This change is intended to streamline the document so as to facilitate timely CASAC and public review and to focus more clearly on issues most relevant to the policy decisions to be made by the Administrator. The modified review process envisions that key policy-relevant issues will be identified earlier in the review process through enhanced collaboration between NCEA and OAQPS, leading to a more efficient linkage between the CD and the Staff Paper. At the CASAC meeting held on May 4-5, 2005, to review the first draft CD, this new format for the CD was met with general approval of CASAC and the public. A second draft CD (EPA, 2005b) was released for CASAC and public review on August 31, 2005. In a February 16, 2006 letter to the Administrator, CASAC offered final comments on all chapters of the CD (Henderson, 2006a), and the final CD was released on March 21, 2006. In a June 8, 2006 letter to the Administrator, CASAC offered additional advice to the Agency concerning Chapter 8 of the final CD (Integrative Synthesis) to help inform the second draft Staff Paper (Henderson, 2006b). The second draft Staff Paper was released on July 17, 2006 and reviewed by the CASAC Ozone Panel on August 24 and 25, 2006. In an October 24, 2006 letter to the Administrator, CASAC provided advice and recommendations to the Agency concerning the second draft Staff Paper (Henderson, 2006c). Advice and recommendations from CASAC as well as public comments have been taken into account in preparing this final Staff Paper.

The schedule for completion of this review is governed by a consent decree resolving a lawsuit filed in March 2003 by a group of plaintiffs representing national environmental organizations, alleging that EPA had failed to complete the current review within the period provided by statute. *American Lung Association v. Whitman* (No. 1:03CV00778, D.D.C. 2003). The modified consent decree that now governs this review, entered by the court on December 16, 2004, provides that EPA sign for publication notices of proposed and final rulemaking concerning its review of the O_3 NAAQS no later than March 28, 2007 and December 19, 2007, respectively. This consent decree was further modified in October 2006 to change these proposed and final rulemaking dates to no later than May 30, 2007 and February 20, 2008, respectively. The EPA expects that these dates for signing the publication notices of proposed and final rulemaking will now be extended to no later than June 20, 2007 and March 12, 2008, respectively.

1.3 GENERAL APPROACH AND ORGANIZATION OF THE DOCUMENT

The policy assessment in this Staff Paper is based on staff's evaluation of the policy implications of the scientific evidence contained in the CD and results of quantitative analyses based on that evidence, as well as the views presented by CASAC and various stakeholders. Taken together, this information informs various conclusions and the identification of a range of options on certain elements of the O_3 standards under review. While the CD focuses on new scientific information available since the last review, it appropriately integrates that information with scientific criteria from previous reviews. The quantitative analyses presented in this Staff Paper (and described in more detail in technical support documents) are based on the most recently available air quality information, so as to provide current characterizations of O_3 air quality patterns and estimated health and environmental risks related to exposure to ambient O_3 concentrations.

Following this introductory chapter, the Staff Paper is organized into three main parts: the characterization of ambient O_3 air quality data; O_3-related health effects and primary O_3 NAAQS; and O_3-related welfare effects and secondary O_3 NAAQS. The content of these parts is discussed more fully below.

The characterization of ambient O_3 and related photochemical oxidants is presented in Chapter 2 and includes information on O_3 properties, current O_3 air quality patterns, historic trends, and background levels. This chapter provides a frame of reference for subsequent discussion of current and alternative O_3 NAAQS and alternative forms of O_3 standards.

Chapters 3 through 6 comprise the second main part of this Staff Paper dealing with human health and primary standards. Chapter 3 presents an overview of key policy-relevant health effects evidence, major health-related conclusions from the CD, and an examination of issues related to the quantitative assessment of evidence from controlled human exposure and epidemiological studies. Chapters 4 and 5 describe the scope and methods used in conducting human exposure and health risk assessments and present results from those assessments. Chapter 6 includes staff conclusions and policy recommendations on the adequacy of the current primary standard and on an appropriate range of alternative primary standards for the Administrator's consideration, together with a discussion of the science and public health policy judgments underlying such standards.

Chapters 7 and 8 comprise the third main part of this Staff Paper. Chapter 7 presents a policy-relevant assessment of O_3 welfare effects evidence and discusses the scope and methods used in conducting vegetation-related exposure and risk assessments. Chapter 8 includes staff conclusions and policy recommendations on the adequacy of the current secondary standard and on an appropriate range of alternative secondary standards that for the Administrator's

consideration, together with a discussion of the science and public welfare policy judgments underlying such standards.

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2. AIR QUALITY CHARACTERIZATION

2.1 INTRODUCTION

This chapter generally characterizes ambient ozone (O_3) and related photochemical oxidants in terms of measurement methods, recent concentrations and trends, relationships between different air quality indicators, and estimates of policy-relevant background. This information is useful for interpreting the available exposure, health, and welfare effects information, and for evaluating the adequacy of the current primary and secondary standards for O_3 and developing options for alternative standards. The information presented in this chapter was drawn from the 2006 Criteria Document (CD) and additional analyses of data from various O_3 monitoring networks.

This chapter particularly focuses on 1-hr, 8-hr, and 24-hr average concentrations metrics in characterizing urban O_3 air quality because these are the metrics most frequently used in the health effect studies discussed in the CD and Chapter 3 of this Staff Paper. For the vegetation exposure and risk assessment discussed in Chapter 7 of this Staff Paper, both the current secondary standard 8-hr. metric and the cumulative, concentration-weighted metrics, SUM06 and W126 are used.

Although this chapter focuses on 2002-2004 air quality data in order to be consistent with the CD, recent observations demonstrates the 2005 O_3 data show that national ambient concentrations have decreased 20% since 1980 and 8% since 1990. 2005 concentrations were the second lowest on record with only 2004 levels being lower by 5%. Meteorological conditions for 2005 were similar to those observed in 2002 which was much hotter than 2004 and, therefore, more conducive to O_3 formation. Furthermore, the O_3 levels in 2005 were approximately 9% lower than those seen in 2002 for sites east of 100 degrees west longitude. One explanation for this difference can possibly be attributed to the implementation of the NO_x SIP Call[1] which occurred in 2002 for many states east of the Mississippi River.

2.2 CHEMICAL AND PHYSICAL PROPERTIES, FORMATION, AND TRANSPORT

2.2.1 Properties and Formation

The atmosphere can be divided into several distinct vertical layers, based primarily on the major mechanisms by which they are heated and cooled. The lowest major layer is the troposphere, which extends from the earth's surface to about 8 km above the surface in polar

[1] EPA's rule, known as the NO_x SIP Call, was designed to reduce regional transport of O_3 and O_3-forming pollutants in the eastern half of the United States by requiring 21 states to reduce O_3- season NO_x emissions that contribute to nonattainment in other states.

regions and to about 16 km above the surface in tropical regions. The planetary boundary layer (PBL) is the lower sub-layer of the troposphere, extending from the surface to about 1 or 2 km, and is most strongly affected by surface conditions. The stratosphere extends from the top of the troposphere, to about 50 km in altitude. The emphasis in this chapter is placed on concentrations of O_3 occurring in the lower troposphere, in particular in the PBL (CD, p.2-1).

Ozone chemistry in the presence of sunlight, nitrogen oxides (NO_x) and volatile organic carbon (VOC) is well understood and a central component of modern air quality models. The chemical formation of O_3 in the troposphere results from the oxidation of nitric oxide (NO) to nitrogen dioxide (NO_2) by organic (RO_2) or hydro-peroxy (HO_2) radicals. Photolysis (the chemical process of breaking down molecules into smaller units through the absorption of light) of NO_2 yields NO and a ground-state oxygen atom, $O(^3P)$, which then reacts with molecular oxygen to form O_3 (CD, p.2-2).

Oxidized nitrogen compounds are emitted to the atmosphere mainly as NO, which is oxidized to NO_2 which subsequently can be reduced back to NO. Consequently, NO and NO_2 are often grouped together into their own family called NO_x (CD, p.2-3). Oxidized nitrogen containing compounds are essential to the formation of O_3 in the air. There are a large number of oxidized nitrogen containing compounds in the atmosphere including NO, NO_2, nitrate (NO_3), nitrous acid (HNO_2), nitric acid (HNO_3), nitrogen pentoxide (N_2O_5), pernitric acid (HNO_4), peroxy acetyl nitrate (PAN) and its homologues, other organic nitrates and particulate nitrate. Collectively these species are referred to as NO_y. NO_x is considered a good surrogate for NO_y and, thus, is commonly monitored and reported (see Table 2-1).

In urban areas, both biogenic and anthropogenic VOCs are important for O_3 formation. Table 2-2 lists a variety of VOC sources (see http://www.epa.gov/airtrends/econ-emissions.html). The categories in the table are self explanatory with the exception of the fires and miscellaneous categories. The fires category includes both wild fires and prescribed burns. The miscellaneous category includes mainly structural fires and sources from agricultural activities. One category not in either table due to insufficient estimates is biogenic emissions. As can be seen in the table, highway vehicles have been the single largest source of anthropogenic VOC emissions over the years ranging from about 49% of total emissions in 1970 to about 27% of total emissions in 2004. Starting in 2001, solvent use and highway vehicles were the two main sources of VOCs with roughly equal contributions to the total emissions.

In non-urban, vegetated areas, biogenic VOCs emitted from the vegetation tend to be the most important. In the remote troposphere, CH_4 and CO are the main carbon-containing precursors to O_3 formation. In coastal environments and other selected environments, atomic Cl and Br radicals can also initiate the oxidation of VOCs (CD, p.2-2 and 2-3).

Table 2-1. NOx Emission Sources, 1970-2004

	Nitrogen Oxides (NOx) National Emissions Totals (thousands of tons)								
Source Category	**1970**	**1975**	**1980**	**1985**	**1990**	**1991**	**1992**	**1993**	**1994**
FUEL COMB. ELEC. UTIL.	4,900	5,694	7,024	6,127	6,663	6,519	6,504	6,651	6,565
FUEL COMB. INDUSTRIAL	4,325	4,007	3,555	3,209	3,035	2,979	3,071	3,151	3,147
FUEL COMB. OTHER	836	785	741	712	1,196	1,281	1,353	1,308	1,303
CHEMICAL & ALLIED PRODUCT MFG	271	221	213	262	168	165	163	155	160
METALS PROCESSING	77	73	65	87	97	76	81	83	91
PETROLEUM & RELATED INDUSTRIES	240	63	72	124	153	121	148	123	117
OTHER INDUSTRIAL PROCESSES	187	182	205	327	378	352	361	370	389
SOLVENT UTILIZATION	0	0	0	2	1	2	3	3	3
STORAGE & TRANSPORT	0	0	0	2	3	6	5	5	5
WASTE DISPOSAL & RECYCLING	440	159	111	87	91	95	96	123	114
HIGHWAY VEHICLES	12,624	12,061	11,493	10,932	9,592	9,449	9,306	9,162	9,019
OFF-HIGHWAY	2,652	2,968	3,353	3,576	3,781	3,849	3,915	3,981	4,047
MISCELLANEOUS	330	165	248	310	369	286	255	241	390
MISCELLANEOUS	NA	NA	NA	NA	NA	NA	NA	NA	NA
TOTAL	26,883	26,377	27,079	25,757	25,529	25,179	25,260	25,357	25,349
Total without FIRES	26,883	26,377	27,079	25,757	25,167	24,932	25,026	25,123	24,967
FIRES	NA	NA	NA	NA	362	247	234	234	382

Table 2-1. NOx Emission Sources, 1970-2004 (cont'd)

Nitrogen Oxides (NO_x)
National Emissions Totals (thousands of tons)

Source Category	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
FUEL COMB. ELEC. UTIL.	6,384	6164	6276	6232	5721	5330	4917	4699	4270	3740
FUEL COMB. INDUSTRIAL	3,144	3151	3101	3050	2709	2723	2757	2870	2870	2870
FUEL COMB. OTHER	1,298	1197	1177	1101	768	766	779	725	725	725
CHEMICAL & ALLIED PRODUCT MFG	158	125	127	129	102	105	107	105	105	105
METALS PROCESSING	98	83	89	89	86	89	94	84	84	84
PETROLEUM & RELATED INDUSTRIES	110	139	143	143	120	122	124	149	149	149
OTHER INDUSTRIAL PROCESSES	399	433	460	467	451	479	504	487	487	487
SOLVENT UTILIZATION	3	2	3	3	4	4	4	8	8	8
STORAGE & TRANSPORT	6	15	16	16	14	15	16	16	16	16
WASTE DISPOSAL & RECYCLING	99	153	157	163	162	129	130	152	152	152
HIGHWAY VEHICLES	8,876	8733	8792	8619	8371	8394	7774	7365	7365	7365
OFF-HIGHWAY	4,113	4179	4178	4156	4084	4167	4156	4086	4086	4086
MISCELLANEOUS	267	412	187	179	251	276	184	356	356	356
MISCELLANEOUS	NA	0	0	0	0	0	0	0	0	0
TOTAL	24,956	24787	24705	24348	22845	22598	21549	21102	20672	20142
FIRES	258	405	179	172	236	263	171	341	341	341
Total without FIRES	24,698	24,382	24,526	24,176	22,609	22,335	21,378	20,761	20,331	19,801

Table 2-2. VOC Emission Sources, 1970-2004

Volatile Organic Compounds (VOC)
National Totals (thousands of tons)

Source Category	1970	1975	1980	1985	1990	1991	1992	1993	1994
FUEL COMB. ELEC. UTIL.	30	40	45	32	47	44	44	45	45
FUEL COMB. INDUSTRIAL	150	150	157	134	182	196	187	186	196
FUEL COMB. OTHER	541	470	848	1,403	776	835	884	762	748
CHEMICAL & ALLIED PRODUCT MFG	1,341	1,351	1,595	881	634	710	715	701	691
METALS PROCESSING	394	336	273	76	122	123	124	124	126
PETROLEUM & RELATED INDUSTRIES	1,194	1,342	1,440	703	611	640	632	649	647
OTHER INDUSTRIAL PROCESSES	270	235	237	390	401	391	414	442	438
SOLVENT UTILIZATION	7,174	5,651	6,584	5,699	5,750	5,782	5,901	6,016	6,162
STORAGE & TRANSPORT	1,954	2,181	1,975	1,747	1,490	1,532	1,583	1,600	1,629
WASTE DISPOSAL & RECYCLING	1,984	984	758	979	986	999	1,010	1,046	1,046
HIGHWAY VEHICLES	16,910	15,392	13,869	12,354	9,388	8,860	8,332	7,804	7,277
OFF-HIGHWAY	1,616	1,917	2,192	2,439	2,662	2,709	2,754	2,799	2,845
MISCELLANEOUS	1,101	716	1,134	566	1,059	756	486	556	720
TOTAL	34,659	30,765	31,106	27,404	24,108	23,577	23,066	22,730	22,569
FIRES	917	587	1,024	465	983	678	407	478	638
Total without FIRES	33,742	30,178	30,082	26,939	23,125	22,899	22,659	22,252	21,931

Table 2-2. VOC Emission Sources, 1970-2004 (cont'd)

Volatile Organic Compounds (VOC)
National Totals (thousands of tons)

Source Category	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
FUEL COMB. ELEC. UTIL.	44	50	52	56	54	62	61	52	52	52
FUEL COMB. INDUSTRIAL	206	179	175	174	172	173	176	170	170	170
FUEL COMB. OTHER	823	893	893	889	919	949	950	790	790	790
CHEMICAL & ALLIED PRODUCT MFG	660	388	388	394	251	254	262	214	214	214
METALS PROCESSING	125	73	78	78	66	67	71	69	69	69
PETROLEUM & RELATED INDUSTRIES	642	477	487	485	457	428	441	375	375	375
OTHER INDUSTRIAL PROCESSES	450	435	438	443	438	454	420	406	406	406
SOLVENT UTILIZATION	6,183	5477	5621	5149	5036	4831	5012	4692	4692	4692
STORAGE & TRANSPORT	1,652	1294	1328	1327	1237	1176	1192	1205	1205	1205
WASTE DISPOSAL & RECYCLING	1,067	509	518	535	487	415	420	457	457	457
HIGHWAY VEHICLES	6,749	6221	5985	5859	5681	5325	4952	4543	4543	4543
OFF-HIGHWAY	2,890	2935	2752	2673	2682	2644	2622	2688	2688	2688
MISCELLANEOUS	551	1940	816	718	791	733	532	883	883	883
TOTAL	22,041	20871	19530	18782	18270	17512	17111	16544	16544	16544
FIRES	464	1870	744	645	667	615	412	785	785	785
Total without FIRES	21,577	19001	18786	18136	17603	16898	16699	15759	15759	15759

The effects of sunlight on O_3 formation, aside from the role of solar radiation in meteorological processes, depend on its intensity and its spectral distribution. Intensity varies diurnally, seasonally, and with latitude, but the effect of latitude is strongest in the winter. Ultraviolet radiation from the sun plays a key role in initiating the photochemical processes leading to O_3 formation and affects individual photolytic reaction steps. However, there is little empirical evidence in the literature directly linking day-to-day variations in observed surface UV radiation levels with variations in tropospheric O_3 levels (CD, p.AX2-90).

2.2.2 Relationship of Ozone to Photochemical Oxidants

The relationship between O_3, other oxidants, and oxidation products is complex and involves many factors. Most notably, O_3 acts as a generator of hydroxyl radicals (OH) propagating a variety of integrated and cascading reactions yielding additional oxidizing species (e.g., hydroperoxy radical, HO_2; organic peroxy radicals, RO_2; hydrogen peroxide, H_2O_2; nitrogen dioxide, NO_2; many of which are short lived in the atmosphere. These "oxidizing" species would be expected, like O_3, to interfere with cellular processes leading to adverse health effects. Oxidants also drive the formation of particle bound sulfate, nitrate and organic carbon; which are principal components of secondarily formed $PM_{2.5}$; and influence positively the formation or reaction of toxic gases such as peroxy acetyl nitrate (PAN), aldehydes (e.g., formaldehyde, acetaldehyde, acrolein), organic nitrates and organic amines. Furthermore, deposition and subsequent adverse aquatic and terrestrial effects generally are enhanced through availability of reactive oxidant species. These "enhanced" deposition impacts include acidification, near field deposition of reactive mercury, and other nitrogen-based eutrophication effects (See Figure 2-1). Benefits of reducing O_3 extend beyond that associated with O_3 autonomously, given the role of O_3 as an oxidizing agent elevating levels of atmospheric components responsible for a broad range of adverse human health and environmental impacts.

2.2.3 Transport

The transport of O_3 and other secondary pollutants is determined by meteorological and chemical processes extending typically over spatial scales of several hundred kilometers (e.g., Civerolo et al., 2003; Rao et al., 2003). An analysis of the output of regional model studies conducted by Kasibhatla and Chameides (2000) suggests that O_3 can be transported over a few thousand kilometers in the upper boundary layer of the eastern half of the United States during specific O_3 episodes. Convection is capable of transporting O_3 and its precursors vertically through the troposphere as shown in Annex AX2.3.2 of the CD. Nocturnal low-level jets (LLJs)

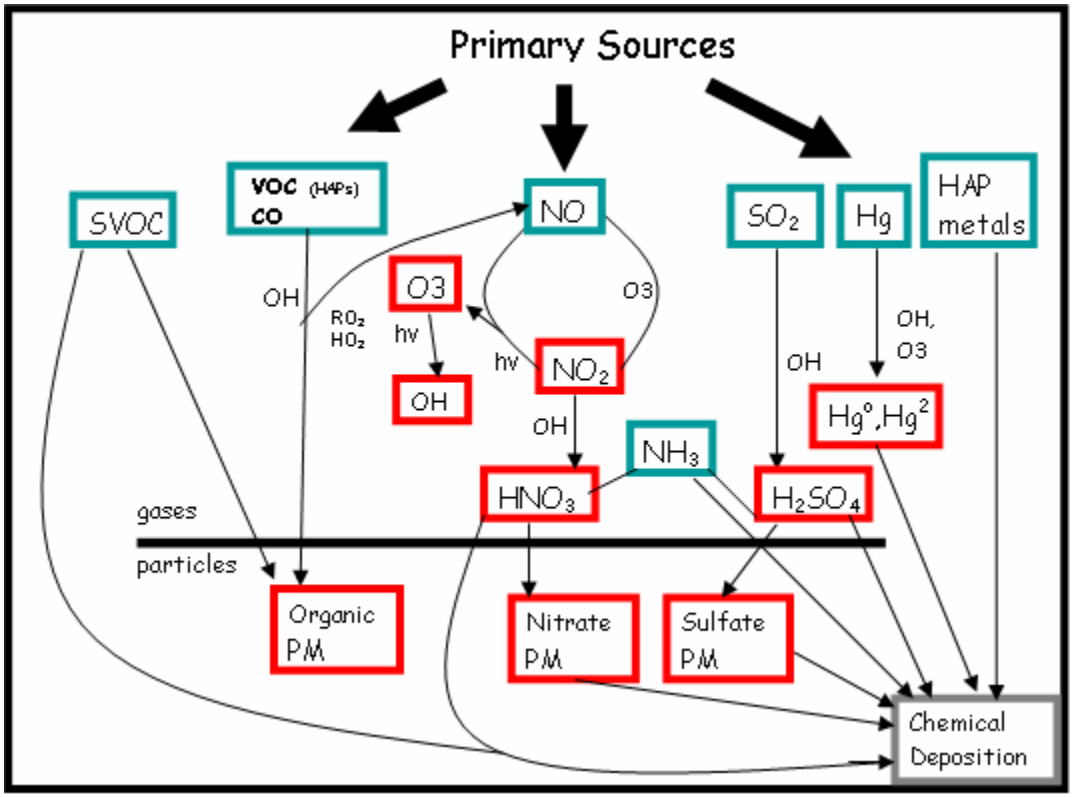

Figure 2-1. Atmospheric Processes Affecting the Formation of Photochemical Oxidants and Particulate Matter. Chemical links illustrating relationships across, criteria pollutants and HAPs including mercury, as well as connections across sources, secondarily formed species, gases, particulate matter and deposition. Primary emissions are distinguished from secondarily formed species. Note that this diagram is a highly condensed model that does not capture numerous various heterogeneous processes and complex chemical pathways. Key atmospheric species that are involved in many reactions across pollutant categories include O_3 and the hydroxyl radical, OH. Primary PM emissions are not included as they interact marginally with other other atmospheric species (adopted from NARSTO, 2002).

can also transport pollutants hundreds of kilometers over the mid-Atlantic region, the central U.S. and California (Zhang et al., 2001). Turbulence associated with LLJs can bring these pollutants to the surface and result in secondary O_3 maxima in the early morning in many locations. However, the presence of mountain barriers can limit both horizontal and vertical dispersion such as observed in Los Angeles and Mexico City resulting in a greater frequency and duration of days with high O_3 concentrations (CD, p.2-10).

2.3 DATA SOURCES

Two main sources of monitoring data were used for this assessment, the state-supplied data from various types of monitors housed in the Air Quality System (AQS) data base (which includes National Park Service monitors) and the Clean Air Status and Trends Network (CASTNET). The vegetation exposure analysis also uses an enhanced Veroni Neighborhood Average (eVNA) spatial interpolation technique to combine 2001 monitor data from both AQS and CASTNET with 2001 modeled data from the Community Multi-scale Air Quality (CMAQ) model. This interpolated surface is used to fill in the gaps left by a sparse rural monitoring network in the western United States.

Air quality models are often used to simulate the formation, transport, and decay of air pollution. The CMAQ modeling system is a comprehensive three-dimensional grid-based Eulerian air quality model designed to estimate O_3 and particulate concentrations and deposition over large spatial scales (Dennis et al., 1996; Byun and Ching, 1999; Byun and Schere, 2006). The CMAQ model is a publicly available, widely-used, peer-reviewed, state-of-the-science model consisting of a number of science attributes that are critical for simulating the oxidant precursors and nonlinear organic and inorganic chemical relationships associated with the formation of O_3, as well as sulfate, nitrate, and organic aerosols.

2.3.1 Air Quality System (AQS)

EPA's ambient air quality surveillance regulations are found at 40 CFR Part 58. Section 58.20 requires States to provide for the establishment of air quality surveillance systems in their State Implementation Plans (SIP). The air quality surveillance system consists of a network of monitoring stations designated as State and Local Air Monitoring Stations (SLAMS), which measure ambient concentrations of those pollutants for which standards have been established in 40 CFR Part 50. SLAMS, National Air Monitoring Stations (NAMS), which are a subset of SLAMS, and Photochemical Assessment Monitoring Stations (PAMS) must meet the requirements of 40 CFR Part 58, Appendices A (Quality Assurance Requirements), C (Ambient Air Quality Monitoring Methodology), D (Network Design Criteria), and E (Probe and Path Siting Criteria).

The Air Quality System (AQS) is EPA's repository of ambient air quality data. AQS stores data from over 10,000 monitors; 5000 of which are currently active. Of these, over 3000 measure and report O_3 concentration data (See Figure 2-2). These monitors make up the SLAMS, PAMS, NAMS, and other special purpose monitors used and operated by the States. AQS also contains meteorological data, descriptive information about each monitoring station (including its geographic location and its operator), and data quality assurance/quality control information. The Office of Air Quality Planning and Standards (OAQPS) and other AQS users rely upon the data system to assess air quality, assist in Attainment/Non-Attainment designations, evaluate State Implementation Plans for Non-Attainment Areas, perform modeling for permit review analysis, and other air quality management functions. AQS information is also used to prepare reports for Congress as mandated by the Clean Air Act (see http://www.epa.gov/ttn/airs/airsaqs/sysoverview.htm).

The NAMS/PAMS/SLAMS O_3 monitor network achieved an overall average bias (upper bound) of 0.2% and an overall mean precision of 3% for 2002. If special purpose and other O_3 monitors are also included the average upper bounds of bias and precision were 0.4% and 2.9% respectively (U.S. EPA 2004a).

2.3.2 CASTNET

CASTNET is the nation's primary source for data on dry acidic deposition and rural, ground-level O_3. Operating since 1987, CASTNET is used in conjunction with other national monitoring networks to provide information for evaluating the effectiveness of national emission control strategies. CASTNET consists of over 80 sites across the eastern and western United States (see Figure 2-2) and is cooperatively operated and funded with the National Park Service. In 1986, EPA established the National Dry Deposition Network (NDDN) to obtain field data on rural deposition patterns and trends at different locations throughout the United States. The network consisted of 50 monitoring sites that derived dry deposition data based on measured air pollutant concentrations and modeled dry deposition velocities estimated from meteorology, land use, and site characteristic data. In 1990, amendments to the Clean Air Act necessitated a long-term, national program to monitor the status and trends of air pollutant emissions, ambient air quality, and pollutant deposition. In response, EPA, in cooperation with the National Oceanic Atmospheric Administration (NOAA), created CASTNET from NDDN. In terms of data quality, CASTNET achieved 98% to 99% of all precision and accuracy audits being within the ±10% criteria for both precision and accuracy. Overall, CASTNET O_3 monitors are stable showing only very small variation (U.S. EPA 2003, p.22).

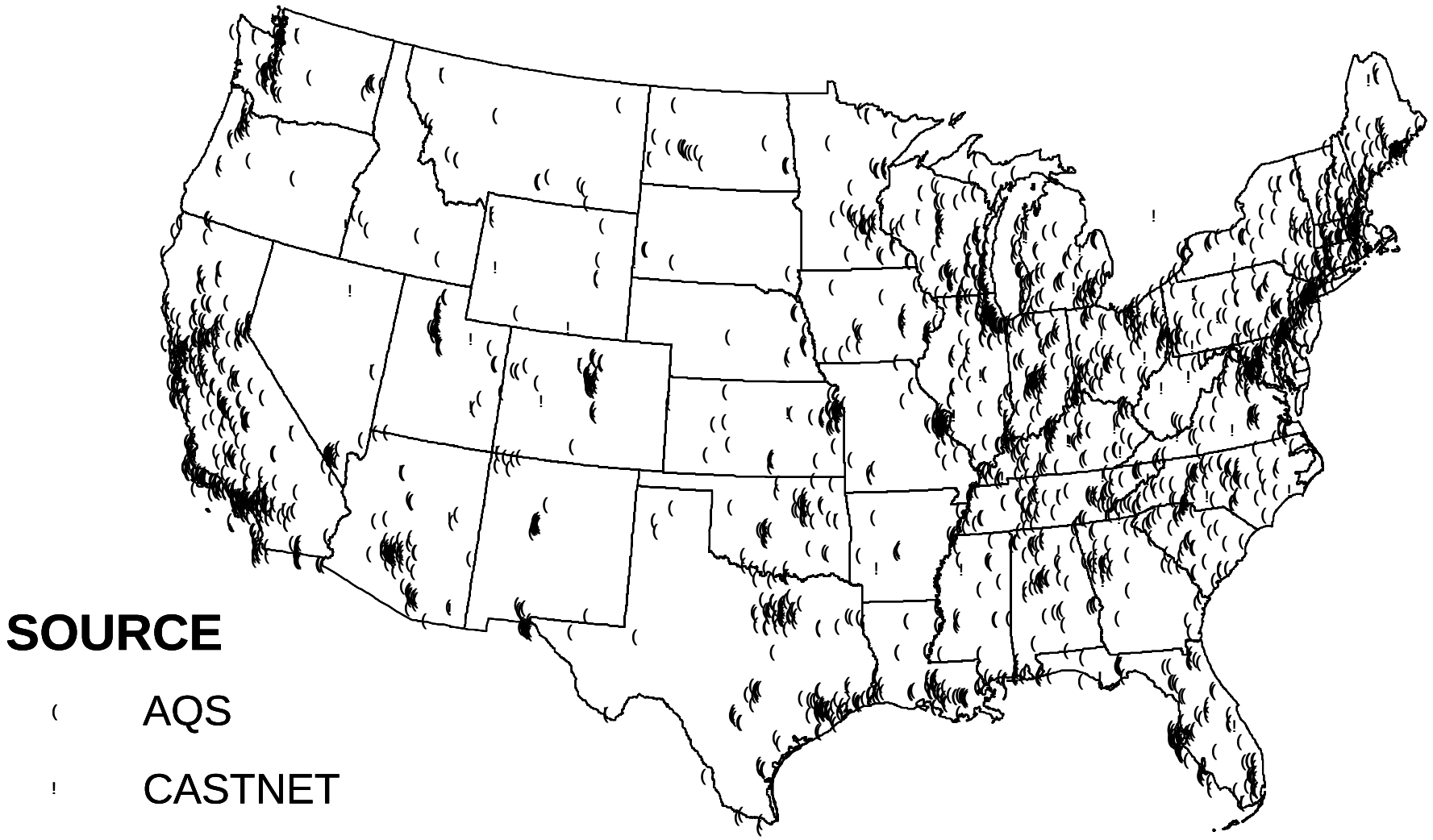

Figure 2-2. Locations of Ozone Monitors from AQS and CASTNET

2.4 OZONE MONITORING METHODS AND DATA QUALITY

2.4.1 Ozone Monitoring Methods

Ozone monitoring is conducted almost exclusively with UV absorption spectrometry with commercial short path instruments, a method that has been thoroughly evaluated in clean air. The ultimate reference method is a relatively long-path UV absorption instrument maintained under carefully controlled conditions at the National Institute of Standards and Technology (NIST) (CD, p.2-22). Most O_3 UV instruments reference the NIST method through a network of Standard Reference Photometers (SRPs) that are maintained and operated by EPA.

Several reports in the reviewed scientific literature have investigated interferences in O_3 detection via UV radiation absorption and chemiluminescence which is the reference method but almost never employed. These include the effects of water vapor, VOC's, aromatic compounds and their oxidation products, and other organic and inorganic compounds. Water vapor had no significant impact on UV absorption-based instruments, but could cause a positive interference of up to 9% in chemiluminescence-based detectors at high humidities (dew point of 24° C). Aromatic compounds and their oxidation products were found to generate a positive but small interference in the UV absorption instruments. However, when the results are applied to ambient concentrations of toluene and NO_x, the effect appears to be very minor (about 3 percent under the study conditions). Other organic and inorganic compounds displayed interferences, but not at levels likely to interfere with accurate determination of O_3 in an urban environment (CD, p.2-25).

Although not widely used, O_3 measurements by differential optical absorption spectroscopy (DOAS) at a variety of wavelengths in the UV and visible parts of the spectrum. Comparisons of DOAS results to those from a UV absorption instrument showed good agreement on the order of 10%. Researchers have reported a positive interference due to an unidentified absorber in the 279 to 289 nm spectral region used by many commercial short-path DOAS systems for the measurement of O_3. Results of that study suggest that compounds from wood burning, used for domestic heating, may be responsible (CD, p.AX2-149).

2.4.1 Effect of Measurement Precision on 8 hour Ozone Averages

Staff conducted an analysis to determine the precision of an 8-hr averaged O_3 concentration (Cox and Camalier, 2006). Daily maximum 8-hr O_3 values were simulated using a Weibull distribution to yield a "true" three-year averaged O_3 design value without the influence of measurement error.

From 2002 to 2004, the average precision in the collected O_3 measurements is approximately 3%. This means, for example, that a 1-hr measured concentration of 0.100 ppm could be between 0.097 ppm and 0.103 ppm. Utilizing precision data from 900 O_3 monitors for

the 2002 through 2004 O_3 seasons, a second set of 8-hr O_3 concentrations was generated to incorporate the precision data from the O_3 monitoring network to account for instrument measurement error. The result was a value which reflected the "true" O_3 design value plus measurement error. The difference between the value with measurement error and the "true value" reflects the impact of the instrument measurement error on the calculated 8-hr design value.

The exercise was repeated 1000 times and the differences between the two previously described design values were summarized. Table 2-3 shows the results of the analysis. The percentiles presented in the table reflect the percentage of sites at or below the corresponding 1-hour precision value. The table shows that even at a precision of approximately 4.5%, of which 95% of the O_3 sites are at or below, the standard deviation of the difference between the 8-hr design values is less than 0.001 ppm.

A second exercise was performed to incorporate systematic bias error which includes the instrument drift, noise, precision and calibration error associated with the UV absorption method. It was assumed that each 8-hr measurement was subjected to this randomly occurring bias which had an average of zero and a standard deviation of approximately 0.004 ppm. The mean and standard deviation utilized for the simulation were believed to be reasonable estimates for monitors operating under normal conditions. The results of this exercise show that assuming a random bias of 0.004 ppm produced an uncertainty in the 8-hr design value of approximately 0.001 ppm. This analysis supports expressing the level of the standard to the nearest thousandth (three decimal places) part per million (ppm), which is equivalent to the nearest part per billion (ppb).[2] The State of California also reached a similar conclusion regarding the precision of the existing ozone monitoring methodology (California Environmental Protection Agency, 2005).

2.5 CHARACTERIZATION OF GROUND-LEVEL OZONE CONCENTRATIONS

2.5.1 Metrics

This section characterizes ground level O_3 concentrations based on several metrics. Two daily maximum statistics, 1-hr and 8-hr averages, and one daily average statistic in the form of a 24-hr concentration, and two cumulative concentration weighted statistics, SUM06 and W126, are summarized to show how O_3 varies over space and time. The 1-hr and 8-hr daily maximum averaging times reflect the former and current O_3 standards, and much of the health effects literature for O_3 has focused on effects associated with these averaging times. The 24-hr daily

[2]Under the current standard, a rounding convention is used to determine attainment where the design value is rounded to the nearest 0.01 ppm. A National Ambient Air Quality Standard expressed to the nearest 0.001 ppm would mean that the current rounding conventions become trivial. However, it is envisioned that the data handling guideline within the current regulation where digits past the third decimal place are truncated would be retained.

Table 2-3. Relationship between Precision of 1-hour Ozone Data and Corresponding Standard Deviation of 8-hour Design Values

Precision of 1- hour ozone value (%)	Nationwide Percentile	Standard Deviation of Difference in DV's (ppb)
1.63	25	0.27
2.22	50	0.34
2.97	75	0.45
3.89	90	0.57
4.52	95	0.63

average has been used for several personal exposure studies (CD, pp.3-72 – 74). The SUM06 and W126 have been used frequently in the scientific literature and CD in studying and assessing the relationship between O_3 exposures and adverse effects on vegetation. The daily maximum 8-hr values are found by first calculating running or moving 8-hr values for all 24 hours in a day (for example, averaging the 1-hr concentrations from 1:00am to 8:00 am, then average the 1-hr values from 2:00am to 9:00 am, etc.). Then the maximum value for each day is found (note that any 8-hr time period that starts in a day is assigned to that day). On an annual basis, the fourth highest of these values is summarized. The daily maximum 1-hr statistic is the maximum value of all 1-hr values in a day. On an annual basis, the second highest of these values in a year is summarized. The 24-hr average is a mean of the 24 individual hourly concentrations measured from midnight to midnight.

The maximum, 3 month, 12 hour SUM06 statistic is calculated by cumulating all 1-hr values greater than or equal to 0.06ppm that occur during the12 hour daytime window (8:00am to 8:00pm Local Standard Time). For each month of the O_3 monitoring season, the largest consecutive 3-month sum of the daily values is calculated according to the secondary standard proposed in 1996 (61 FR 65638), but not adopted in 1997 (62 FR 38856). The SUM06 has a weighting function that is 0 when the concentration is less than 0.06 and is 1.0 when the concentration is greater than or equal to 0.06. The W126 seasonal cumulative statistics is calculated similarly to the SUM06 statistic. The only difference is the weighting function where the W126 statistic is a continuous, sigmoidal weighting function with an inflections point between 0.06ppm and 0.07ppm (Lefohn and Runeckles, 1987).

2.5.2 Spatial Variability

This section characterizes the spatial variability of O_3 based on all the metrics discussed above. Spatial variability is based on maps displaying county levels of the various metrics. In this way different levels of O_3 for different areas of the country are displayed. It should be noted that county areas can be much larger in the West than in the East, but monitors are not spread evenly within a county. As a result, the assigned concentration range might not represent conditions throughout a particular county.

2.5.2.1 Distributions of 1-hr, 8-hr, and 24-hr Ozone Metrics

Figures 2-3 to 2-5 show the distributions for measured 1-hr, 8-hr, and 24-hour daily average O_3 concentrations for 12 major urban areas in the United States. The Los Angeles area clearly has a distribution which is different from the other 11 cities, in that the hourly concentration interquartile range is within 0.057 to 0.089 ppm as opposed to the next highest interquartile range of Sacramento where 50% of the hourly concentrations lie between 0.056 and 0.079 ppm. In comparison, Houston which also has several 1-hr concentrations greater than

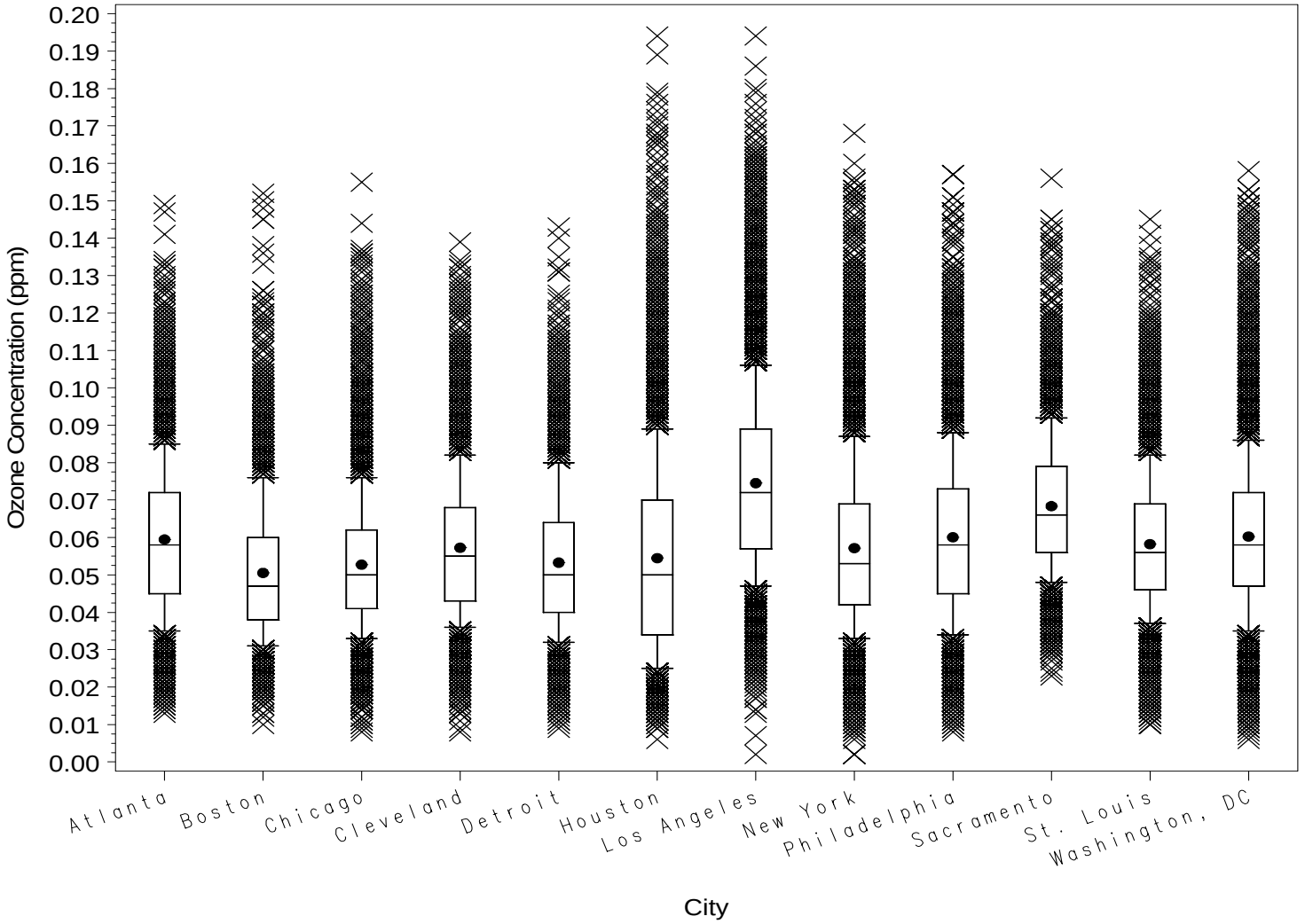

Figure 2-3. 1-hr Ozone Distributions across 12 Risk Areas, 2002-2004. Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; and 'x' depicts values outside the 10th and 90th percentile.
Data Source: AQS

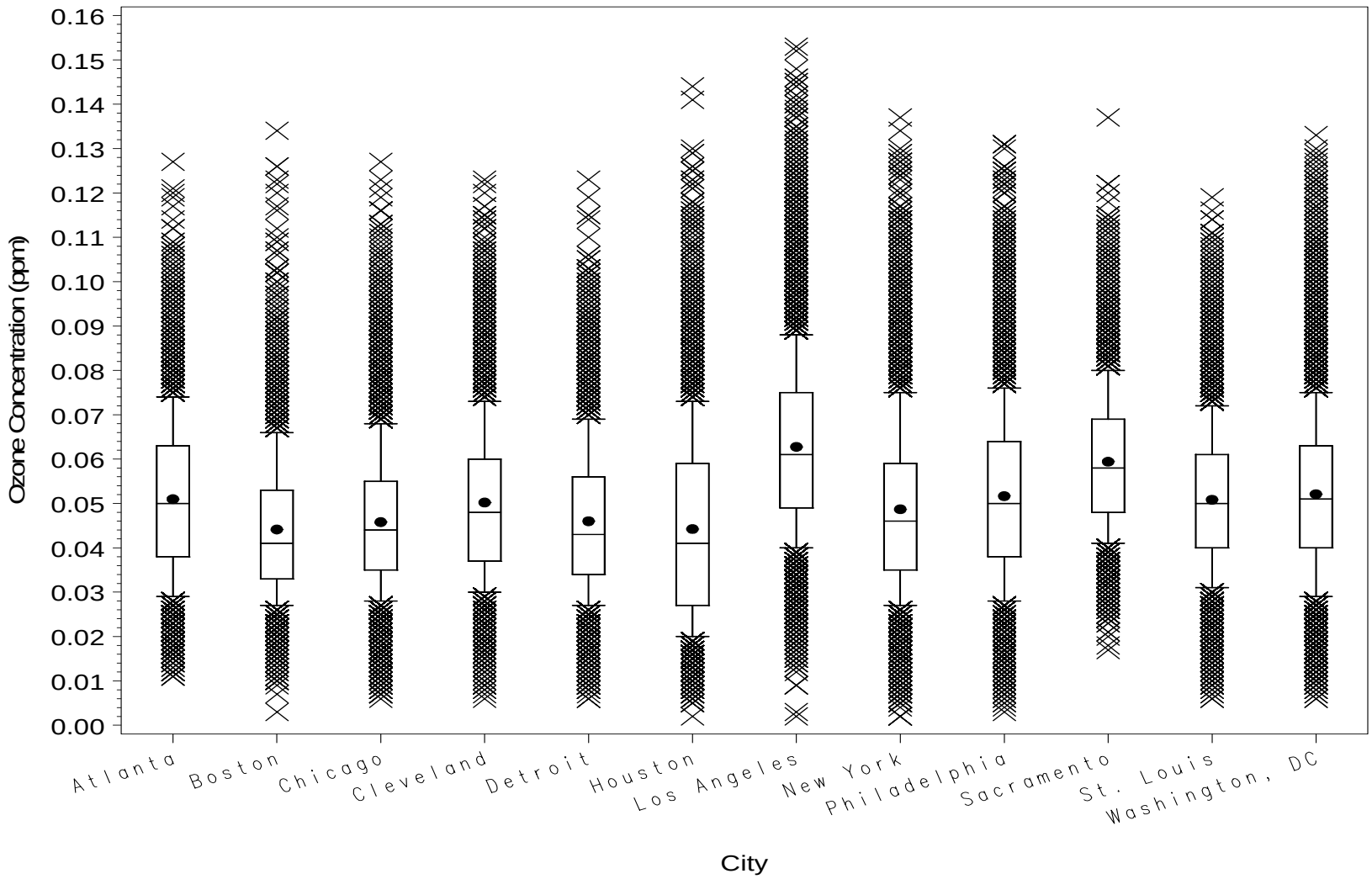

Figure 2-4. 8-hr Ozone Distributions across 12 Risk Areas, 2002-2004. Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; and 'x' depicts values outside the 10th and 90th percentile.
Data Source: AQS

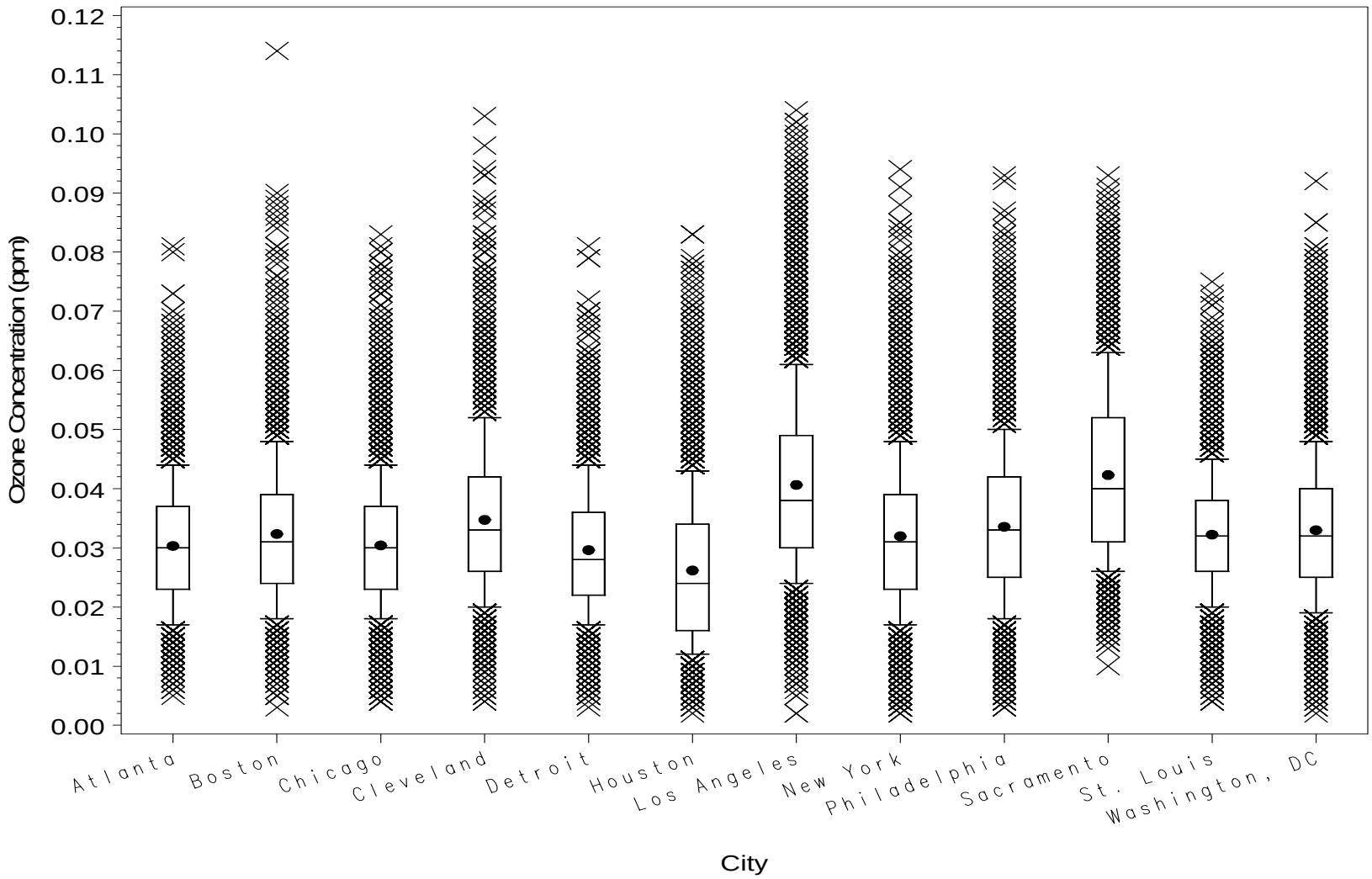

Figure 2-5. 24-hr Ozone Distributions across 12 Risk Areas, 2002-2004. Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; and 'x' depicts values outside the 10th and 90th percentile.
Data Source: AQS

0.125 ppm has a lower interquartile range of 0.034 to 0.07 ppm with 10% of its hourly values greater than 0.089 ppm as opposed to approximately 0.106 ppm for Los Angeles. Houston also has a larger interquartile range of 0.036 ppm when compared to the average of the remaining 11 cities of 0.025 ppm. This trend is also observed in the 8-hr averaged concentrations. The remaining 9 cities exhibit similar distributions to one another for all three metrics (1-hr, 8-hr, and 24-hr).

For the 24-hour daily averaged concentration distributions, Houston shows a lower 75th percentile than the other cities with areas like Cleveland, Philadelphia and New York having higher distributions. The lower 24 hour concentrations in Houston indicate a wider range between the daily O_3 minima and maxima unlike an area like Cleveland, which has a higher interquartile range.

2.5.2.2 8-Hour and 1-Hour Statistics

High 8-hr average O_3 concentrations tend to occur near larger urban areas exhibiting similar patterns as corresponding 1-hr concentrations (see Figure 2-6). Elevated 8-hr levels occurring in smaller urban and non-urban areas are most likely caused by transport. Higher 8-hr O_3 levels observed in smaller urban and non-urban areas are most obvious at the end of the northeast corridor (the highly urbanized area running from Washington, DC to Boston, MA), North-central New York, and the Northern coast of Lake Michigan. Some of the highest levels occur not only in California, but also in Texas as well as some counties in the Northeast Corridor and isolated counties in the East (see Figure 2-7) (Fitz-Simons, et al., 2005). The highest 1-hr levels occur in California. (Fitz-Simons, et al., 2005).

2.5.2.3 Cumulative Concentration-Weighted Statistics

The highest SUM06 and W126 levels in 2001 (most of the analyses in Chapter 7 center on 2001 data) occurred in most of the agricultural areas of California. When the data were from CASTNET sites, more purely rural counties showed higher values (See Figures 2-8 through 2-11) (Fitz-Simons, et al., 2005). The SUM06 and W126 values experienced a sharp decline in 2004 when compared to 2002 primarily in the eastern part of the United States (See Figure 2-12 through 2-15). Although there were reductions in the West, the decreases in the East were more substantial. As discussed in section 2.1, the overall reductions across the country could possibly be due to lower temperatures experienced during the O_3 season. However, the eastern half of the country was also subject to the emission control requirements implemented under the NO_x SIP Call which occurred after 2002. The improvements seen in 2004 for the East are most likely due to a combination of cooler weather, the emission reductions from the NO_x SIP Call, and emission reductions from mobile source and other stationary source rules.

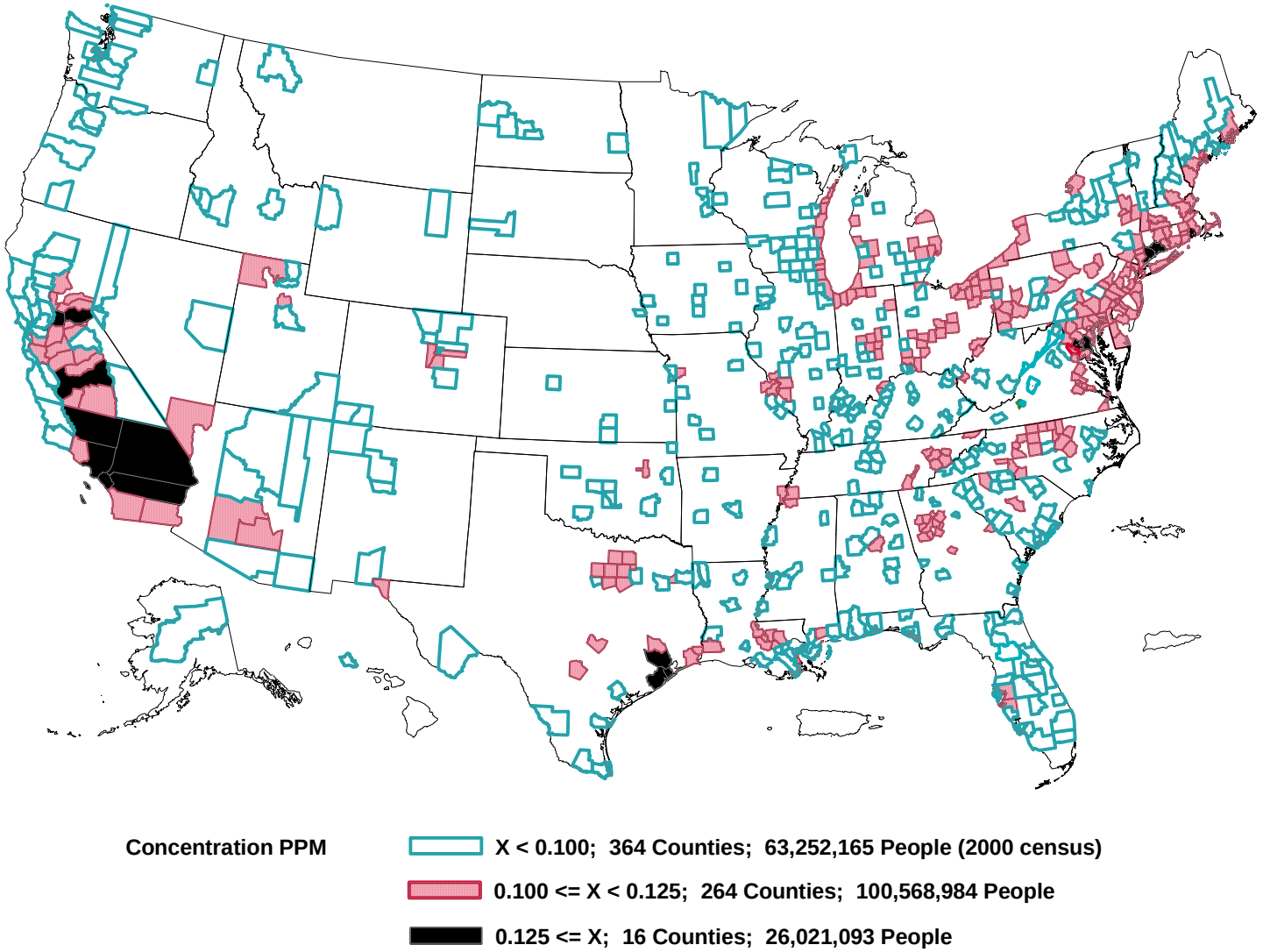

Figure 2-6. Average 2nd Highest Daily Maximum 1-hour Values in U.S. Counties, 2002-2004 AQS Data.

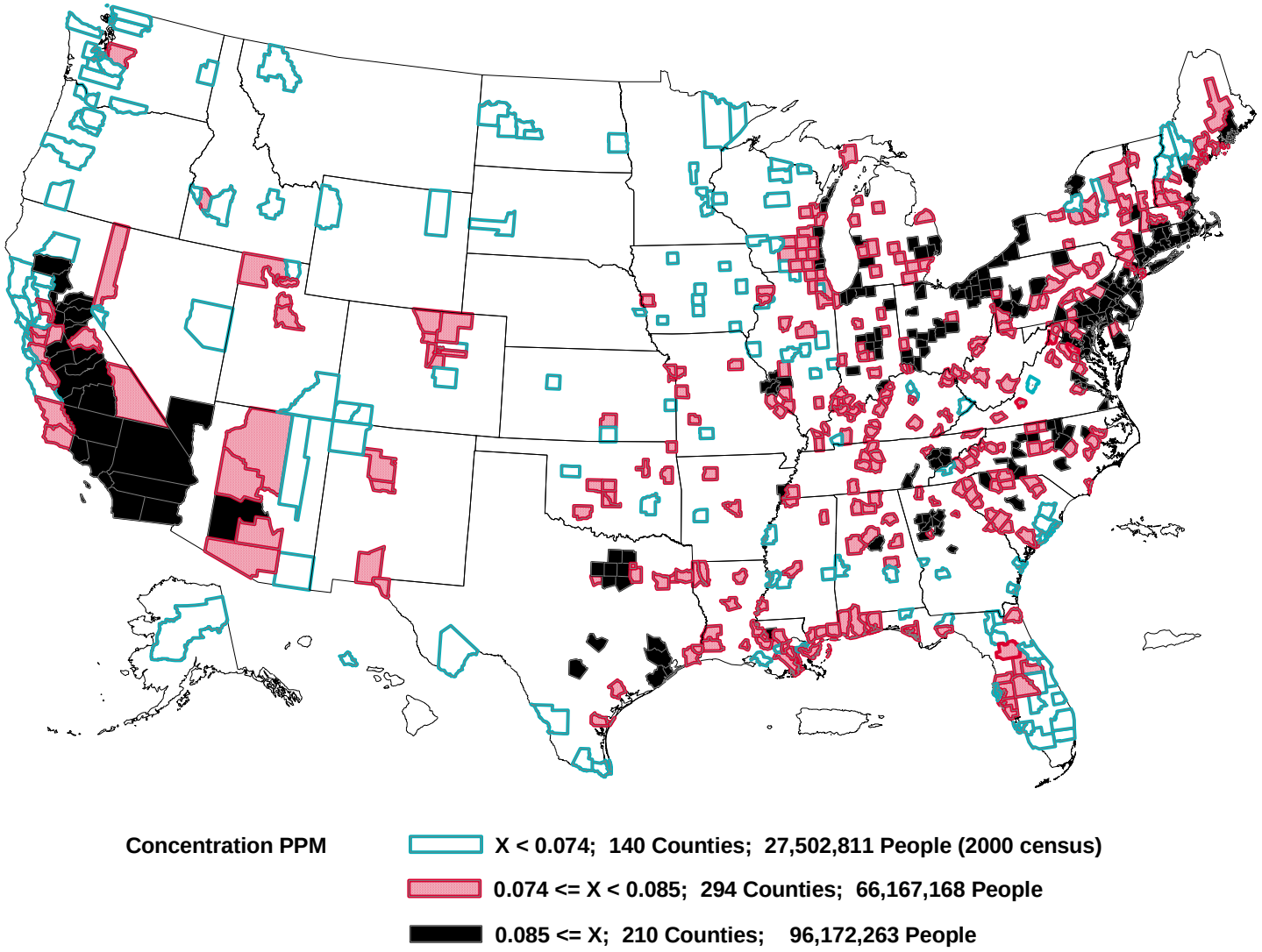

Figure 2-7. Average 4th Highest Daily Maximum 8-hour Values in U.S. Counties, 2002-2004 AQS Data.

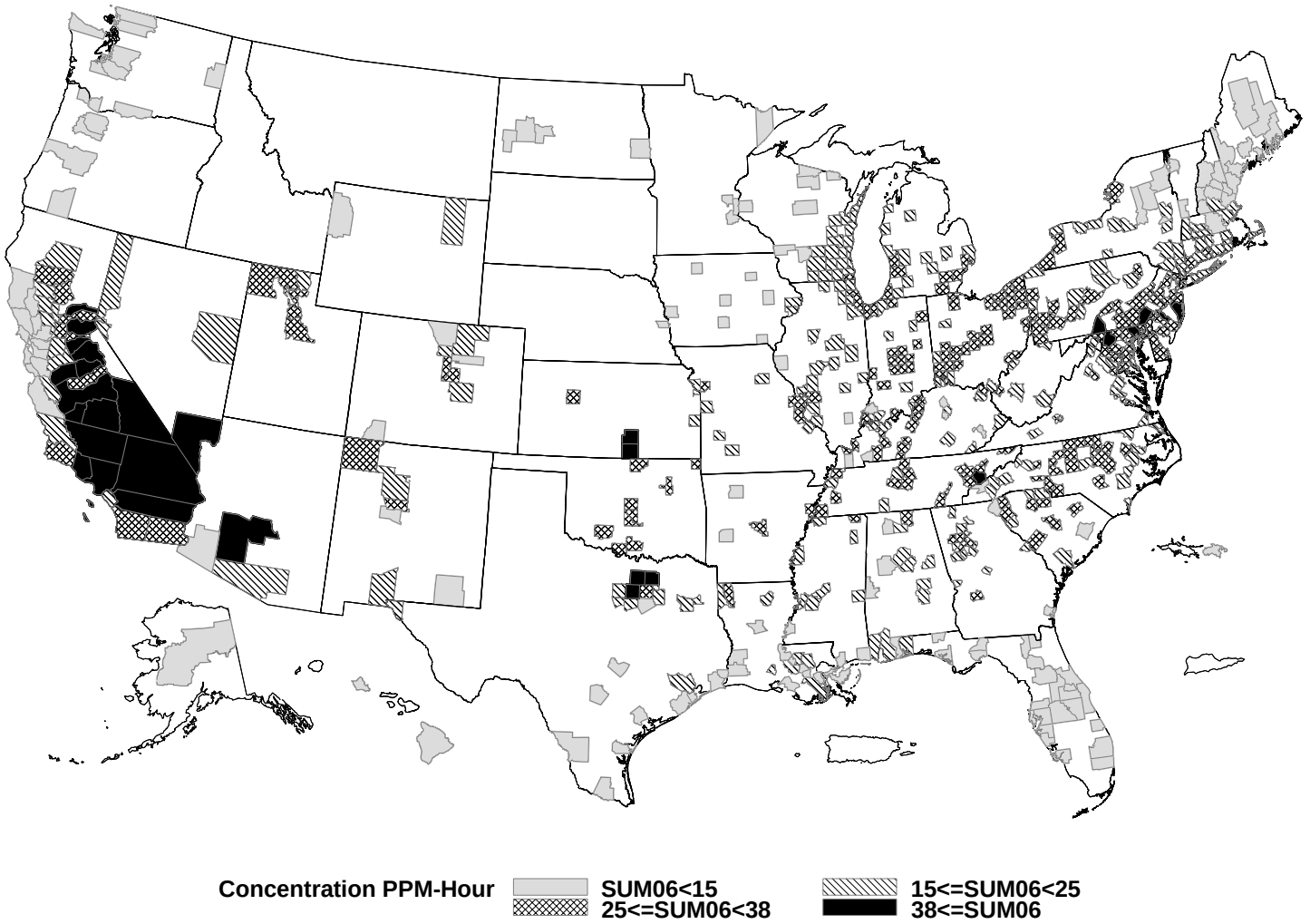

Figure 2-8. Highest 3-month 12-hour SUM06 Exposure Index in U.S. Counties, 2001 AQS Data.

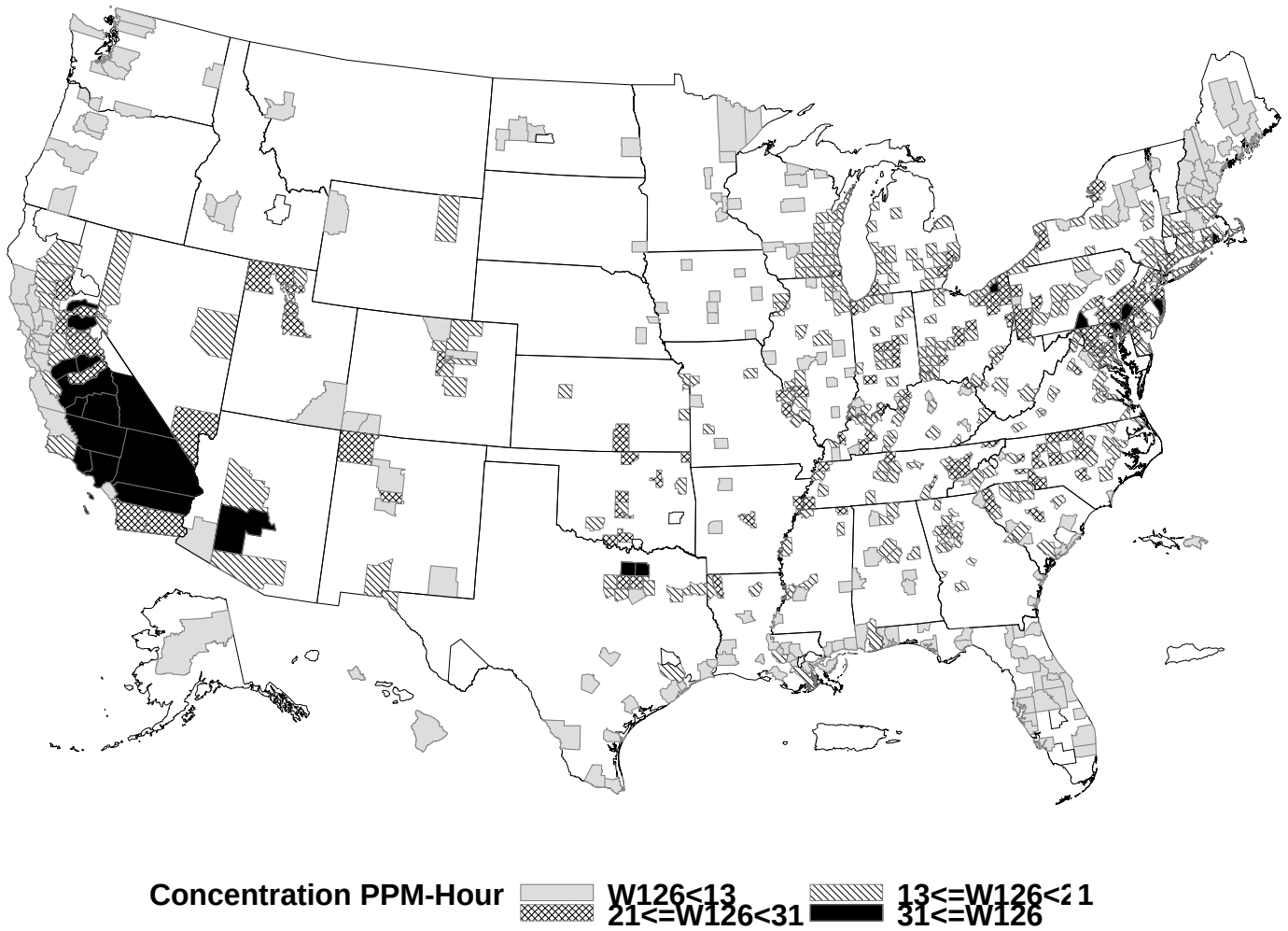

Figure 2-9. Highest 3-month 12-hour W126 Exposure Index in U.S. Counties, 2001 AQS Data.

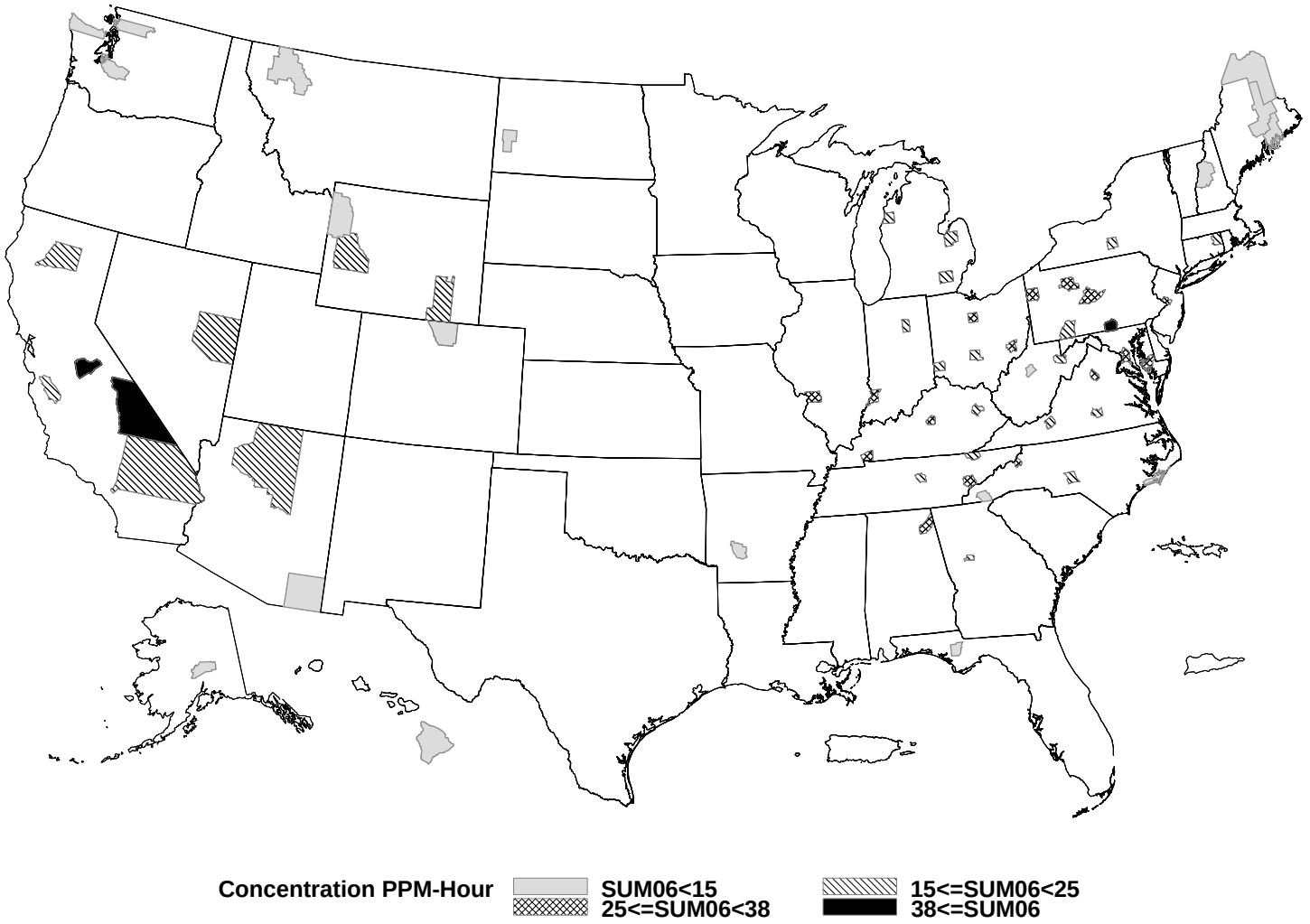

Figure 2-10. Highest 3-month 12-hour SUM06 Exposure Index in U.S. Counties, 2001 CASTNET Data.

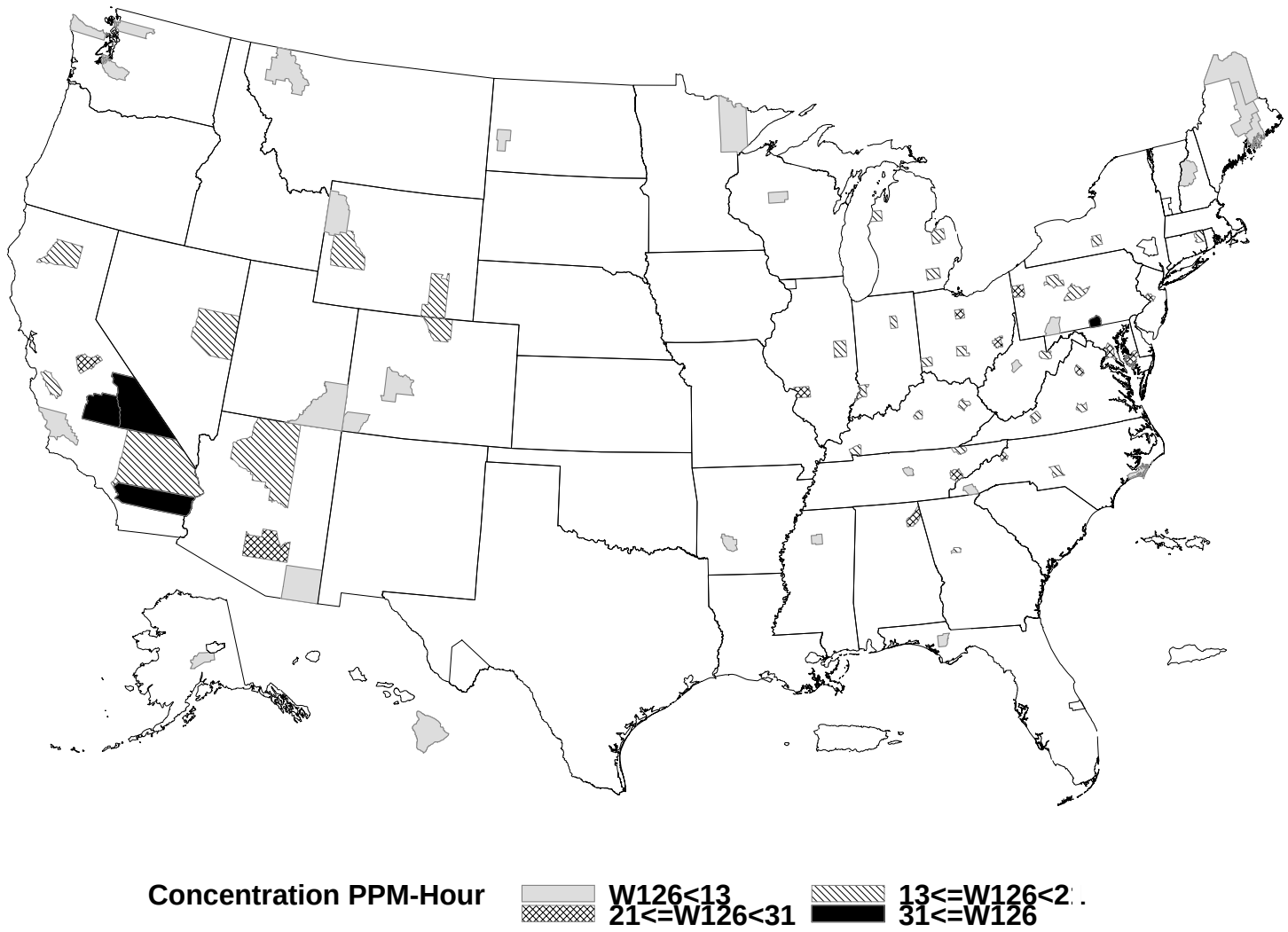

Figure 2-11. Highest 3-month 12-hour W126 Exposure Index in U.S. Counties, 2001 CASTNET Data.

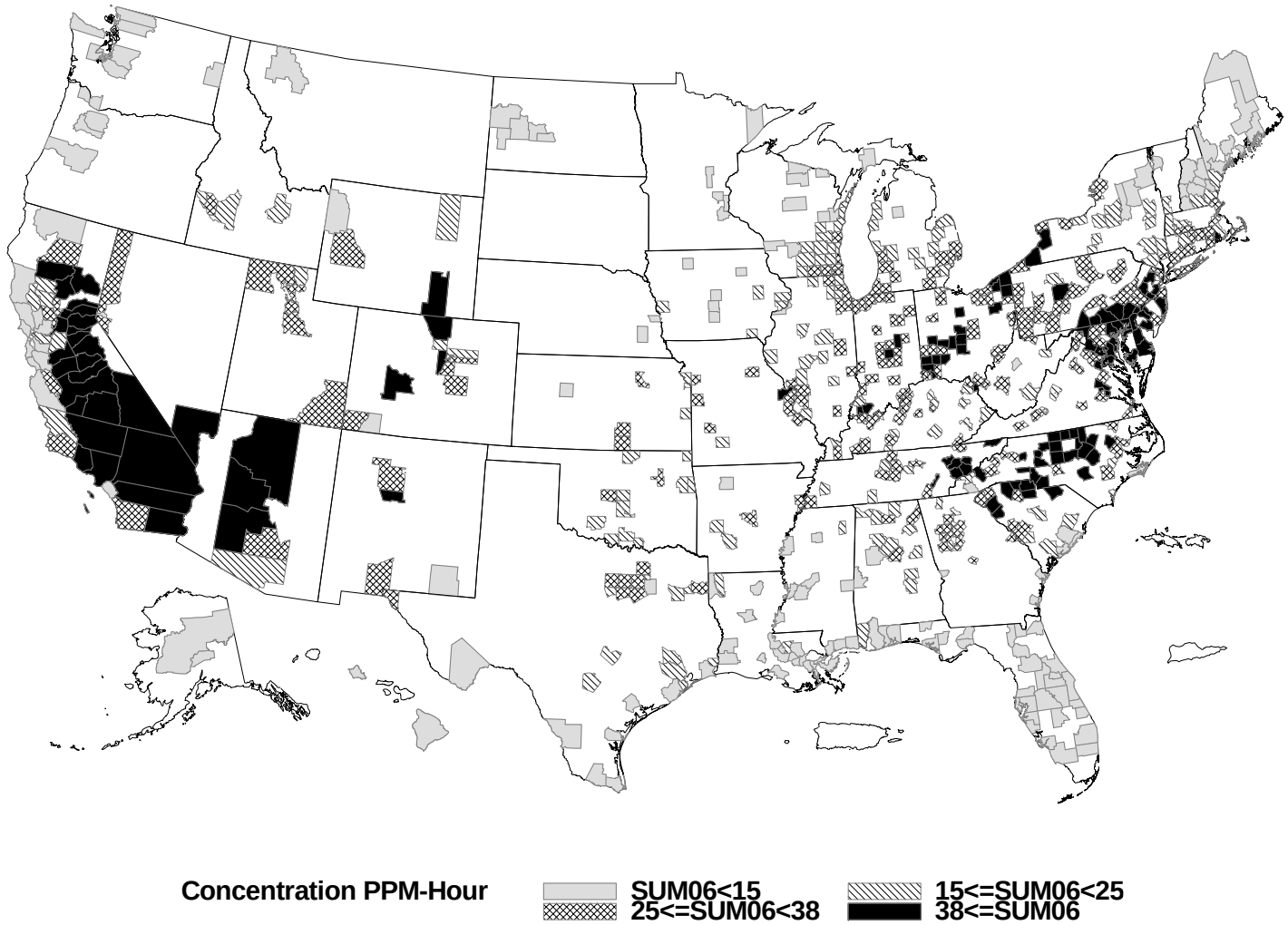

Figure 2-12. Highest 3-month 12-hour SUM06 Exposure Index in U.S. Counties, 2002 AQS and CASTNET Data.

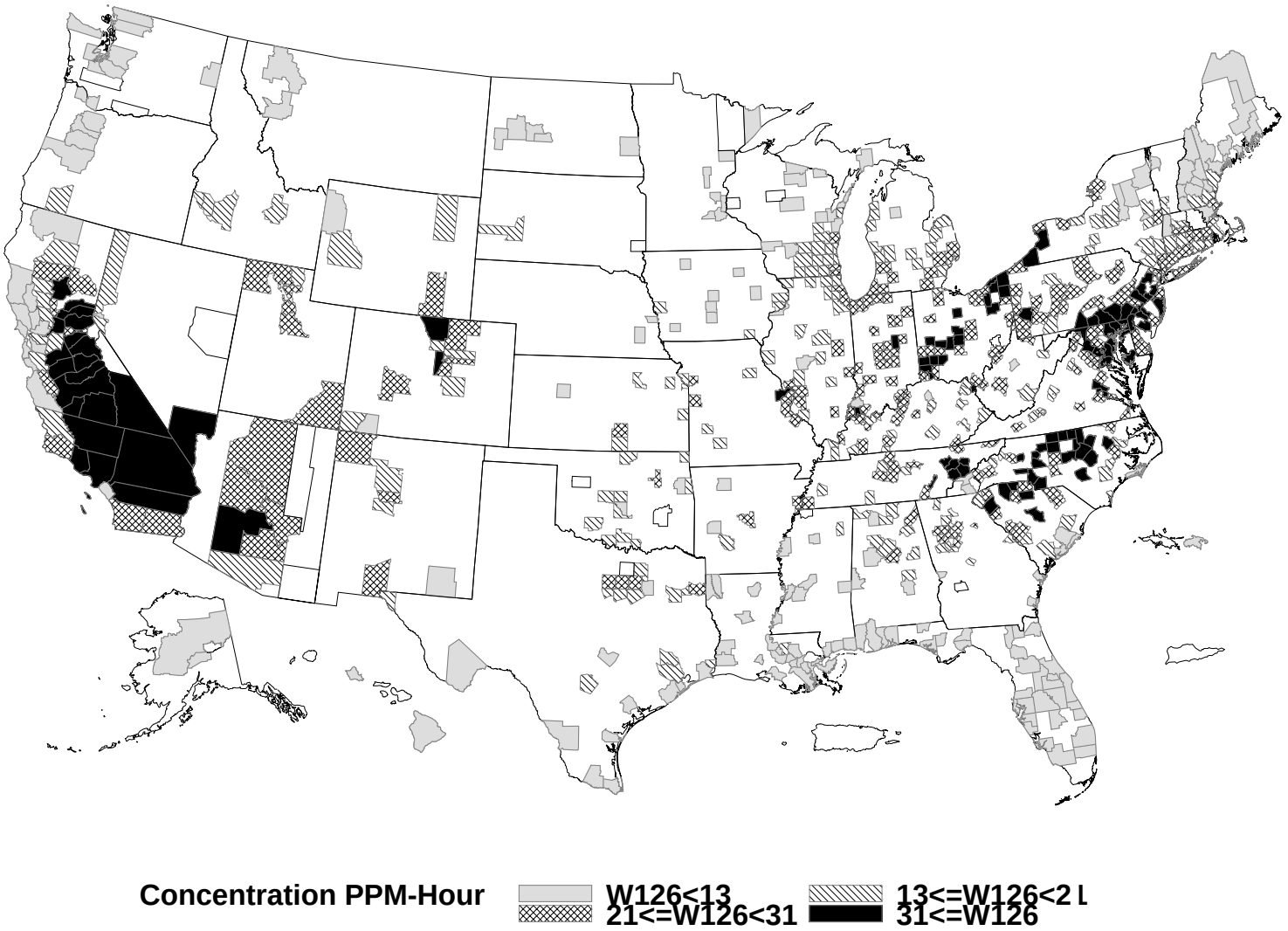

Figure 2-13. Highest 3-month 12-hour W1126 Exposure Index in U.S. Counties, 2002 AQS and CASTNET Data.

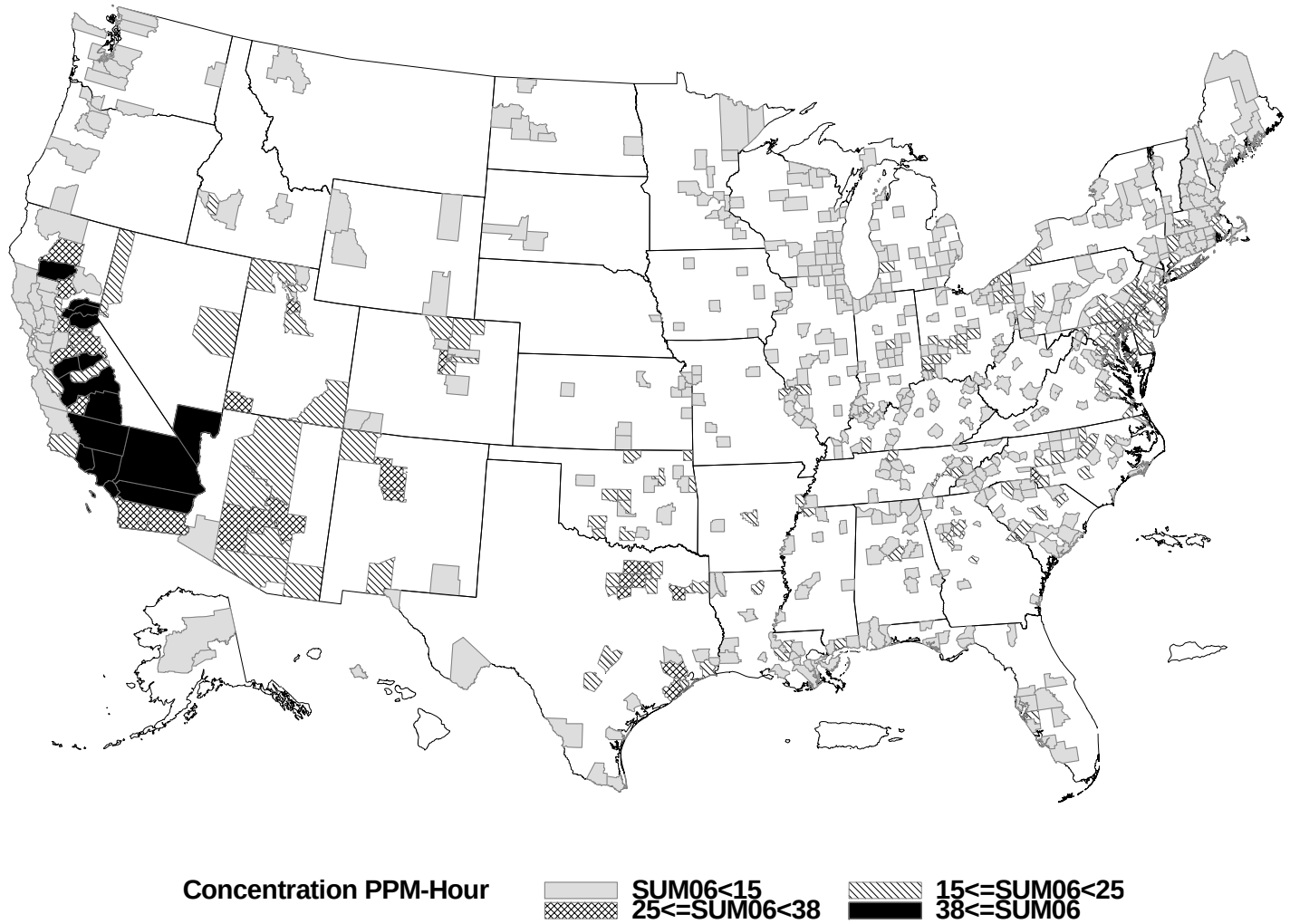

Figure 2-14. Highest 3-month 12-hour SUM06 Exposure Index in U.S. Counties, 2004 AQS and CASTNET Data.

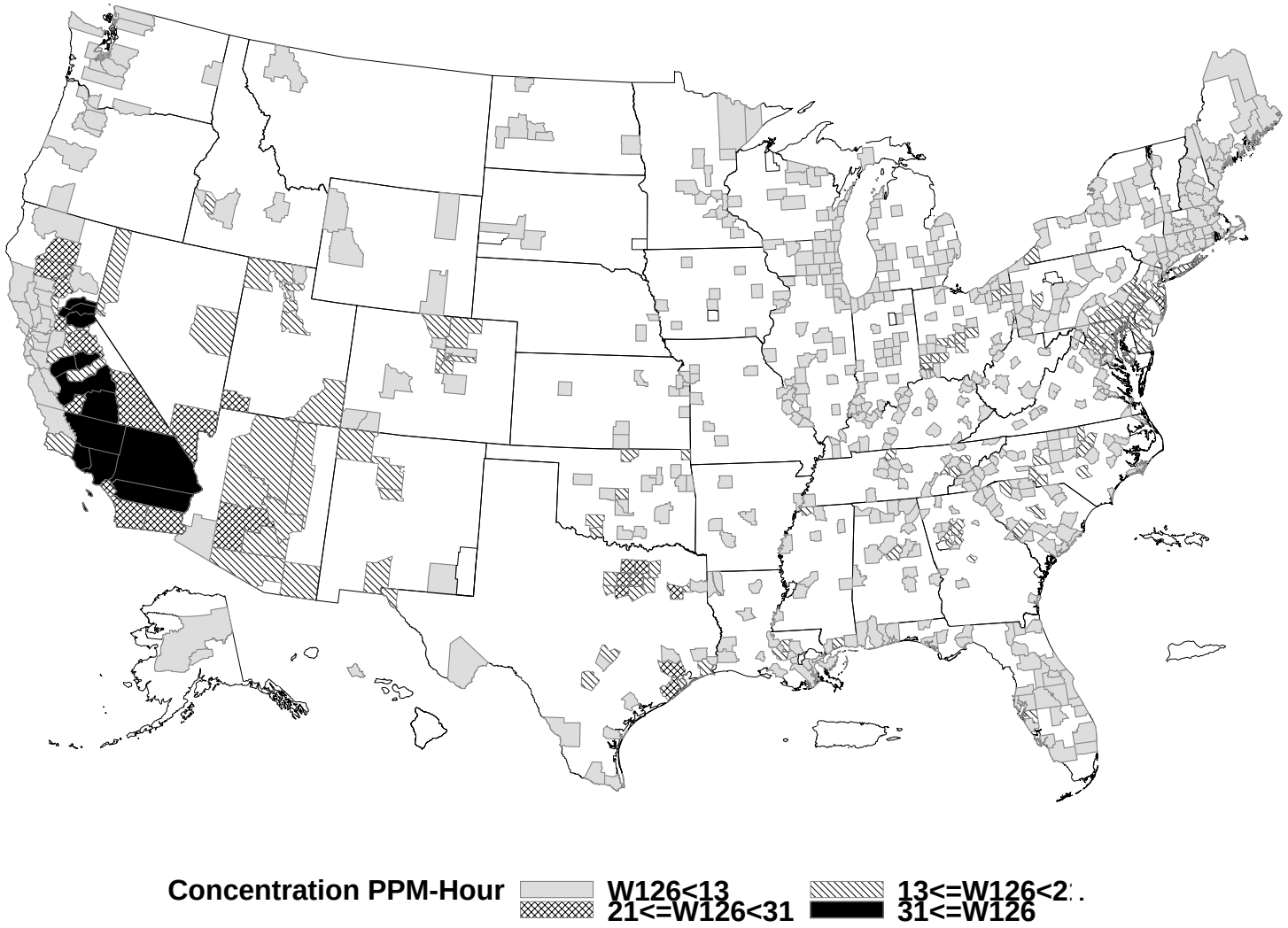

Figure 2-15. Highest 3-month 12-hour W126 Exposure Index in U.S. Counties, 2004 AQS and CASTNET Data.

2.5.3 Temporal Variability

Temporal variability consists of several time frames when considering characterization of ground level air quality data. Multi-year variability characterizes long term variability or year to year variability. Trends usually provide evidence on whether or not air quality is improving over time. For the purposes of displaying long term trends, the data from both AQS and CASTNET are screened for temporally consistent data (only data from sites that meet a data completeness criteria of 12 complete years out of 15 and no gaps of more than 3 consecutive years are included). Seasonal variability characterizes month to month variability to demonstrate when in the year the highest concentrations occur. Diurnal variability characterizes hour-to-hour changes demonstrating when, in the day, the highest concentrations occur (Fitz-Simons, et al., 2005).

2.5.3.1 Long Term Variability – Trends

Long term, nationwide trends for 8-hr O_3 values are presented in Figures 2-16 and 2-17. Figure 2-16 presents data from sites in the AQS that meet trends criteria and have locations described as Urban and Center City. Figure 2-17 presents data from CASTNET which are rural locations.

The rural and urban trends are similar, but the urban trends have more data and more variation. The rural means are slightly lower than the urban means; however the largest urban concentrations are much higher than the largest rural concentrations (Fitz-Simons, et al., 2005).

Long term trends for 1-hr O_3 values are presented in Figures 2-18 and 2-19. Figure 2-18 presents data from sites in the AQS that meet trends criteria and have locations described as Urban and Center City. Figure 2-19 presents data from CASTNET which are rural locations. As with the 8-hr data, the 1-hr urban trends and rural trends are similar, but urban have more data and more variation. The 1-hr means for the urban trends are higher than the means for the rural trends. This difference is more pronounced than in the 8-hr trends (Fitz-Simons, et al., 2005).

The long term trends for both 1-hr and 8-hr O_3 data are similar. The 8-hr concentrations are lower, but the trends are basically parallel. The urban area peak values in both the 1-hr and 8-hr concentrations have shown a gradual decline during the 15-year period, while the mean and median concentrations have not varied much. The highest means occur in 1990, 1991, 1995, 1998 and 2002. The highest extreme values are clearly in the 1990s. In many cases, short term variation (3 years or less) is associated with meteorological conditions that are generally more or less conducive to O_3 formation in a particular year. One high year between two low years or one low year between two higher years are examples of this 3 years or less variation (see Evaluating Ozone Control Programs in the Eastern United States: NO_X p.17, U.S. EPA, 2005b)

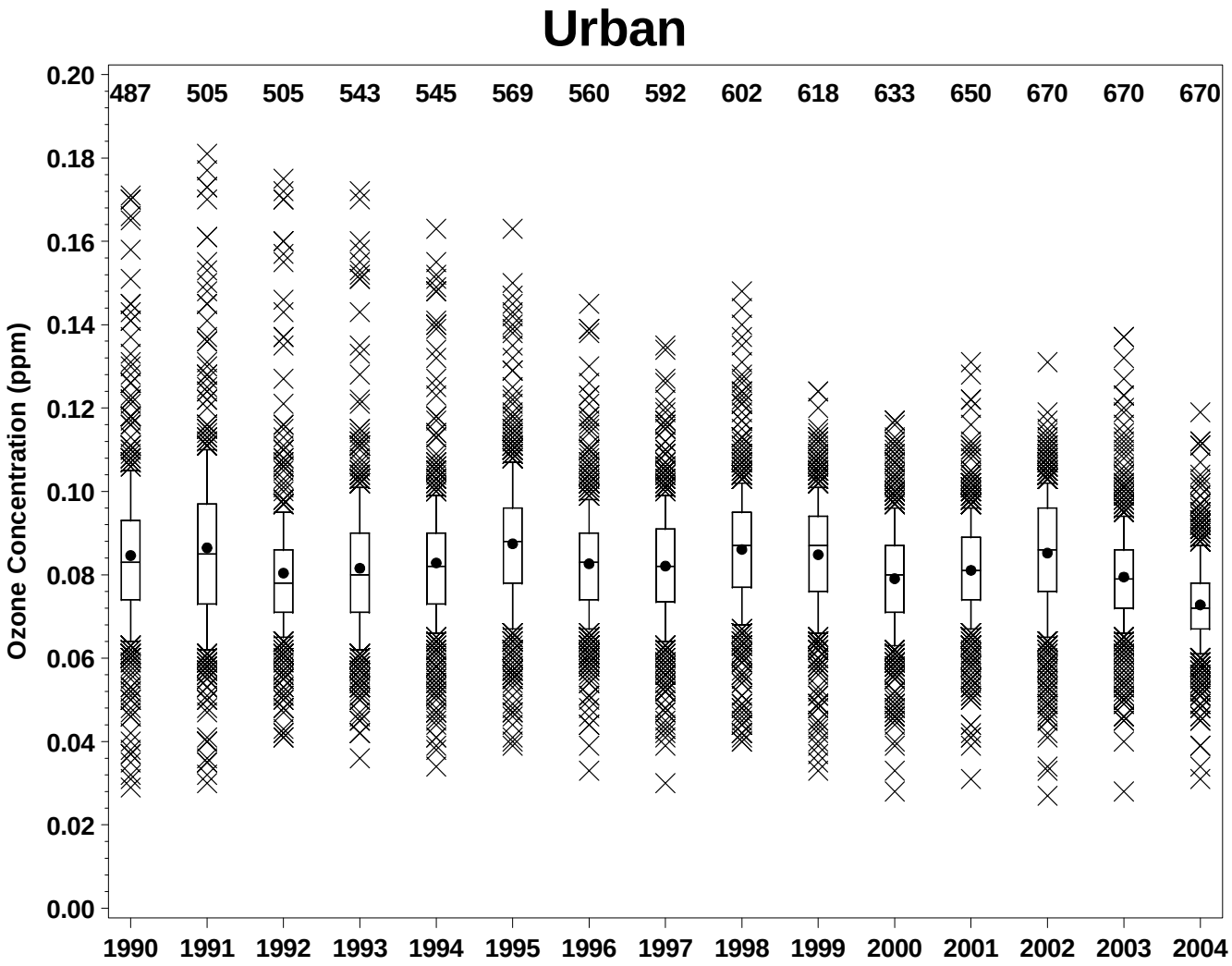

Figure 2-16. 4th Highest Daily Maximum 8-hour Ozone Values 1990-2004 (Urban). Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; 'x' depicts values outside the 10th and 90th percentile; and numbers above the boxes depicts the number of sites.

Data Source: AQS

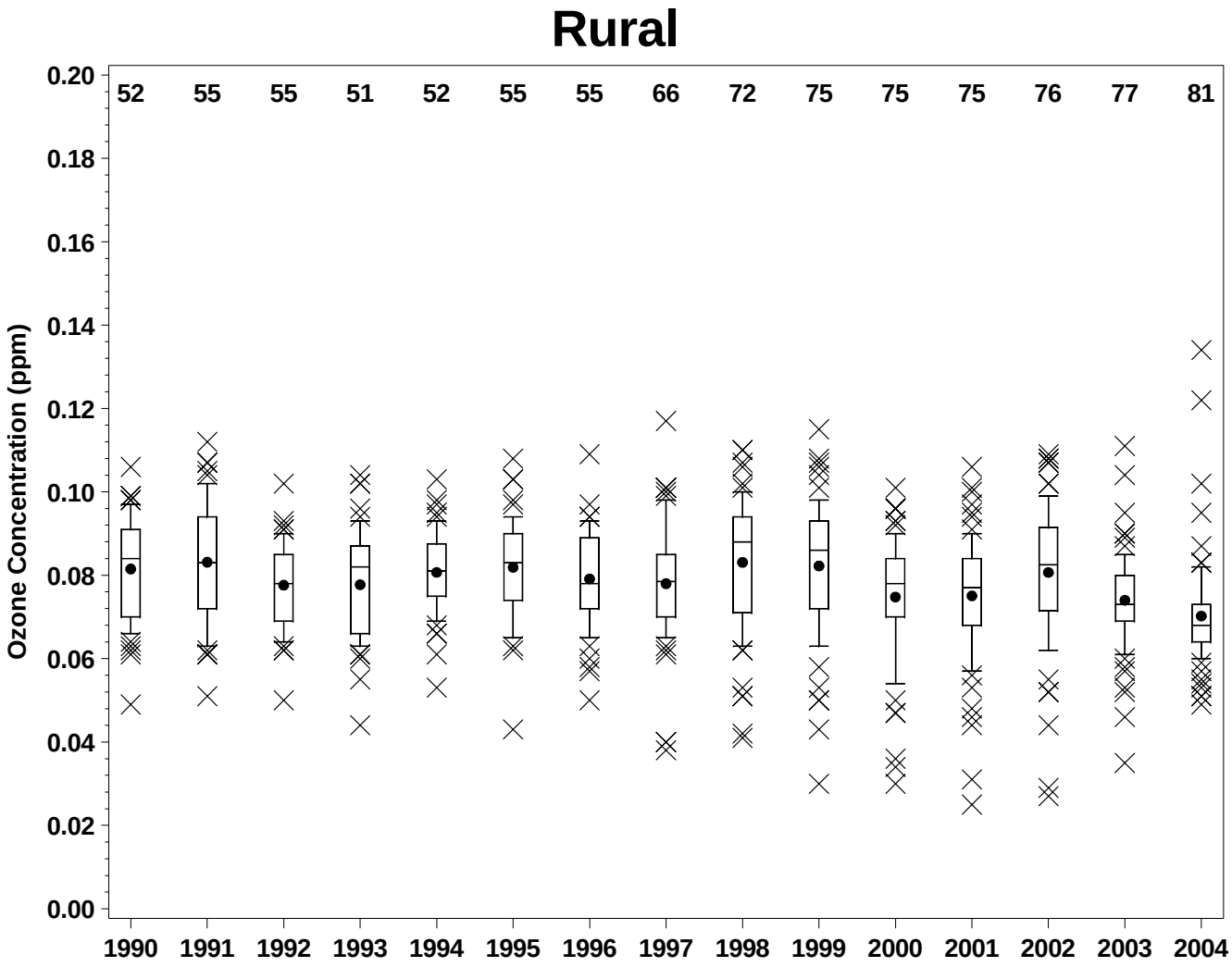

Figure 2-17. 4th Highest Daily Maximum 8-hour Ozone Values 1990-2004 (Rural). Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; 'x' depicts values outside the 10th and 90th percentile; and numbers above the boxes depicts the number of sites.

Data Source: CASTNET

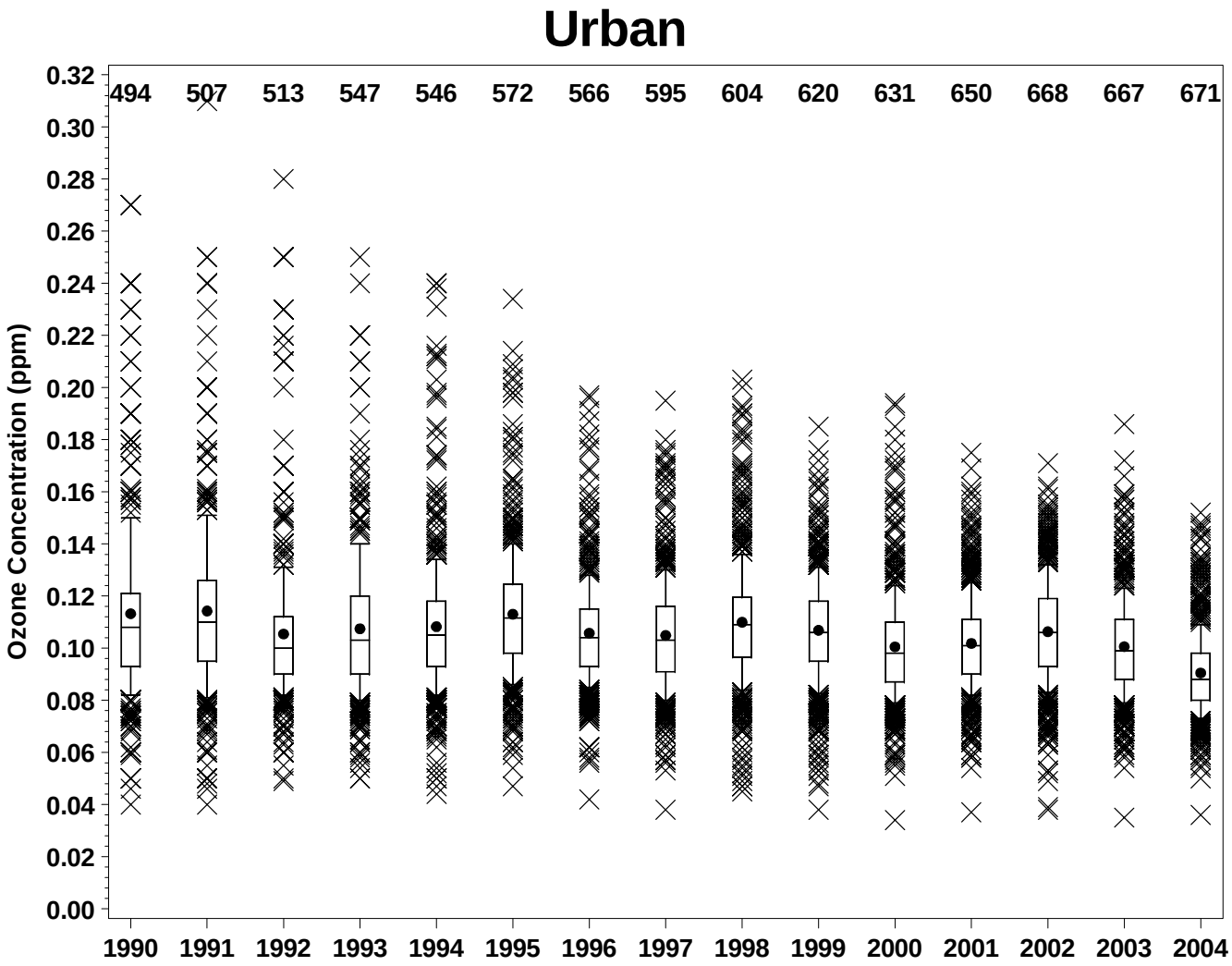

Figure 2-18. 2nd Highest Daily Maximum 1-hour Ozone Values 1990-2004 (Urban). Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; 'x' depicts values outside the 10th and 90th percentile; and numbers above the boxes depicts the number of sites.

Data Source: AQS

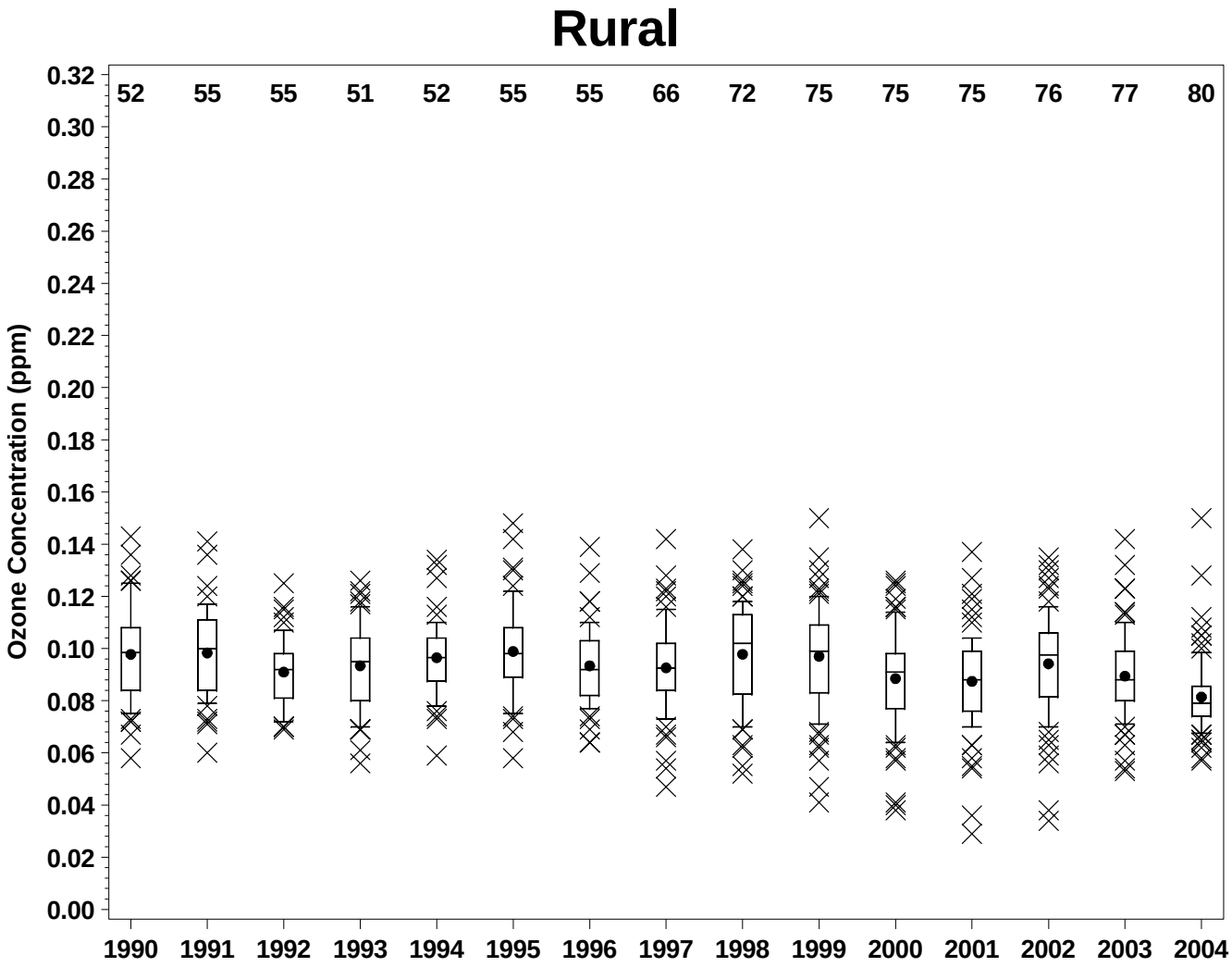

Figure 2-19. 2nd Highest Daily Maximum 1-hour Ozone Values 1990-2004 (Rural). Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; 'x' depicts values outside the 10th and 90th percentile; and numbers above the boxes depicts the number of sites.

Data Source: CASTNET

2.5.3.2 Short Term Variability – Annual

Figure 2-20 shows a map of the number of exceedance days for 2002 and 2004 at 12 urban locations in the United States. Each grouping of two bars represents the number of exceedance days for 1-hr, 8-hr averaged and 24-hr averaged O_3 concentrations. The 1-hr measured concentrations were compared to the previous 1-hr O_3 standard of 0.12 ppm, 8 hour averaged concentrations were compared to the current O_3 standard of 0.08 ppm and the average 24 hour concentrations were compared to 0.055 ppm which is the 95th percentile for 24 hour O_3 concentrations across the United States for 2002 through 2004. The data show that in all sites in the Midwest and the East, O_3 concentrations were down dramatically in 2004 when compared to 2002. This is due in part to the fact that 2004 was much cooler than 2002. The reduction in peak O_3 concentrations also reflects the improvement in air quality due to emission reductions from the NO_x SIP Call, mobile source and other stationary source rules. The NOx SIP Call provided large NO_x reductions in the eastern part of the country in 2003 and 2004, thereby reducing peak O_3 concentrations (U.S. EPA, 2005b). The number of 8-hr exceedance days actually increased for Houston while showing a decrease in Los Angeles. The difference between 2002 and 2004 for days greater than 0.055 ppm for the 24 hour averaged concentrations is smaller for the three cities west of the Mississippi River than it is for cities in the eastern United States.

2.5.3.3 Seasonal Variability

Monthly statistics are the best method to characterize seasonal variation in O_3 concentrations. However in many areas, monitors are not active during cooler months. As a result, data from May through September are the only universally available data for all monitors. Although this is a limited characterization of seasonal variability, it is consistent across the entire national network.

Figure 2-21 shows box-plots of all 2004 data from May through September for the second highest daily 1-hr maximums. The center of the distribution shows a slight, steady increase from May to September while the extreme values show a more pronounced but more variable increase for the same period (Fitz-Simons, et al., 2005).

Figure 2-22 shows box-plots of all 2004 data from May through September for the fourth highest daily 8-hr maximums. The center of the distribution and the extremes show a slight, steady increase from May to July followed by a slight decrease from July through September (Fitz-Simons, et al., 2005).

2.5.3.4 Short Term Variability – Diurnal

The daily cycles of human activity and the solar phase drive the hour-to-hour daily cycle seen in ground level O_3 concentrations. The daily 1-hr peak levels generally occur in the afternoon with the lowest concentration occurring in the early morning. However, on any given

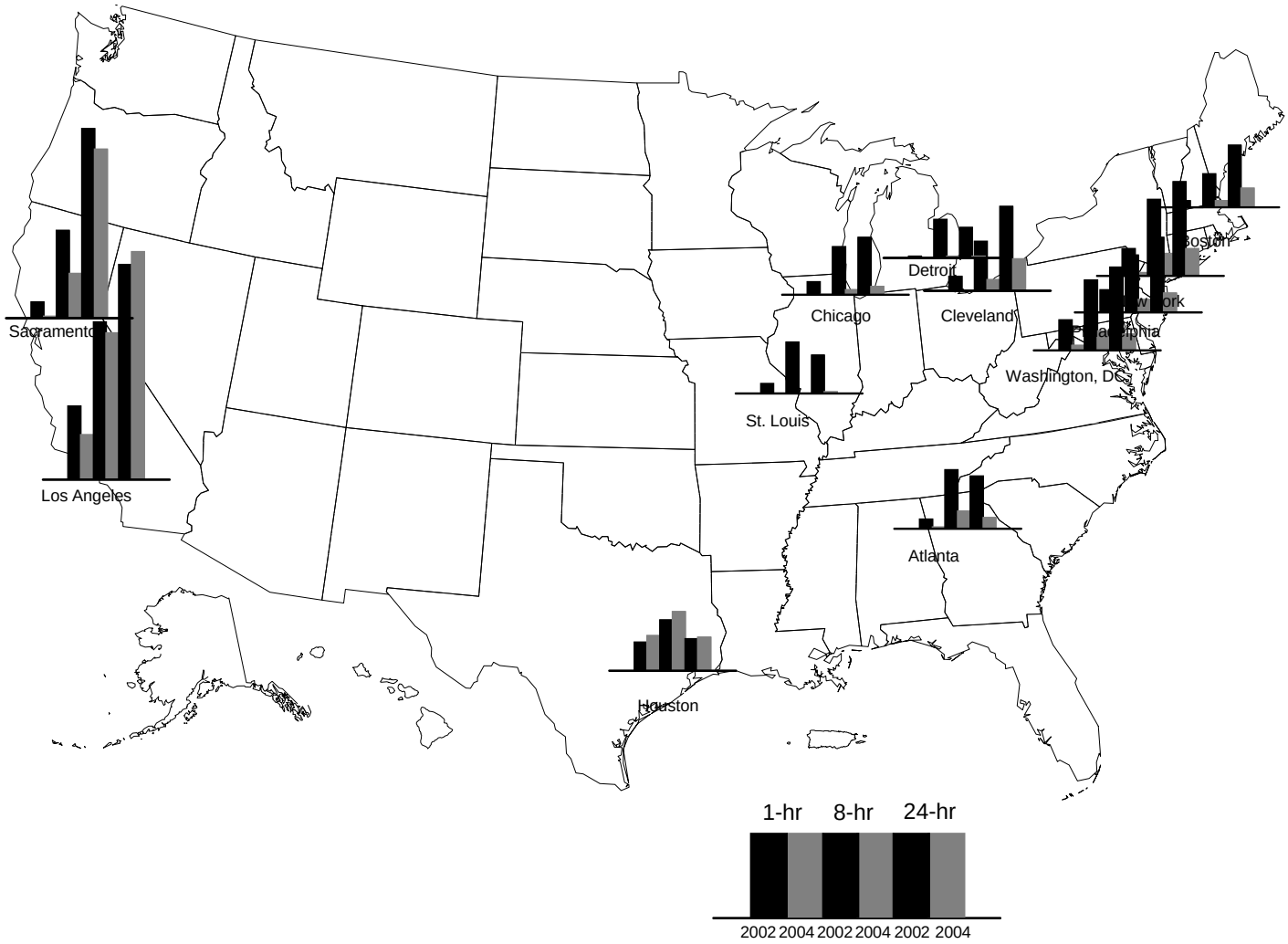

Figure 2-20. Comparison of 1-hr, 8-hr, and 24-hr Metrics for 2002 and 2004, 12 Risk Areas

Data Source: AQS

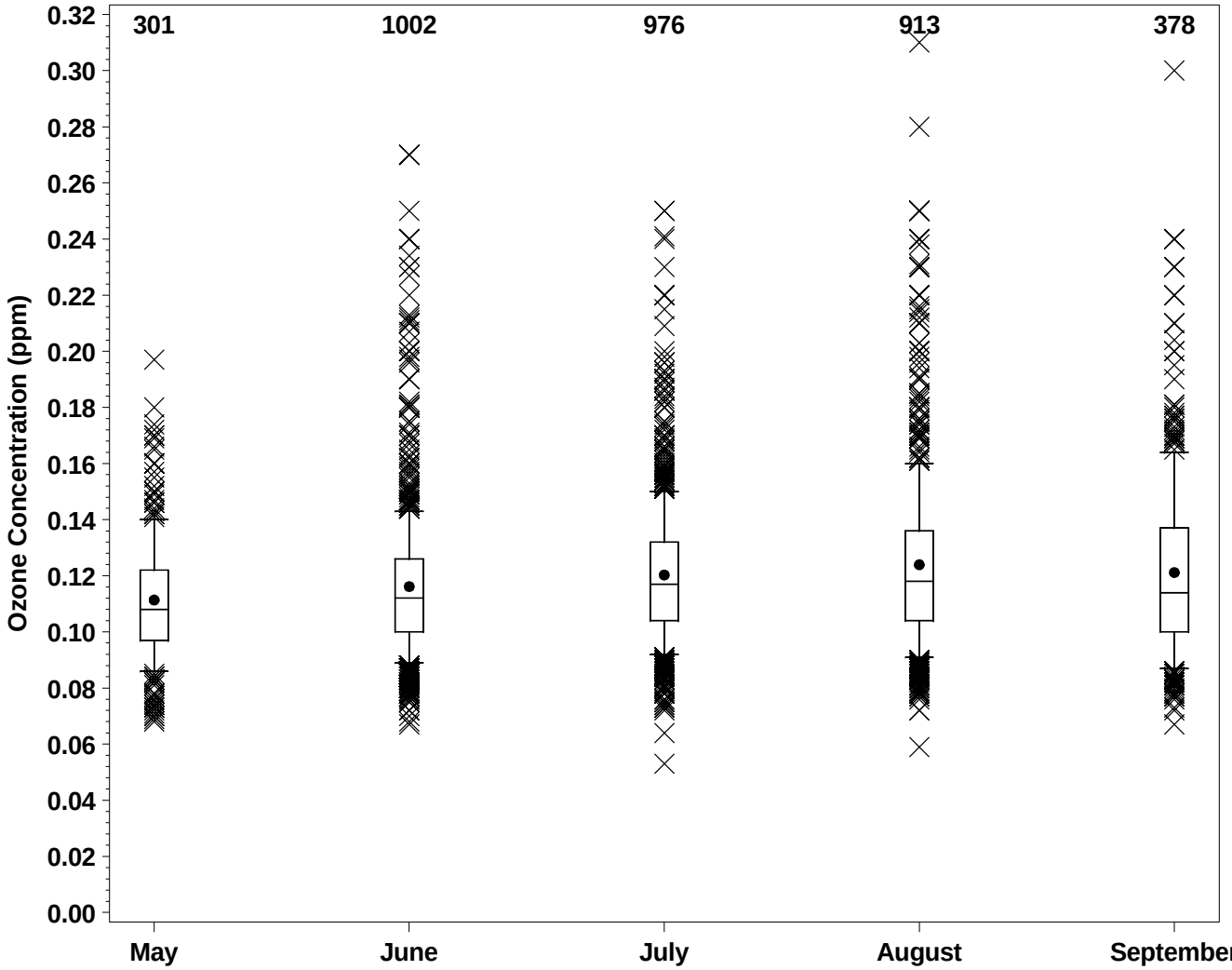

Figure 2-21. 2nd Highest Daily Maximum 1-hour Ozone Values from 2004 by Month. Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; 'x' depicts values outside the 10th and 90th percentile; and numbers above the boxes depicts the number of sites.

Data Source: AQS

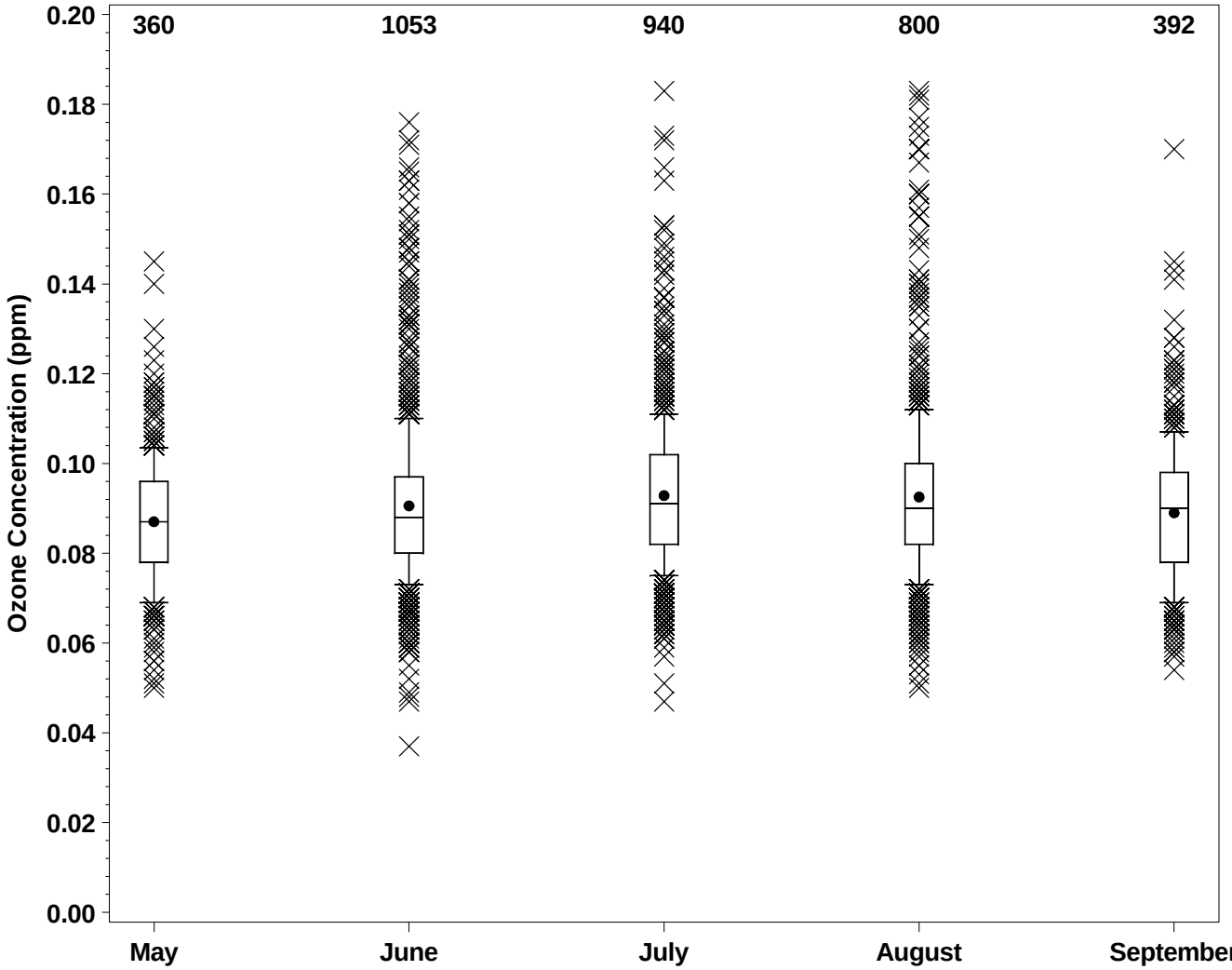

Figure 2-22. 4th Highest Daily Maximum 8-hour Ozone Values from 2004 by Month. Box Depicts interquartile range and median; the dot depicts the mean; whiskers depict 10th and 90th percentile; 'x' depicts values outside the 10th and 90th percentile; and numbers above the boxes depicts the number of sites.

Data Source: AQS

day when conditions are right, this phase can be reversed with the highest values occurring at night or early morning. Ozone transport can also effect at what time peaks can occur. For example, some sites in Maine peak late in the evening due to transport.

In order to examine diurnal patterns, box-plots summarize 1-hr values and 8-hr for each hour in the day. Figures 2-23 and 2-24 summarize 1-hr data from AQS that was classified as urban and center city. The pattern is similar for both weekend and week day data. The pattern of the center of the distribution of values shows a smooth sinusoidal portion of the curve from 6:00AM until 8:00PM and reaches a peak at 1:00 PM to 3:00 PM. Then the pattern alters to a gradual decrease from 9:00 PM to 6:00AM (Fitz-Simons, et al., 2005).

Figures 2-25 and 2-26 show the same set of summaries for 8-hr data. 8-hr values run from 0 to 23 hours. Hour1 is the average of 1-hr values from 1 to 8 while hour 2 is the average of hours 2 to 9 and so on. The main difference between the 1-hr data and the 8-hr data is that the 8-hr data exhibit a smoother sinusoidal pattern throughout the day with a peak for the center of the distribution occurring at 10:00 AM or 11:00 AM and a minimum at about 12:00 midnight. The week end pattern is similar to the week day pattern (Fitz-Simons, et al., 2005).

Figures 2-27 through 2-30 summarize 1-hr and 8-hr data from CASTNET sites which are considered rural. Several differences are noted here. The patterns for the center of the distribution are similar to the patterns for the urban sites. The largest values of the 1-hr data exhibit no pattern but the largest values for the 8-hr data have a discernable pattern that differs from the patterns for the values in the center of the distribution. The weekday pattern for the highest values, shown in figure 2-29, has a smooth sinusoidal pattern but reaches 2 peaks in the day (12:00 midnight and 12:00 noon). The weekend pattern, shown in figure 2-30, also shows a pronounced peak in the afternoon at about 1:00 PM which occurs about 2 hours after the peak for the values in the center of the distribution (Fitz-Simons, et al., 2005).

2.6 CHARACTERIZATION OF OZONE EPISODES

Major episodes of high O_3 concentrations in the United States are associated with slow moving, high pressure systems. High pressure systems during the warmer seasons are associated with the sinking of air, resulting in warm, generally cloudless skies, with light winds. These conditions result in the development of stable air masses near the surface which inhibit the vertical mixing of O_3 precursors. The combination of inhibited limited vertical mixing and light winds minimizes the dispersal of pollutants emitted in urban areas, allowing their concentrations to build up. Photochemical activity involving these precursors is also enhanced because of higher temperatures and the availability of sunlight. Downward entrainment of overnight transported O_3 and precursors trapped aloft begins on the following day as the PBL starts growing. In the

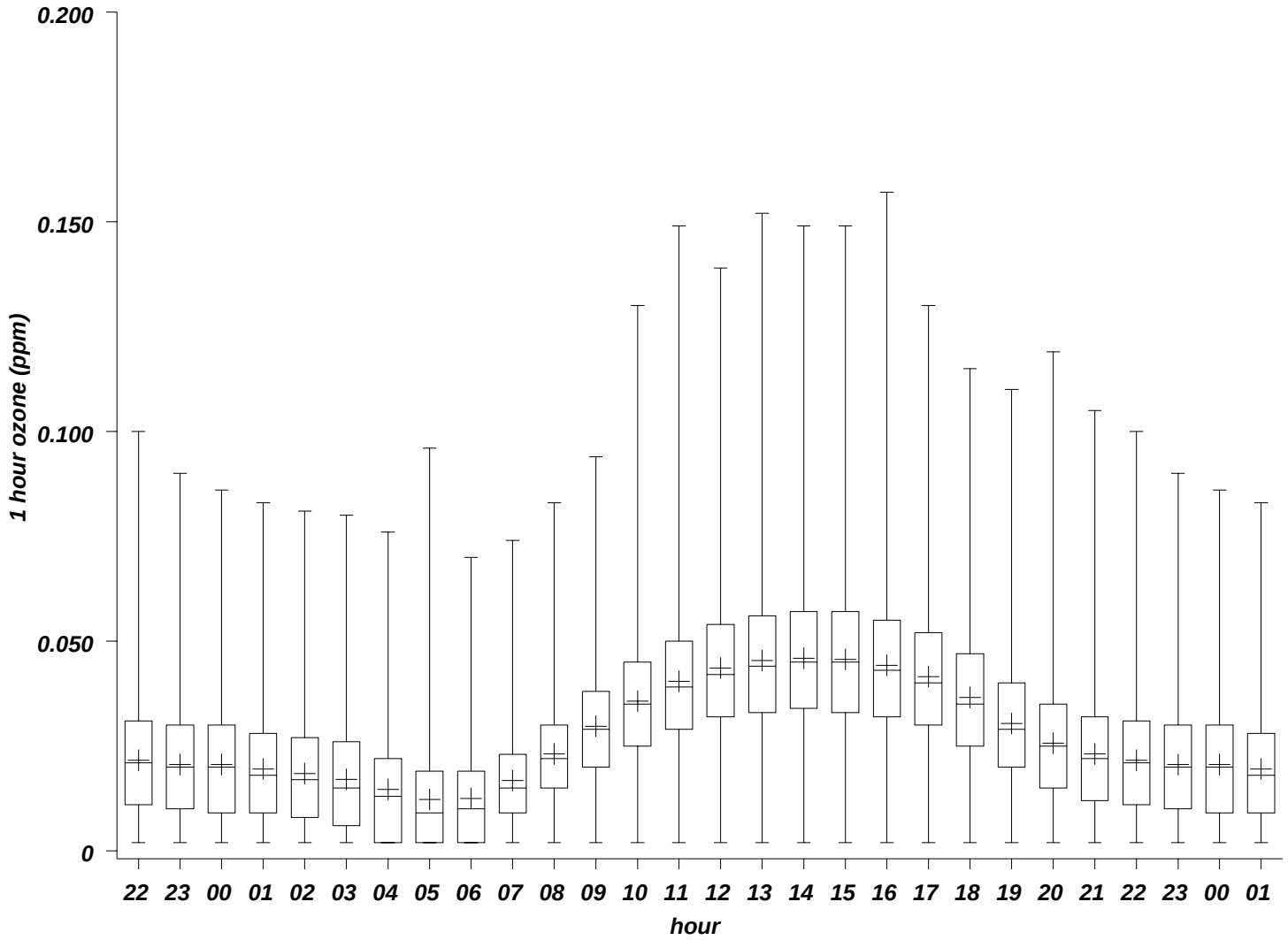

Figure 2-23. 1-Hour Diurnal Week Day Pattern for Urban Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.
Data Source: AQS

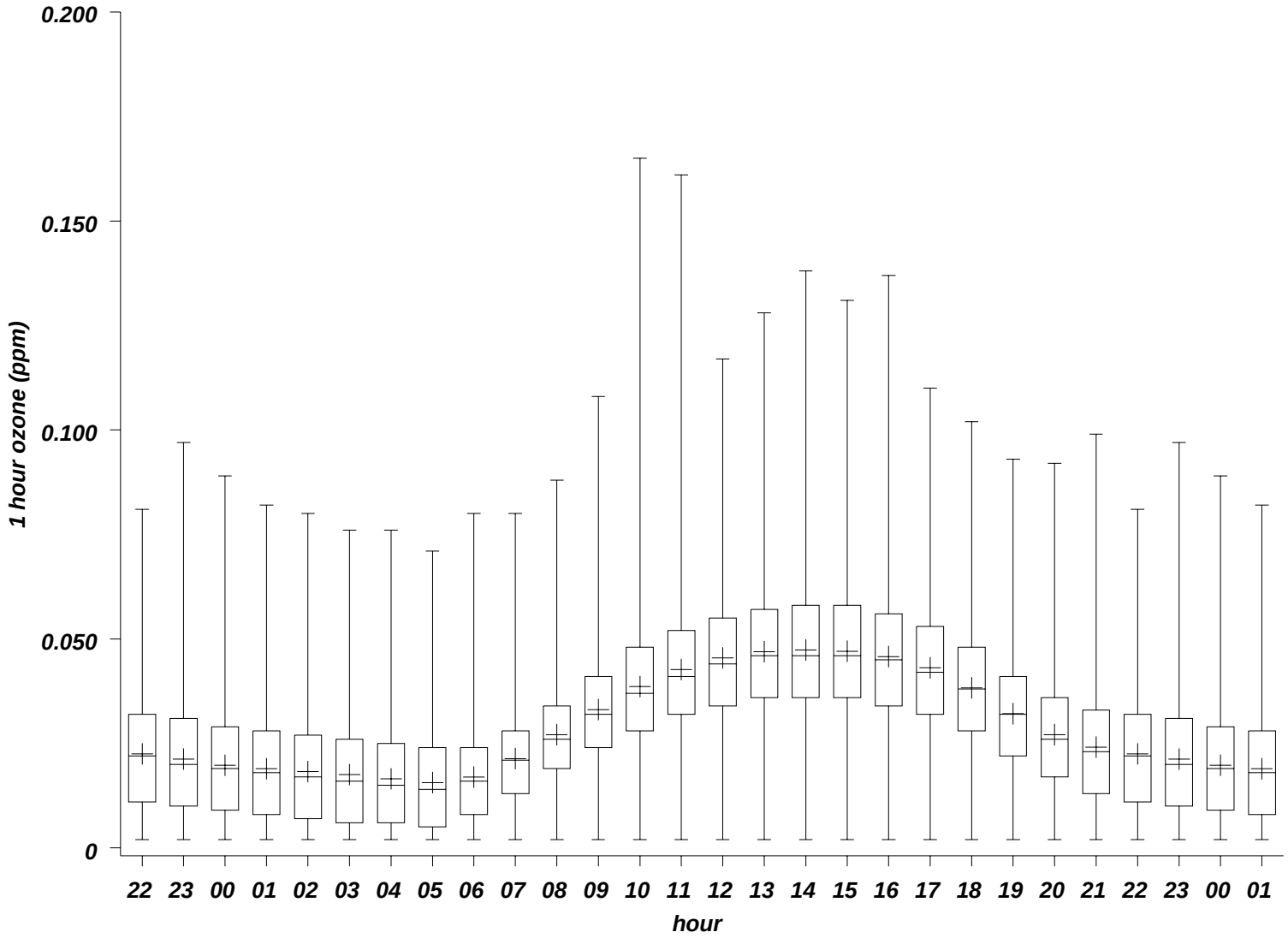

Figure 2-24. 1-Hour Diurnal Week End Pattern for Urban Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.

Data Source: AQS

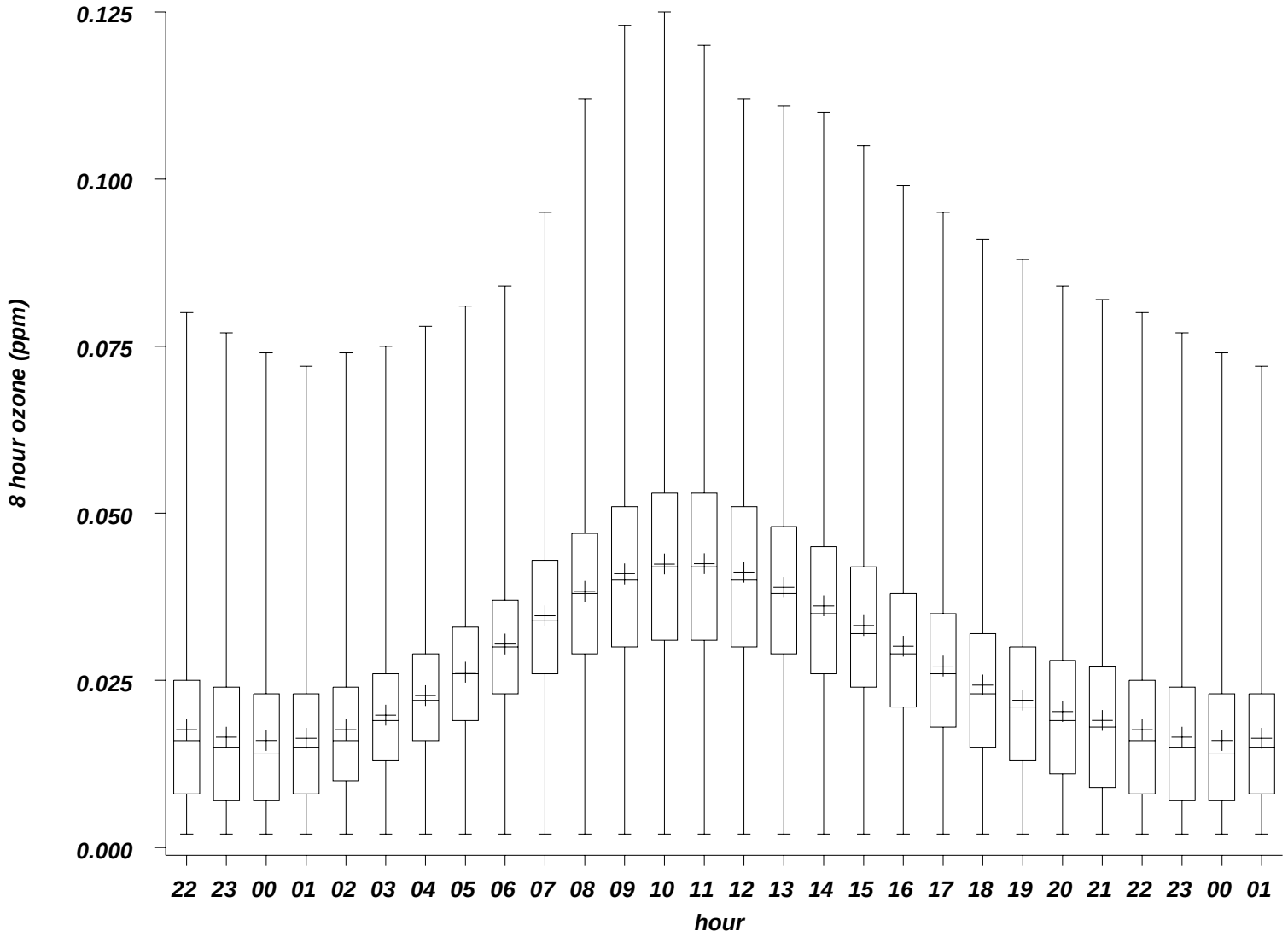

Figure 2-25. 8-Hour Diurnal Week Day Pattern for Urban Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.
Data Source: AQS

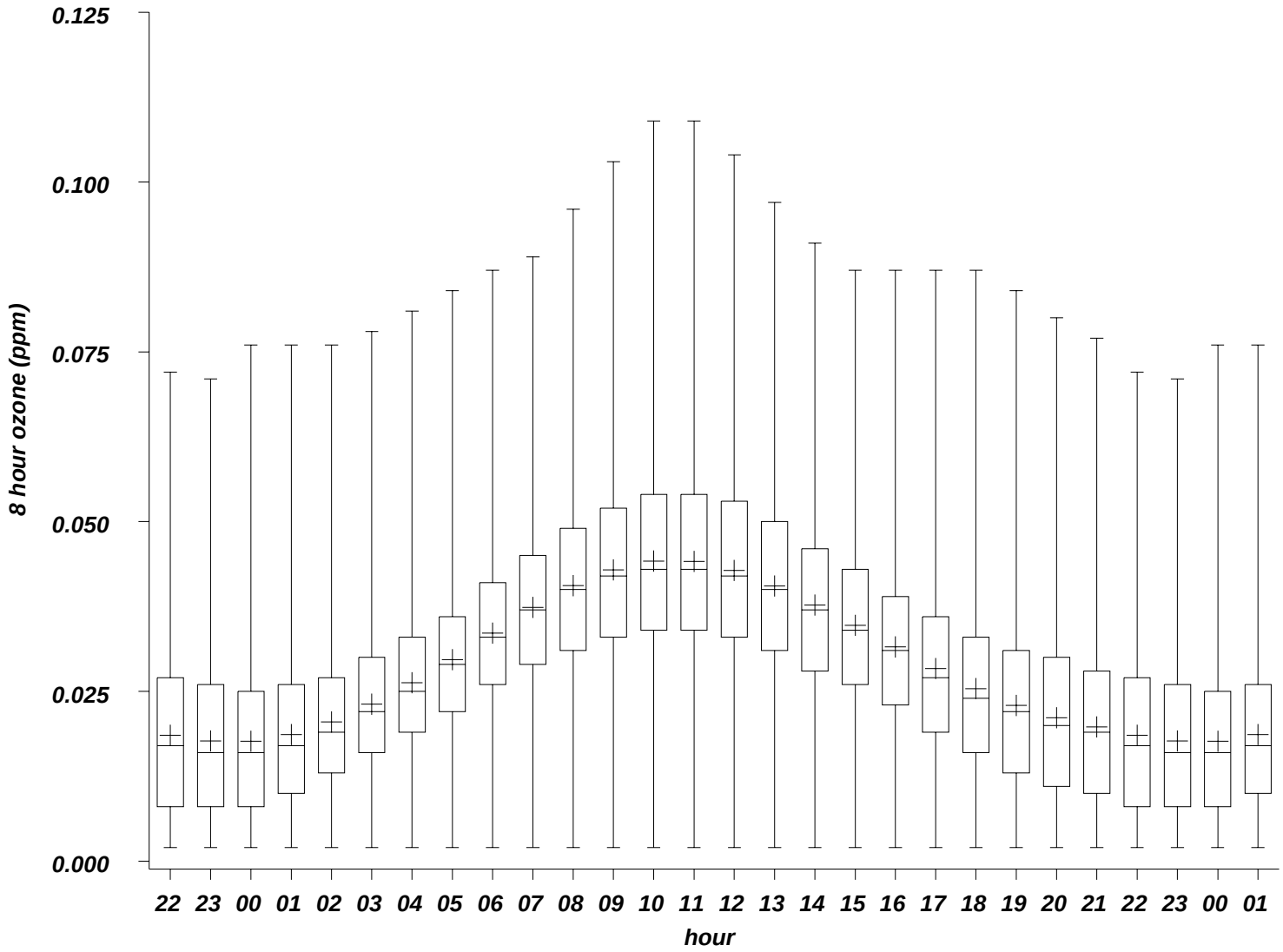

Figure 2-26. 8-Hour Diurnal Week End Pattern for Urban Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.

Data Source: AQS

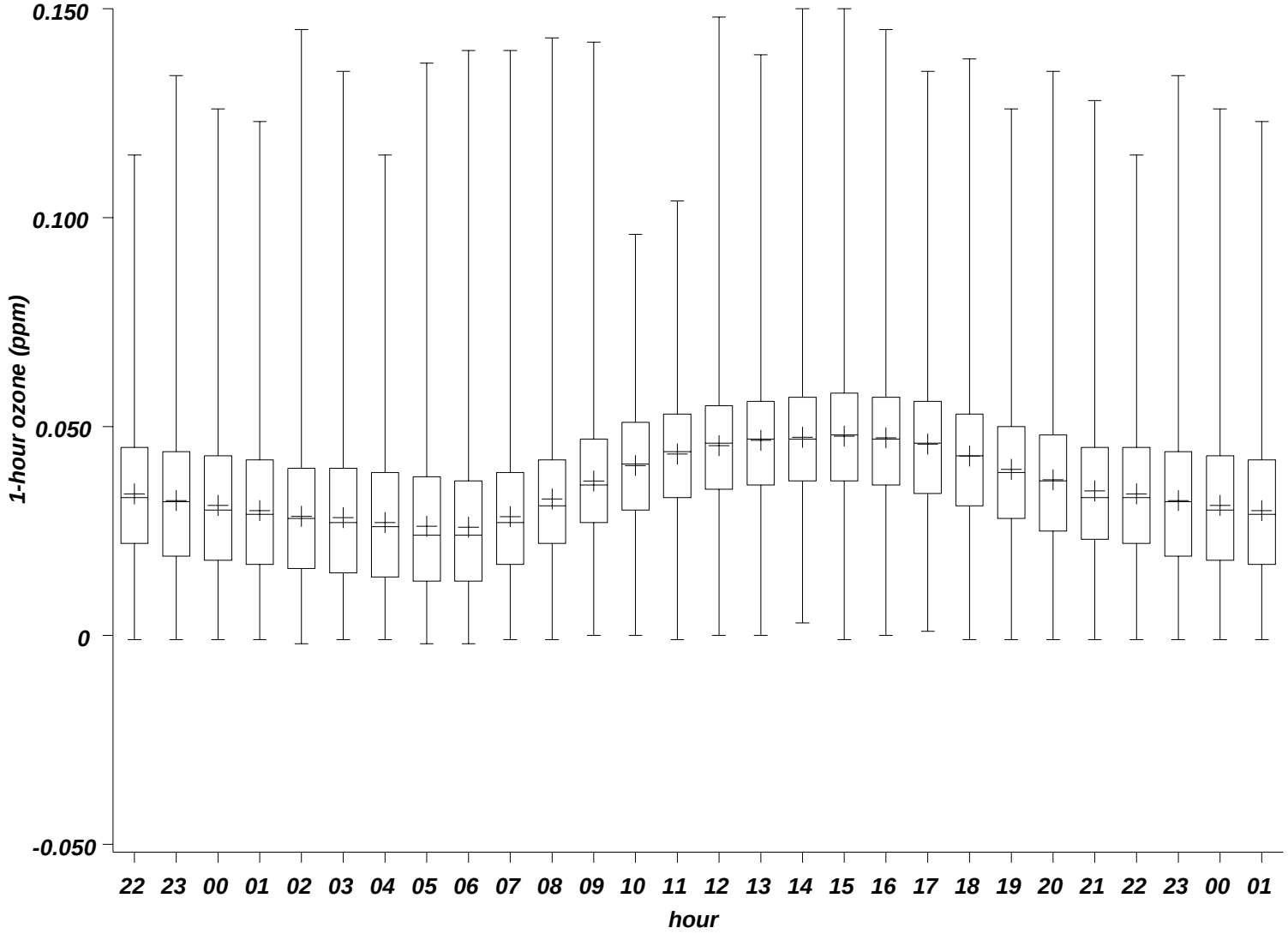

Figure 2-27 1-Hour Week Day Diurnal Pattern for Rural Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.

Data Source: CASTNET

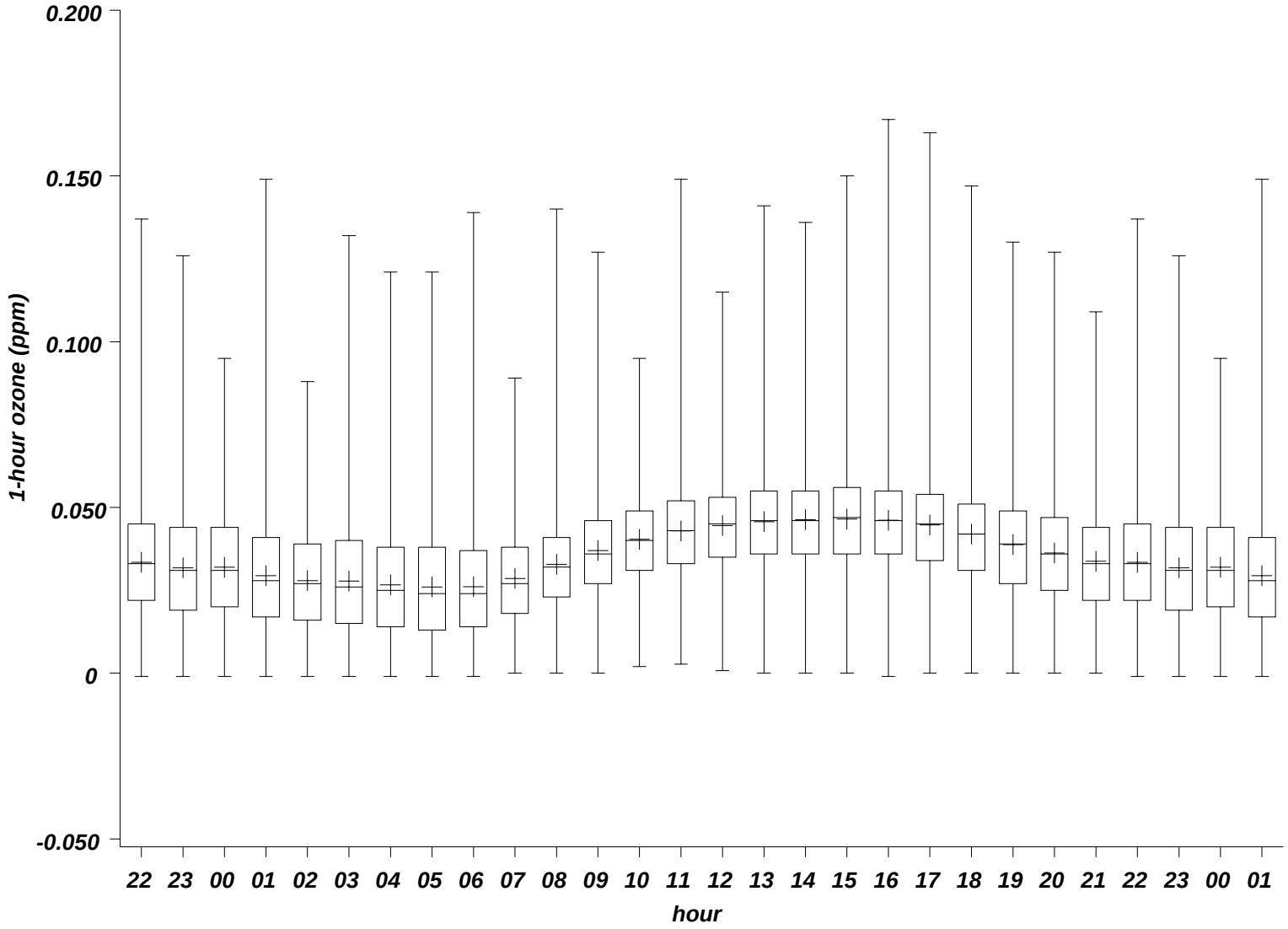

Figure 2-28. 1-Hour Week End Diurnal Pattern for Rural Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.

Data Source: CASTNET

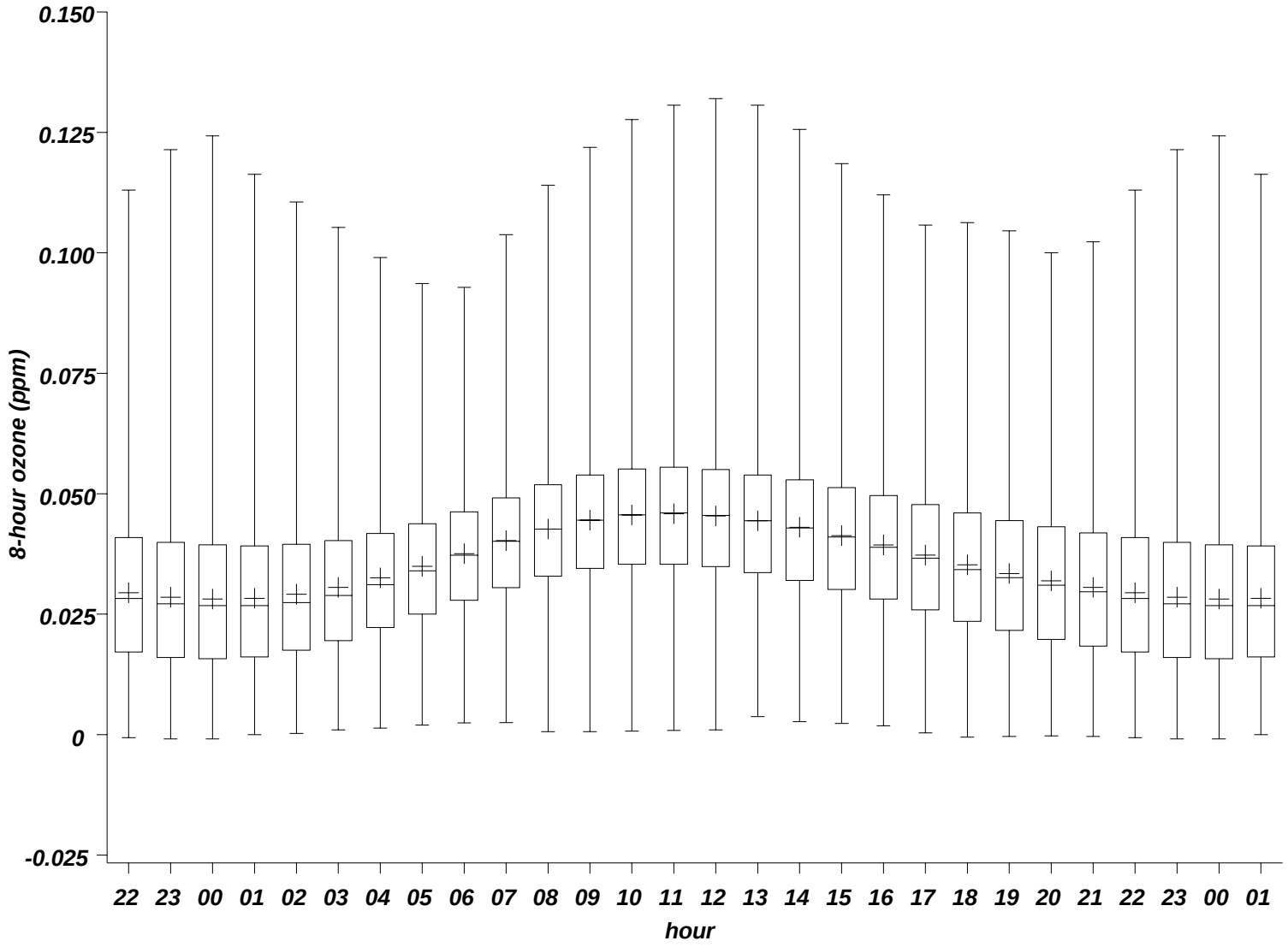

Figure 2-29. 8-Hour Week Day Diurnal Pattern for Rural Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.

Data Source: CASTNET

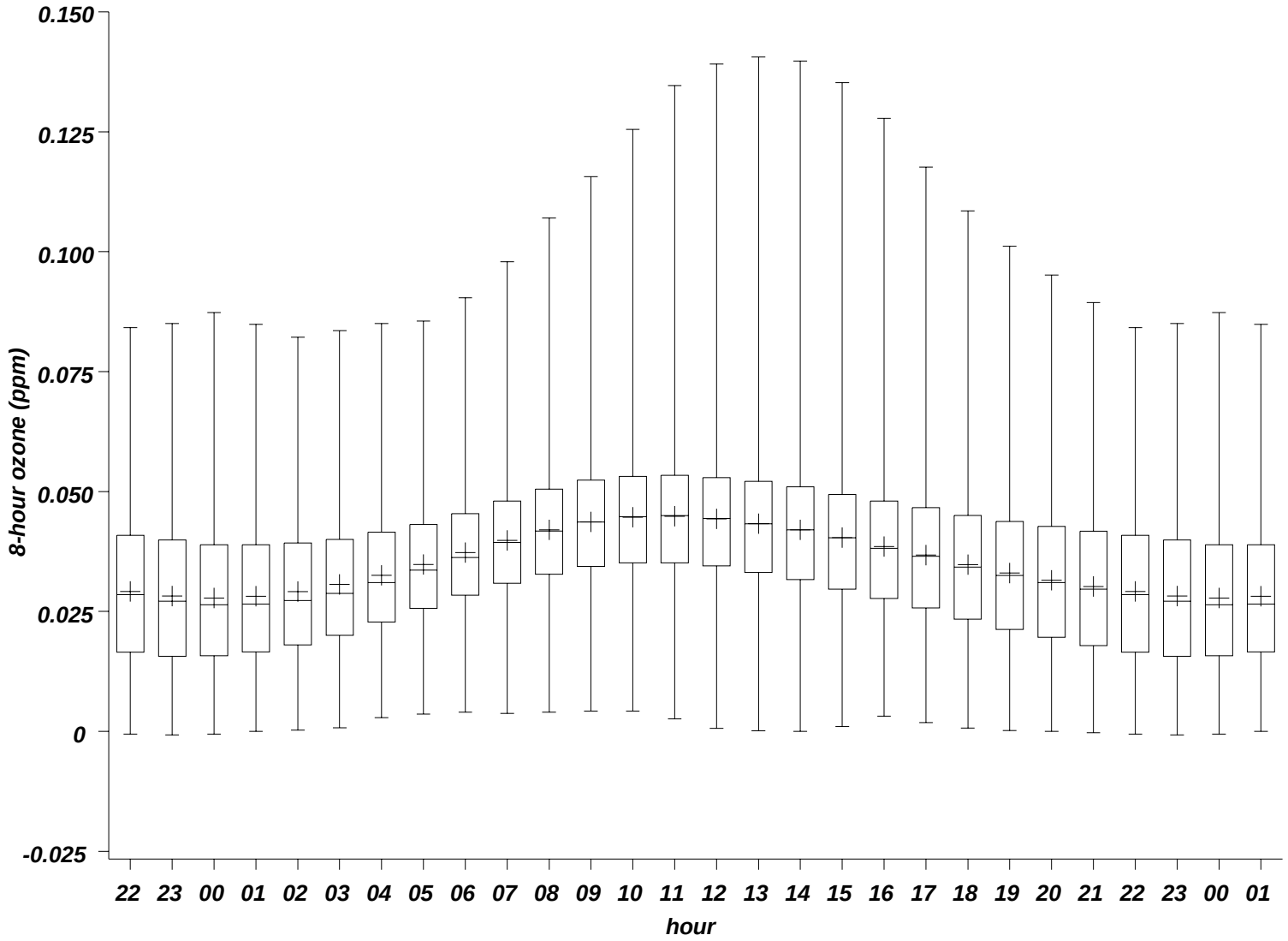

Figure 2-30. 8-Hour Week End Diurnal Pattern for Rural Sites, May through September 2004. Box Depicts interquartile range and median; whiskers depict maximum and minimum values; and '+' depicts the mean.

Data Source: CASTNET

eastern United States, high O_3 concentrations during an episode can extend over hundreds of thousands of square kilometers for several days.

Episodes have two main characteristics: the concentration level reached and the length of time that this level is reached in consecutive days. The following discussion addresses how these characteristics of episodes have varied through both space and time.

Numbers of episodes defined by daily maximum 1-hr O_3 concentrations reaching a level of 0.12 ppm for 1 day generally follow the long term trend of central values (means or medians) of the 1-hr O_3 data (See Figures 2-18 and 2-31). As the length of these episodes increases, the frequency of these episodes decreases. In the most recent years (1997-2004) episodes lasting 5 days or more often have not occurred at all (Fitz-Simons, et al., 2005). From this we conclude that control strategies have been reducing peak 1-hr O_3 across major urban areas.

Numbers of episodes defined by daily maximum 8-hr O_3 concentrations reaching a level of 0.08 ppm for 1 day generally follow the long term trend of central values of the 8-hr O_3 data (See Figures 2-16 and 2-32). As the length of these episodes increase, the frequency of these episodes decreases. However, some of the longer episodes (6 days of more) continue to occur at this level even in the most recent years. In fact, the episode must be defined by a level of 0.10 ppm before these longer episodes disappear in the most recent years (Fitz-Simons, et al., 2005).

As episode length and level increase for both 1-hr and 8-hr O_3 data the frequency decreases (Figure 2-33 and 2-34). The longer periods and higher levels disappear altogether in the period from 2000-2004 (Fitz-Simons, et al., 2005).

One final aspect of episodes to examine is the return time or the number of days between episodes. Looking at the intervals between episodes of 0.08ppm for 8-hr data, the most prevalent gap length in days is 1 day. There is a slight peak again at 4 days followed by a gradual decrease in frequency as the gap-length increases (see Figure 2-35). Looking at the same data for episodes of 0.12ppm, it appears that some periodicities appear at 1 day, 5-6 days, 21 days, and 33-34 days. The frequencies for these episodes are so small compared to frequencies lower level episodes that these indications should not be considered real or significant indications of periodicities. The 1-hr O_3 data exhibit much the same lack of periodicity as the 8-hr data (Fitz-Simons, et al., 2005).

2.7 POLICY RELEVANT BACKGROUND LEVELS

For purposes of this document, background or policy relevant background (PRB) O_3 is defined as the distribution of O_3 concentrations that would be observed in the U.S. in the absence of anthropogenic (man-made) emissions of precursor emissions (e.g., VOC, NOx, and CO) in the U.S., Canada, and Mexico. This is referred to as policy-relevant background, since this definition of background facilitates separating pollution levels that can be controlled by U.S.

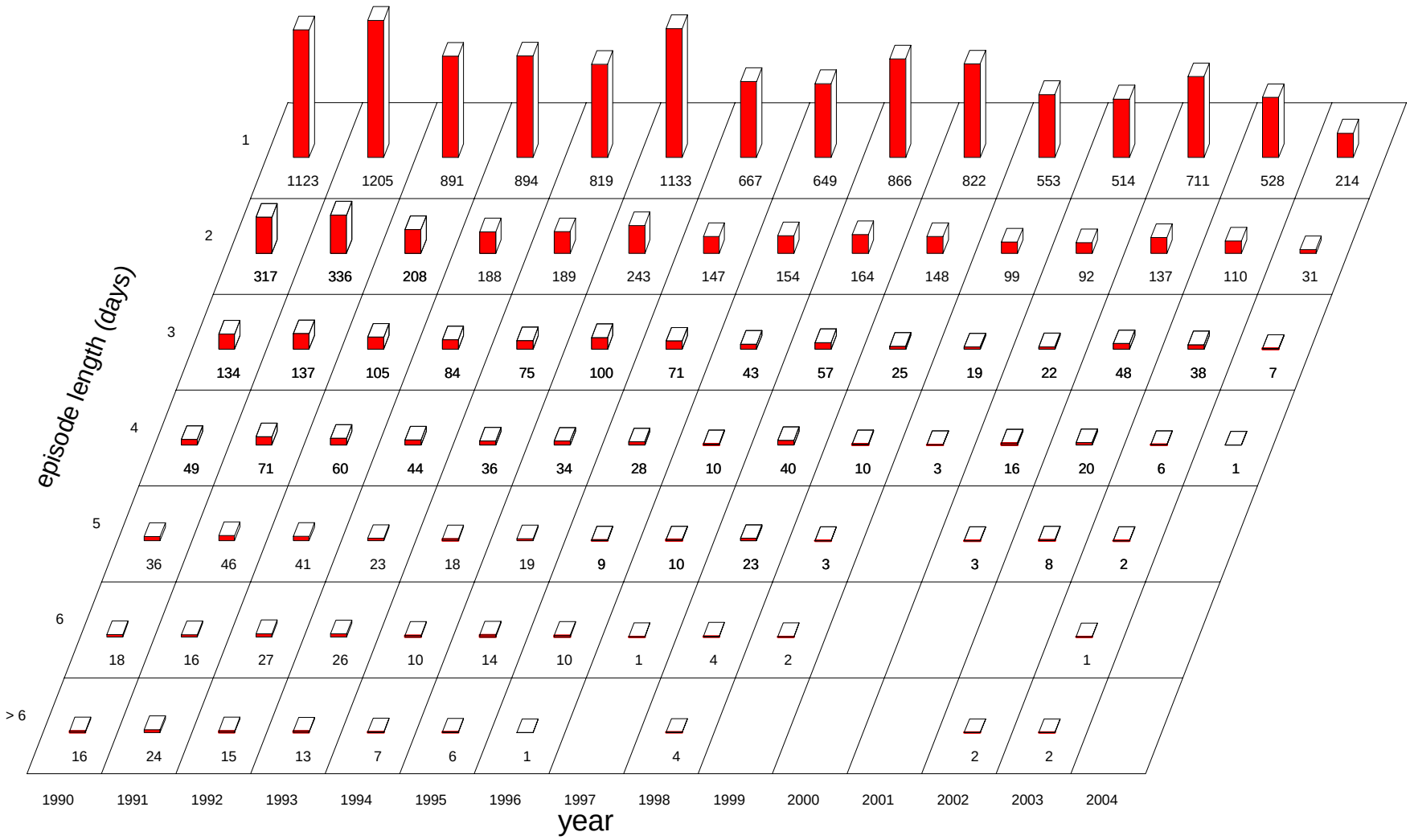

Figure 2-31. Length of Consecutive Day Episodes over 0.12 ppm by Year for 1-hour Ozone Data across all Monitors.

Data Source: AQS

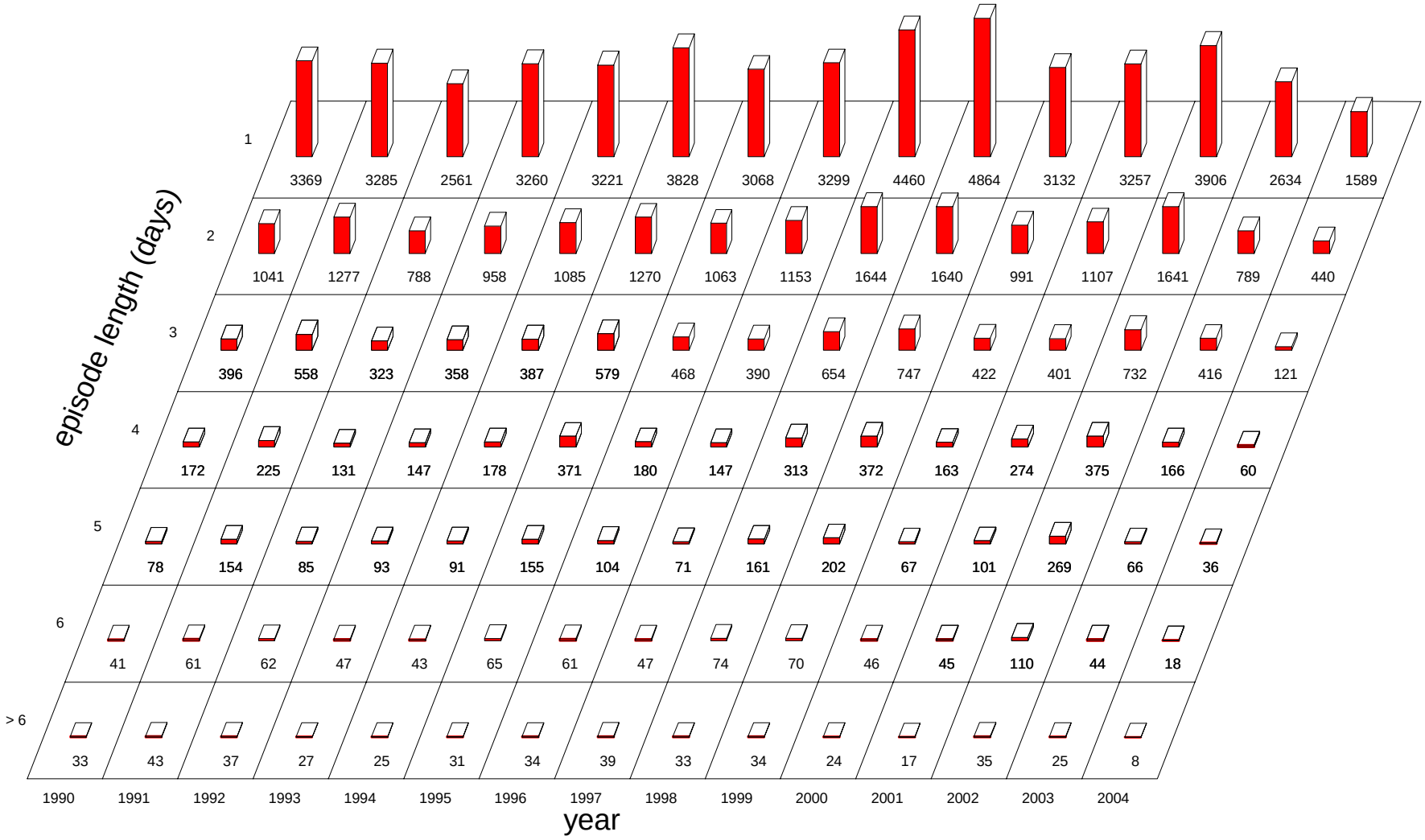

Figure 2-32. Length of Consecutive Day Episodes over 0.08 ppm by Year for 8-hour Ozone Data across all Monitors.

Data Source: AQS

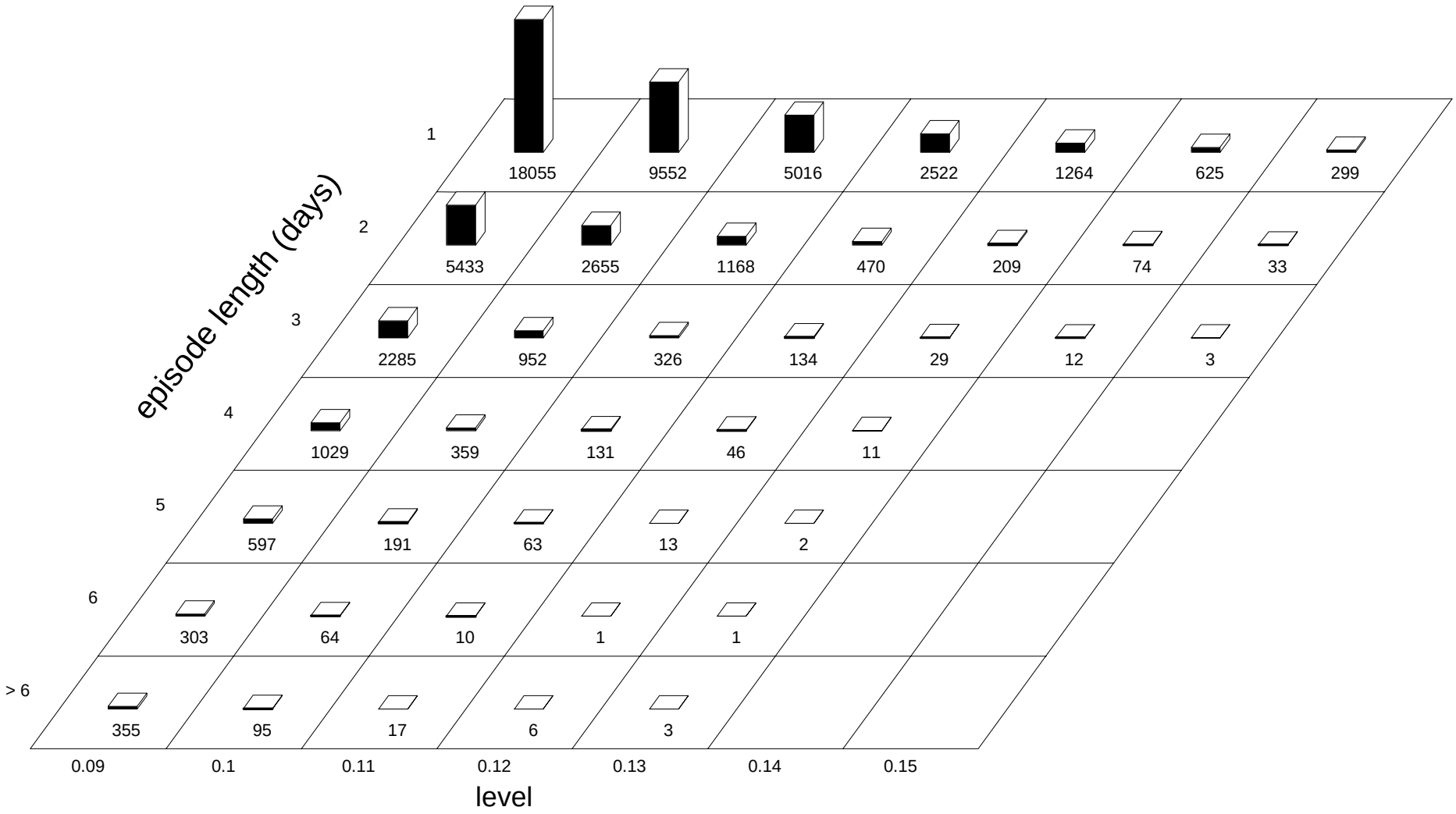

Figure 2-33. Length of Consecutive Day Episodes over Displayed Levels for 1-hour Ozone Data (2000-2004) across all Monitors.

Data Source: AQS

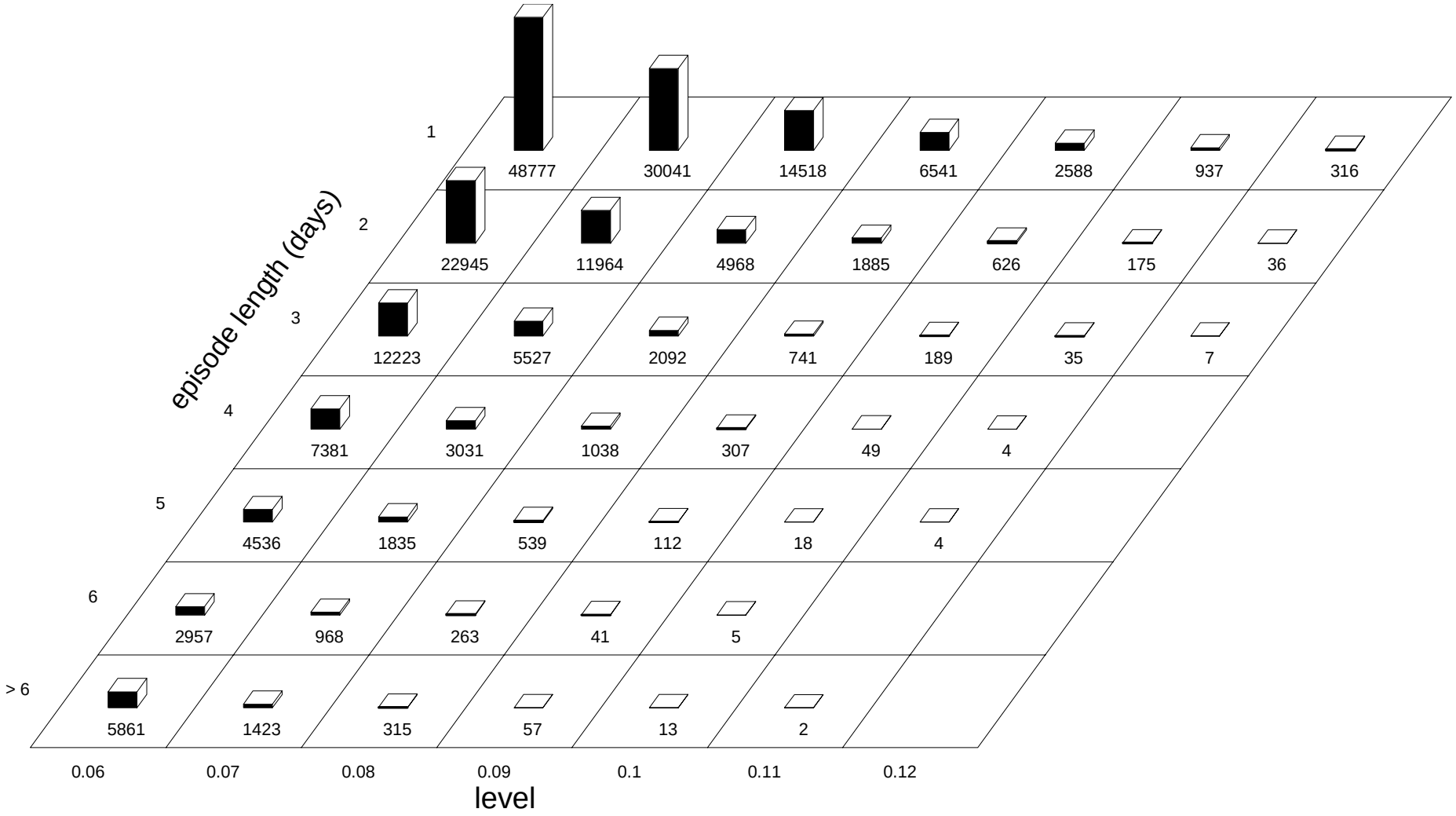

Figure 2-34. Length of Consecutive Day Episodes over Displayed Levels for 8-hour Ozone Data (2000-2004) across all Monitors.

Data Source: AQS

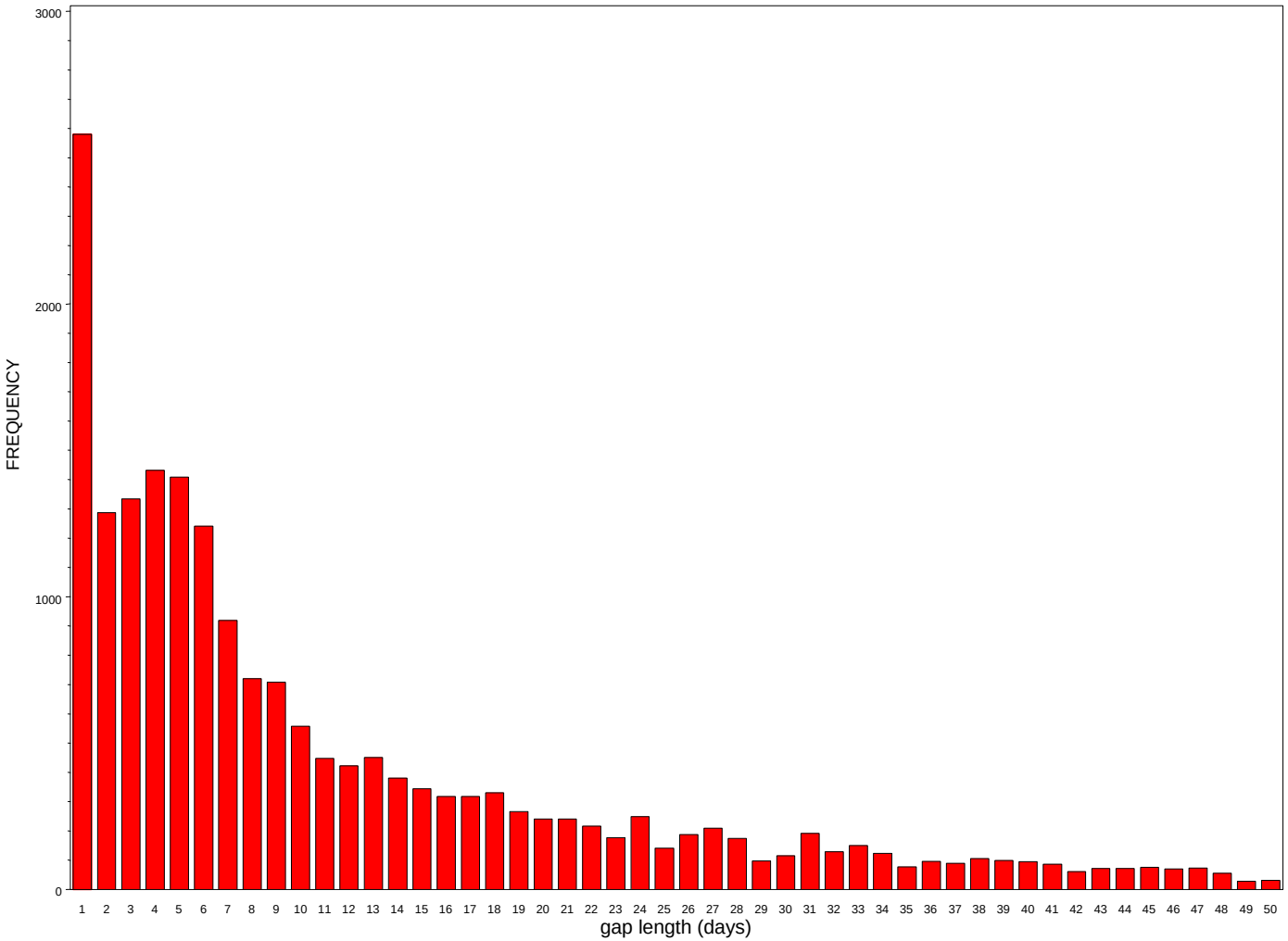

Figure 2-35. Length of Gaps in Days Between Episodes over 0.08 ppm for 8-hour Ozone Data (2000-2004).

regulations (or through international agreements with neighboring countries) from levels that are not generally controllable in this manner. As defined here, PRB includes (1) O_3 in the U.S. from natural sources of emissions in the U.S., Canada, and Mexico and (2) O_3 in the U.S. from the transport of O_3 or the transport of emissions from both natural and man-made sources, from outside of the U.S. and its neighboring countries. As discussed in Chapter 5 of this Staff Paper, PRB concentrations enter into the assessments of risk to human health.

Contributions to PRB levels of O_3 include: photochemical interactions involving natural emissions of VOCs, NO_X, and CO; the long-range transport of O_3 and its precursors from outside North America; and stratospheric-tropospheric exchange (STE). Processes involved in STE are described in detail in Annex AX2.3 of the CD. Natural sources of O_3 precursors include biogenic emissions, wildfires, and lightning. Biogenic emissions from agricultural activities are not considered in the formation of PRB (CD, p.AX2-145).

As a result of long-range transport of O_3 and its precursors from anthropogenic sources within North America, estimates of PRB O_3 concentrations cannot be derived solely from measurements of O_3, and must be based on modeling. The global photochemical transport model GEOS-CHEM (Fiore et al., 2003) has been applied to estimate PRB O_3 concentrations across the U.S. (U.S. EPA, 2005a, AX3-131). The CD refers to a number of GEOS-CHEM publications (Bey et al., 2001; Liu et al., 2002; Martin et al., 2002; Fusco and Logan, 2003; Li et al., 2002, 2005), summarizing their conclusions as "results indicate no significant bias, and agreement to generally within 5 ppbv (parts per billion volume) for monthly mean concentrations at different altitudes." The CD goes on to review detailed evaluations of GEOS-CHEM with O_3 observations at U.S. surface sites (Fiore et al., 2002, 2003) and comparisons of GEOS-CHEM predictions with observations at Trinidad Head, CA (Goldstein et al., 2004). The comparisons at Trinidad Head are especially relevant because sources of the O_3 found there are often limited to those in the PRB definition. The observations, filtered to remove local influence, averaged 41 ± 5 ppbv, as compared to GEOS-CHEM predictions of 39 ± 5 ppbv, indicating no significant differences between the model predictions and observations for conditions suggestive of PRB. The CD further notes that "several other papers have evaluated the GEOS-CHEM simulation for surface O_3 and its precursors over the United States." Summarizing their assessment of the validity of the GEOS-CHEM model, the CD states "in conclusion, we estimate that the PRB O_3 values reported by Fiore et al. (2003) for afternoon surface air over the United States are likely 10 ppbv too high in the southeast in summer, and accurate within 5 ppbv in other regions and seasons." These error estimates are based on comparison of model output with observations for conditions which most nearly reflect those given in the PRB definition, i.e., at the lower end of the probability distribution. For O_3 (cf. Figures 8 and 9 of Fiore et al., (2003) for the SE and

Figure 3 of Fiore et al. (2002) for the NE.) it can be seen that GEOS-CHEM overestimates O_3 for the SE and underestimates it for the NE.

The GEOS-CHEM model shows that PRB O_3 concentrations are related to season, altitude and total surface O_3 concentration. PRB O_3 concentrations at the surface are generally predicted to be in the range of 0.015 to 0.035 ppm in the afternoon, and they tend to decline under conditions conducive to high O_3 episodes. They are highest during spring and decline into summer. Higher values tend to occur at higher elevations during spring due to contributions from hemispheric pollution and stratospheric intrusions. The stratospheric contribution to surface O_3 is typically well below 0.020 ppm and only rarely elevates O_3 concentrations at low-altitude sites and only slightly more often elevates them at high-altitude sites (U.S. EPA, 2005a, AX3-148).

In the previous review of the O_3 NAAQS, the criteria document and staff paper adopted a value of 40 ppb for PRB O_3. However, Figure 3-17 in the CD shows that mean daily maximum 8-h O_3 concentrations were less than 40 ppb at over 10 % of U.S. sites. In 2004, the mean daily maximum 8-h O_3 concentrations were less than 40 ppb at 25 % of U.S. sites. It is highly unlikely that this fraction of O_3 monitoring sites would be unaffected by O_3 generated by sources from within continental North America. Figure 3-19, in the CD, shows that over the past 15 years, mean daily maximum 8-h O_3 concentrations at Voyageurs National Park were typically less than 40 ppb. Simulations of O_3 at Voyageurs, the site with the lowest O_3 show that there is still a substantial regional contribution to O_3 (Figure AX2-86). Thus, 40 ppb is likely to be too high for the mean PRB O_3 concentration.

The exposure and health risk analyses described in Chapter 4 and 5 use estimates of PRB based on runs of the GEOS-CHEM model applied for the 2001 warm season (i.e., April to September). The GEOS-CHEM data consist of hourly gridded values with latitude running from 12º to 80 º in 2 º steps and longitude running from -177.5 º to -47.5º in 2.5 º steps. These data are used to create monthly average diurnal profiles which are fixed for each month during the O_3 season. The PRB estimates from the grid nearest each of the 12 urban areas included in the exposure and risk analyses have been used to estimate PRB in each of these areas. Appendix 2A provides plots of the PRB estimates by month for each of the 12 urban areas.

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3. POLICY-RELEVANT ASSESSMENT OF HEALTH EFFECTS EVIDENCE

3.1 INTRODUCTION

This chapter assesses key policy-relevant information on the known and potential health effects associated with exposure to ambient O_3, alone and in combination with other pollutants that are routinely present in ambient air. This assessment focuses specifically on the health effects evidence evaluated in Chapters 4 through 7 of the CD with particular emphasis on the integrative synthesis presented in Chapter 8. That integrative synthesis focuses on integrating newly available scientific information with that available from the last review, as well as integrating information from various disciplines, to address a set of issues central to the assessment of scientific information upon which this review of the O_3 NAAQS is based. This chapter also addresses key issues relevant to quantitative assessment of controlled-human exposure and epidemiological evidence, to provide a foundation for the quantitative human exposure and health risk assessments presented in Chapters 4 and 5. Those quantitative assessments, together with this evidence-based assessment, provide the foundation for the development of staff conclusions and identification of options for consideration related to primary standards for O_3 presented in Chapter 6.

The decision in the last review focused primarily on evidence from short-term and prolonged controlled-exposure studies reporting lung function decrements, respiratory symptoms, and respiratory inflammation in humans, as well as epidemiology studies reporting excess hospital admissions and emergency department (ED) visits for respiratory causes. The CD prepared for this review emphasizes a large number of epidemiological studies published since the last review with these and additional health endpoints, including the effects of acute and chronic exposures to O_3 on premature mortality, enhanced respiratory symptoms and lung function decrements in asthmatic individuals, and school absences. It also emphasizes important new information from toxicology, dosimetry, and controlled human exposure studies.

As discussed in more detail below (section 3.3), highlights of the new evidence include:

- New controlled human-exposure studies have observed that very small (<5%) lung function decrements occur in some healthy adults under moderate exertion for 6.6 hr exposures to levels as low as 0.04 and 0.06 ppm; however, in a few subjects, decrements of >10% were observed.
- New controlled human-exposure studies offer evidence of increased airway responsiveness to allergens in subjects with allergic asthma and allergic rhinitis exposed to O_3.
- Numerous controlled human-exposure studies have reported indicators of O_3-induced inflammatory response in both the upper respiratory tract (URT) and lower respiratory

tract (LRT), while other studies have shown significant changes in host defense capability following O_3 exposure of healthy young adults.

- Animal toxicology studies provide new information regarding mechanisms of action, increased susceptibility to respiratory infection, and the biological plausibility of acute effects and chronic, irreversible respiratory damage.
- Numerous acute exposure epidemiological studies published during the past decade offer added evidence of ambient O_3-related lung function decrements and respiratory symptoms in physically active healthy subjects and asthmatic subjects, as well as evidence on new health endpoints, such as the relationships between ambient O_3 concentrations and school absenteeism and between ambient O_3 and cardiac physiologic endpoints.
- Several new studies have been published over the last decade examining the temporal associations between O_3 exposures and ED visits for respiratory diseases and on respiratory-related hospital admissions.
- Newly available, large multicity studies, designed specifically to examine the effects of acute exposure to PM and O_3 on mortality, provide much more robust and credible information than was available in the last review. The results from two key studies carried out in 95 U.S. communities (U.S. National Morbidity, Mortality Air Pollution Study [NMMAPS]) and in 23 European cities (Air Pollution and Health: European Approach [APHEA]) reported positive and significant O_3 effect estimates for all cause (nonaccidental) mortality.
- In a recent study, Bell et al. (2006) applied several statistical models to data on air pollution, weather, and mortality for the 98 NMMAPS communities to evaluate whether a threshold level exists for premature mortality. The results indicate that even low levels of tropospheric O_3 are associated with premature mortality.
- Three recent meta-analyses evaluated potential sources of heterogeneity in O_3-mortality associations, and these studies provide evidence of a robust association between ambient O_3 and mortality, especially for the warm O_3 season.

Section 3.2 provides an overview of mechanisms of toxicity, with more detailed discussion in Appendix 3A. Section 3.3 summarizes the nature of effects induced by O_3 exposure or associated with exposure to O_3, alone and in combination with other pollutants, drawing on information in Chapters 5-8 of the CD. Section 3.4 summarizes conclusions and judgments from the CD's integrative assessment of the epidemiological evidence regarding the extent to which causal inferences can be made about observed associations between health endpoints and exposure to O_3, and discusses key issues related to quantitative risk assessment based on such evidence. Section 3.5 discusses biological plausibility and coherence of evidence for O_3-related adverse health effects, including short-term respiratory effects, short-term cardiovascular effects,

long-term health effects, and mortality-related health endpoint. Drawing from the CD's integrative synthesis, section 3.6 discusses factors that modify responsiveness to O_3; potentially susceptible and vulnerable populations groups; and public health impacts of exposure to ambient O_3. Finally, section 3.7, summarizes key policy-relevant conclusions from the CD about O_3-related health effects, in the context of a discussion of issues related to our confidence in and the utility of the underlying evidence.

3.2 MECHANISMS OF TOXICITY

Evidence is covered in Chapters 5 and 6 of the CD on possible mechanisms by which exposure to O_3 may result in acute and chronic health effects. While most of the available evidence addresses mechanisms for O_3, we recognize that O_3 serves as an indicator for the total photochemical oxidant mixture found in the ambient air. Some effects may be caused by one or more components in the overall pollutant mix, either separately or in combination with O_3. Evidence from dosimetry, toxicology, and human exposure studies has contributed to an understanding of the mechanisms that help to explain the biological plausibility and coherence of evidence for O_3-induced respiratory health effects reported in epidemiological studies. In the past, however, little information was available to help explain potential biological mechanisms which linked O_3 exposure to premature mortality or cardiovascular effects. More recently, however, an emerging body of animal toxicology evidence is beginning to suggest mechanisms that may mediate acute O_3 cardiovascular effects.

Scientific evidence discussed in the CD (section 5.2) indicates that reactions with lipids and antioxidants are the initial step in mediating deleterious health effects of O_3. There is subsequent activation of a cascade of events starting with inflammation, altered permeability of the epithelial barrier, impaired host defense (including clearance mechanisms), and pulmonary structural alterations that can potentially exacerbate a preexisting disease status. According to the CD, the scientific evidence is still lacking for clearly establishing a role for one or a group of mechanistic pathways underlying O_3 health effects observed in epidemiological studies. Appendix 3A provides a further discussion of mechanisms of toxicity.

3.3 NATURE OF EFFECTS

The CD provides new evidence that notably enhances our understanding of short-term exposure effects, including effects on lung function, symptoms, and inflammatory effects reported in controlled exposure studies. These studies support and extend the findings of the previous CD. There is also a significant body of new epidemiological evidence of associations between short-term exposure to O_3 and effects such as premature mortality, hospital admissions

and ED visits for respiratory (e.g., asthma) causes. Key epidemiological and human controlled exposure studies are summarized in Appendices 3B and 3C, respectively.

The following discussions of O_3-related health effects are based on scientific evidence critically reviewed in chapters 5, 6, and 7 of the CD, as well as the CD's integration of scientific evidence contained in Chapter 8. In addition, these health effects discussions rely on the more detailed information and tables presented in the CD's annexes AX5, AX6, and AX7. Conclusions drawn about O_3-related health effects depend on the full body of evidence from controlled-exposure human, epidemiological and toxicological data contained in the CD. Section 3.3.1 focuses on a broad array of morbidity effects, including both acute and chronic exposures. Section 3.3.2 focuses on the expanded body of evidence on associations between acute O_3 exposure and mortality, as well as the more limited evidence on chronic O_3 exposures and mortality.

3.3.1 Morbidity

This section summarizes scientific information contained in the CD on respiratory and cardiovascular effects associated with exposure to O_3. Evidence of the effects of short-term and long-term exposure to O_3 on the respiratory system is discussed in sections 3.3.1.1 and 3.3.1.2, and evidence of O_3-related cardiovascular effects in section 3.3.1.3.

3.3.1.1 Effects on the Respiratory System from Short-term Exposures

Short-term exposures to O_3 have been reported to induce a wide variety of respiratory health effects. These effects include a range of effects, such as morphological changes in the respiratory tract, pulmonary function decrements, respiratory symptoms, respiratory inflammation, increased airway responsiveness, changes in host defense capability, acute morphological effects, increased ED visits and hospital admissions, and effects on exercise performance. Short-term O_3 exposure has also been associated with increases in restricted activity days and school absences but evidence is limited for these effects.

3.3.1.1.1 Pulmonary Function Decrements, Respiratory Symptoms, and Asthma Medication Use

A very large literature base of studies published prior to 1996, which investigated the health effects on the respiratory system from short-term O_3 exposures, was reviewed in the 1986 and 1996 CDs (U.S. Environmental Protection Agency, 1986, 1996). In the last review, the lowest O_3 concentration at which statistically significant reductions in forced vital capacity (FVC) and forced expiratory volume in 1 second (FEV_1) had been reported in sedentary subjects was 0.5 ppm (CD, p 6-3). During exercise, spirometric and symptomatic responses were observed at much lower O_3 exposures. When minute ventilation was considerably increased by

continuous exercise (CE) during O_3 exposures lasting 2 hr or less at $\geq$ 0.12 ppm, healthy subjects generally experienced decreases in FEV_1, FVC, total lung capacity (TLC), inspiratory capacity (IC), mean forced expiratory flow from 25% to 75% of FVC (FEF_{25-75}), and tidal volume (V_T); increases in specific airway resistance (sRaw), breathing frequency (f_B), and airway responsiveness; and symptoms such as cough, pain on deep inspiration, shortness of breath, throat irritation, and wheezing. When exposures were increased to 4- to 8-hr in duration, statistically significant spirometric and symptom responses were reported at O_3 concentrations as low as 0.08 ppm and at lower minute ventilation (i.e., moderate rather than high level exercise) than the shorter duration studies (CD. p. 6-6).

The most important observations drawn from studies reviewed in the 1996 CD were that: (1) young healthy adults exposed to O_3 concentrations $\geq$ 0.08 ppm develop significant, reversible, transient decrements in pulmonary function if minute ventilation or duration of exposure is increased sufficiently, (2) children experience similar spirometric responses but lesser symptoms from O_3 exposure relative to young adults, (3) O_3-induced spirometric responses are decreased in the elderly relative to young adults, (4) there is a large degree of intersubject variability in physiologic and symptomatic responses to O_3 but responses tend to be reproducible within a given individual over a period of several months, (5) subjects exposed repeatedly to O_3 for several days show an attenuation of response upon successive exposures; this attenuation is lost after about a week without exposure; and (6) acute O_3 exposure initiates an inflammatory response which may persist for at least 18 to 24 hr post exposure (CD, p. 6-2).

Since 1996, there have been a number of studies published investigating spirometric and symptomatic responses, and they generally support the observations previously drawn. Recent studies for acute exposures of 1 to 2 hr and 6 to 8 hr in duration are summarized in Tables AX6-1 and AX6-2 of the CD (p. AX6-5 to AX 6-7 and p. AX6-11 to AX6-12) and reproduced as Tables 3C-1 and 3C-2 in Appendix 3C. Among the more important of the recent studies was McDonnell et al. (1997) which examined reported changes in FEV_1 in 485 white males (ages 18-36) exposed for 2 hr to O_3 concentrations from as low as 0.08 ppm up to 0.40 ppm, at rest or with intermittent exercise (IE). Decrements in FEV_1 were modeled by sigmoid-shaped curve as a function of subject age, O_3 concentration, minute ventilation, and duration of exposure. In another study, Ultman et al. (2004) found that exposing 60 young, healthy subjects to 0.25 ppm O_3 for 1 hr with continuous exercise produced considerable intersubject variability in FEV_1 decrements ranging from 4% improvement to a 56% decrement, which was consistent with findings in the 1996 CD. One third of subjects had FEV_1 decrements > 15% and 7% had decrements > 40%. Foster et al. (1993, 1997) examined the effects of O_3 on ventilation distribution and reported results suggesting a prolonged O_3 effect on the small airways and ventilation distribution (CD, p. 6-5).

For prolonged exposures (4 to 8 hr) in the range of 0.08 to 0.16 ppm O_3 using moderate quasi-continuous exercise (QCE; 50 min exercise [minute ventilation of 35 to 40 L/min] and 10 min rest per hr), several pre- and post-1996 studies (Folinsbee et al., 1988,1994; Horstman et al., 1990; Adams, 2002, 2003a, 2006) have reported statistically significant spirometric responses and increased symptoms in healthy adults with increasing duration of exposure, O_3 concentration, and minute ventilation. Based on review of several prolonged exposure studies, the CD (p. 6-6) concluded that FEV_1 decrements are a function of minute ventilation in 6.6 hr exposure studies and that data from recent studies do not support the contention that minute ventilation should be normalized to body surface area (BSA) for adults. Triangular exposure (i.e., integrated exposures that begin at a low level, rise to a peak, and return to a low level during time of exposure) studies (Hazucha et al., 1992; Adams 2003a, 2006) suggest that, depending upon the profile of the exposure, the triangular exposure, which may reflect the pattern of ambient exposures in some locations, can potentially lead to greater FEV_1 decrements than square wave exposures (i.e., a constant exposure level during time of exposure) when the overall O_3 doses are equal (CD, p. 6-10), suggesting that peak exposures are important in terms of O_3 toxicology.

McDonnell (1996) used data from a series of studies to investigate the frequency distributions of FEV_1 decrements following 6.6 hr exposures and found that average FEV_1 responses were relatively small (between 5 and 10 %) at 0.08 ppm O_3 (CD, p. 8-17)[1]. However, about 18% of the exposed subjects had moderate functional decrements (10 to 20%), and about 8% experienced large decrements (>20%). Figure 3-1A,B,C (CD, Figures 8-1A,B and 8-2, pp. 8-17 and 8-19) is based on study data that are in McDonnell (1996) together with data from Adams (2002, 2006) that were not published but were obtained from the author. This figure demonstrates that while average responses may appear small and insignificant, some individuals can experience much more significant and severe effects that may be clinically significant. The FEV_1 responses illustrated in this figure were not corrected for the effect of exercise in clear air. When that is done for the Adams (2002, 2006) data, the percentage of subjects experiencing $\geq$10% FEV_1 decrements changes to 7% at O_3 exposures of 0.04 ppm, to 7% at O_3 exposures of

[1] The studies conducted in EPA's clinical research facility in Chapel Hill, NC that are considered in the lung function risk assessment measured ozone concentrations to within +/- 5% or +/- 0.004 ppm at the 0.08 ppm exposure level. The accuracy of these measurements was confirmed in an email sent by Steve Jackson, Program Manager for the Human Studies Division in the National Health & Environmental Effects Research Laboratory, Office of Research and Development, USEPA. He has overall responsibility for the monitoring program at the USEPA clinical research facility in Chapel Hill, NC. This email has been placed in the Ozone NAAQS review docket.

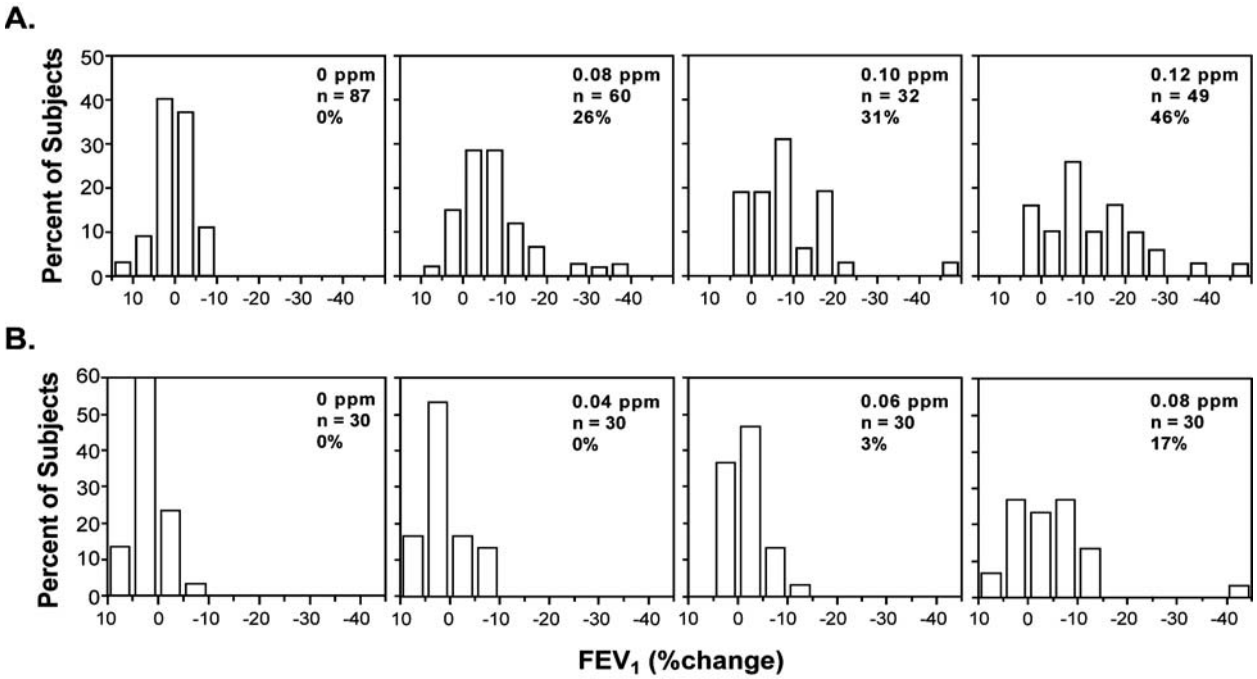


Figure 3-1A and B. Frequency distributions of FEV_1 changes following 6.6-h exposures to a constant concentration of O_3 or filtered air. Note that the percentage in each panel indicates the distributions of % decrement.

Source:Panel A, McDonnell (1996); Panel B, Adams (2002, 2006), pre- and post-FEV_1 data for each subject provided by author.

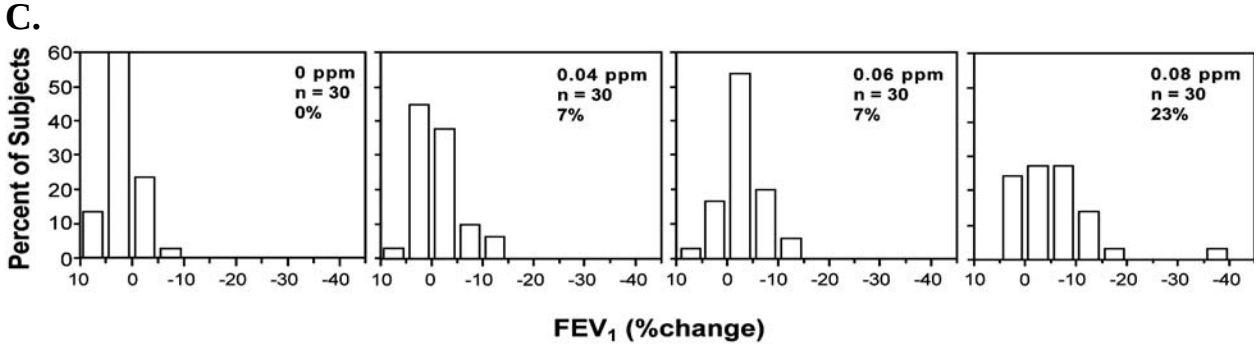


Figure 3-1C. Frequency distributions of FEV_1 changes following 6.6-h exposures to a constant concentration of O_3 or filtered air. The FEV_1 changes following O_3 exposures have been corrected for filtered air responses, i.e., they are O_3-induced FEV_1 changes. Note that the percentage in each panel indicates the distributions of % decrement.

Source: Adams (2002, 2006), pre- and post- FEV_1 data for each subject provided by author.

0.06 ppm, and to 23% at O_3 exposures of 0.08 ppm, in studies conducted in California (CD, p. 8-18). The development of these effects is time-dependent during both exposure and recovery periods, with great overlap for development and disappearance of the effects. In healthy human subjects exposed to typical ambient O_3 levels near 0.12 ppm, spirometric responses largely resolve within 4 to 6 hr postexposure, but cellular effects persist for about 24 hr. In these healthy subjects, small residual lung function effects are almost completely gone within 24 hr, while in hyperresponsive subjects, recovery can take as much as 48 hr to return to baseline. The majority of these responses are attenuated after repeated exposure, but such attenuation to O_3 is lost one week postexposure (CD, p. 8-19).

In the Adams (2006) investigation of the effects of square-wave (0.00, 0.06, and 0.08 ppm O_3) and triangular (averaging 0.04, 0.06, and 0.08 ppm O_3) exposures for 6.6 hr during quasi continuous exercise on pulmonary function in 30 healthy adults, the study was designed to compare pulmonary function responses between the six exposure protocols at each of six time points (1, 2, 3, 4.6, 5.6, and 6.6 hr)[2]. Accordingly, the author utilized a multiple comparison technique to avoid Type I error (falsely rejecting the null hypothesis of no difference). At 6.6 hr, FEV1 responses from the 0.08 ppm O_3 exposures were found to be significantly different from the responses observed for the 0.0, 0.04, and 0.06 ppm O_3 exposures. The FEV_1 responses did not differ significantly at 6.6 hr between the two 0.08 ppm O_3 exposures (i.e., the square-wave vs. the triangular). Another statistically insignificant comparison was between the FEV_1 responses at 0.06 ppm O_3 and filtered air (0.0 ppm O_3) at 6.6 hr.

On examination of the group mean FEV_1 responses in Figure 1 of Adams (2006), however, responses during the 0.06 ppm O_3 exposures appear to diverge from responses for filtered-air and 0.04 ppm O_3 (CD, 8-42). In addition to reducing Type I error, the correction for the multiple comparisons by Adams (2006) may have also increased Type II error (falsely accepting the null) for the simple evaluation of pre- to postexposure effects of O_3 versus filtered air on FEV_1, as has been commonly assessed by others (e.g., Horstman et al., 1990; McDonnell et al., 1991). A cursory evaluation of pre- to postexposure effects can be completed utilizing the summary data in Table 3 of the Adams (2006) publication. For the filtered air, 0.06 ppm O_3 (square-wave), and 0.06 ppm O_3 (triangular) exposures, the FEV_1 responses were 1.35±0.54 [mean±standard error (SE)], −1.51±0.77, and −1.43±1.09%, respectively. Under the null hypothesis of no pre- to postexposure difference in FEV_1 responses between filtered air and O_3 exposure, the lack of an overlap in the range of responses (i.e., the means±SEs) at 0.06 ppm O_3 versus filtered air is suggestive of a significant effect on FEV_1. Furthermore, in a prior publication (Adams, 2002), the author stated that, “some sensitive subjects experience notable

[2] These studies reported O_3 concentrations to be accurate within +/- 0.003 ppm over the range of concentrations included in these studies.

effects at 0.06 ppm," based on the observation that 20% of subjects exposed to 0.06 ppm O_3 (in a face mask exposure study) had greater than a 10% decrement in FEV_1 even though the group mean response was not statistically different from the filtered air response. The effects described by Adams (2002), along with the cursory evaluation of the Adams (2006) data as described above, strongly suggest that exposure to 0.06 ppm O_3 causes small group mean FEV_1 decrements in healthy adults with some individuals having notable effects.

Although not mentioned in the CD, Adams (2006) reported that total subjective symptom scores (TSS) during the triangular 0.06 ppm exposure reached statistical significance (relative to preexposure) at 5.6 and 6.6 hr, whereas they did not reach significance during the square-wave 0.06 ppm exposure. Data in Table 4 of the Adams (2006) publication allow further evaluation of pre- to postexposure effects on respiratory symptoms, both TSS and pain on deep inspiration (PDI). For the filtered air, 0.06 ppm (square-wave), and 0.06 ppm (triangular) exposures, the TSS responses were 0.6±0.40 (mean±SE), 2.5±0.89, and 3.9±1.35, respectively, and the PDI responses were 0.2±0.16 (mean±SE), 1.4±0.53, and 2.0±0.80, respecitively. As noted above for FEV_1 changes, the lack of an overlap in the ranges of responses (i.e., the means±SEs) at 0.06 ppm O_3 versus filtered air for those two sympom scores is suggestive of a significant effect on respiratory symptoms.

A relatively large number of field studies investigating the effects of ambient O_3 concentrations, in combination with other air pollutants, on lung function decrements and respiratory symptoms have been published since 1996 (see CD, sections 7.2.3, 7.2.4, and 8.4.4.1). These newer studies support the major findings of the 1996 CD that lung function changes, as measured by decrements in FEV_1 or peak expiratory flow (PEF), and respiratory symptoms in healthy adults and asthmatic children are closely correlated to ambient O_3 concentrations. Pre-1996 field studies focused primarily on children attending summer camps and found O_3-related impacts on measures of lung function, but not respiratory symptoms, in healthy children. The newer studies have expanded to evaluate O_3-related effects on outdoor workers, athletes, the elderly, hikers, school children, and asthmatics. Collectively, these studies confirm and extend clinical observations that prolonged exposure periods, combined with elevated levels of exertion or exercise, may magnify the effect of O_3 on lung function. The most representative data come from the hiker study (Korrick et al., 1998), which provided outcome measures stratified by several factors (e.g., gender, age, smoking status, presence of asthma) within a population capable of more than normal exertion. In this study, lung function was measured before and after hiking, and both ambient and personal O_3 exposure measurements were made. The mean 8-hr average O_3 concentration was 40 ppb (SD 12). Decreased lung function was associated with O_3 exposure, with the greatest effect estimates reported for the

subgroup that reported having asthma or wheezing, and for those who hiked for longer periods of time, thus increasing the exposure period (CD, p. 7-36).

Asthma panel studies, conducted both in the U.S. and in other countries, have reported that decrements in PEF are associated with routine O_3 exposures among asthmatic and healthy persons (CD, sections 7.2.3.2 and 8.4.4.1). One large U.S. multicity study (Mortimer et al., 2002) examined O_3-related changes in PEF in 846 asthmatic children from 8 urban areas and reported that the incidence of $\geq$ 10% decrements in morning PEF are associated with a 30 ppb increase in 8-hr average O_3 for a 5-day cumulative lag, suggesting that O_3 exposure may be associated with clinically significant changes in PEF in asthmatic children; however, no associations were reported with evening PEF (CD, p. 7-43). The authors also reported that the associations reported with morning PEF remained statistically significant when days with 8-hr O_3 concentrations above 80 ppb were excluded (CD, p. 7-46). Two studies (Romieu et al., 1996, 1997) carried out simultaneously in northern and southwestern Mexico City with mildly asthmatic school children reported statistically significant O_3-related reductions in PEF, with variations in effect depending on lag time and time of day. In the northern study, the mean 1-hr max O_3 concentrations were 190 ppb (SD 80), and in the southwestern study, mean 1-hr max levels were 196 ppb (SD 78). While several studies (Gielen et al., 1997; Jalaludin et al., 2000; Ross et al., 2002; Thurston et al., 1997) report statistically significant associations between O_3 exposure and reduced PEF in asthmatics, other studies (Hiltermann et al., 1998; Delfino et al., 1997a) did not, possibly due to very low levels of O_3. Collectively, however, these studies indicate that O_3 may be associated with short-term declines in lung function in asthmatic individuals and that they occurred at concentrations below those used in chamber studies using exercise (CD, p. 7-40 to 7-46).

Mortimer et al. (2002) discussed biological mechanisms for delayed effects on pulmonary function in asthma, which included increased nonspecific airway responsiveness secondary to airway inflammation due to O_3 exposure (CD, p. 7-43). Animal toxicological and human chamber studies (CD, Chapters 5 and 6) provide supporting evidence that exposure to O_3 may augment cellular infiltration and cellular activation, enhance release of cytotoxic inflammatory mediators, and alter membrane permeability (CD, p.7-44). In most laboratory animals studied, biochemical markers of lung injury and associated morphological changes were not found to be attenuated, even though at similar exposures pulmonary function changes might be attenuated.

Most of the panel studies which have investigated associations between O_3 exposure and respiratory symptoms or increased use of asthma medication are focused on asthmatic children (CD, sections 7.2.4 and 8.4.4.1). Two large U.S. studies (Mortimer et al., 2002; Gent et al., 2003), as well as several smaller U.S. (Delfino et al., 2003; Just et al., 2002; Newhouse et al., 2004; Romieu et al., 1996, 1997; Ross et al., 2002; Thurston et al., 1997) and international

studies (Hilterman et al., 1998; Desqueyroux et al., 2002a,b), have reported fairly robust associations between ambient O_3 concentrations and daily symptoms/asthma medication use, even after adjustment for copollutants.

The National Cooperative Inner-City Asthma Study (NCICAS) reported morning symptoms in 846 asthmatic children from 8 U.S. urban areas to be most strongly associated with a cumulative 1- to 4-day lag of O_3 concentrations (Mortimer et al., 2002). The NCICAS used standard protocols that included instructing caretakers of the subjects to record symptoms in the daily diary by observing or asking the child (Mitchell et al., 1997). Symptoms reported included cough, chest tightness, and wheeze. In the analysis pooling individual subject data from all eight cities, the odds ratio for the incidence of symptoms was 1.35 (95% CI: 1.04, 1.69) per 30 ppb increase in 8-hr avg O_3 (10 a.m.-6 p.m.). The mean 8-hr avg O_3 was 48 ppb across the 8 cities. Excluding days when 8-hr avg O_3 was greater than 80 ppb (less than 5% of days), the odds ratio was 1.37 (95% CI: 1.02, 1.82) for incidence of morning symptoms.

Gent and colleagues (2003) followed 271 asthmatic children under age 12 and living in southern New England for 6 months (April through September) in a diary study of daily symptoms in relation to O_3 and $PM_{2.5}$. Mean 1-hr max O_3 and 8-hr max O_3 concentrations were 58.6 ppb (SD 19.0) and 51.3 ppb (SD 15.5), respectively. The data were analyzed for two separate groups of subjects, 130 who used maintenance asthma medications during the follow-up period and 141 who did not. The need for regular medication was considered to be a proxy for more severe asthma. Not taking any medication on a regular basis and not needing to use a bronchodilator would suggest the presence of very mild asthma. Effects of 1-day lag O_3 were observed on a variety of respiratory symptoms only in the medication user group. Both daily 1-hr max and 8-hr max O_3 concentrations were similarly related to symptoms such as chest tightness and shortness of breath. Effects of O_3, but not $PM_{2.5}$, remained significant and even increased in magnitude in two-pollutant models. Some of the associations were noted at 1-hr max O_3 levels below 60 ppb. In contrast, no effects were observed among asthmatics not using maintenance medication. In terms of person days of follow-up, this is one of the larger studies currently available that address symptom outcomes in relation to O_3, and provides supportive evidence for effects of O_3 independent of $PM_{2.5}$. Study limitations include limited control for meteorological factors and the post-hoc nature of the population stratification by medication use (CD, p. 7-53).

The multicities study by Mortimer et al. (2002), which provides an asthmatic population most representative of the United States, and several single-city studies indicate a robust association of O_3 concentrations with respiratory symptoms and increased medication use in asthmatics. While there are a number of well-conducted, albeit relatively smaller, studies which showed only limited or a lack of evidence for symptom increases associated with O_3 exposure,

these studies had less statistical power and/or were conducted in areas with relatively low O_3 levels (CD, p. 7-54). The CD (p. 7-55) concludes that the asthma panel studies, as a group, and the NCICAS in particular, indicate a positive association between ambient concentrations and respiratory symptoms and increased medication use in asthmatics. The evidence has continued to expand since 1996 and now is considered to be much stronger than in the previous review of the O_3 primary standard.

The association between school absenteeism and ambient O_3 concentrations was assessed in two relatively large field studies (CD, section 7.2.6). Chen et al. (2000) examined daily school absenteeism in 27,793 elementary school students in Nevada over a 2-year period (after adjusting for PM_{10} and CO concentrations) and found that ambient O_3 concentrations were associated with 10.41% excess rate of school absences per 40 ppb increase in 1-hr max O_3 with a distributed lag of 1 to 14 days. Gilliland et al. (2001) studied O_3-related absences among 1,933 4^{th} grade students in 12 southern California communities and found significant associations between 30-day distributed lag of 8-hr average O_3 concentrations and all absence categories, and particularly for respiratory causes. Neither PM_{10} nor NO_2 were associated with any respiratory or nonrespiratory illness-related absences in single pollutant models. The CD concludes that these studies of school absences suggest that ambient O_3 concentrations, accumulated over two to four weeks, may be associated with school absenteeism, and particularly illness-related absences, but further replication is needed before firm conclusions can be reached regarding the effect of O_3 on school absences (CD, p. 7-60).

3.3.1.1.2 Airway Responsiveness

Airway hyperresponsiveness (AHR), also know as bronchial hyperreactivity, refers to a condition in which the propensity for the airways to bronchoconstrict due to a variety of stimuli (e.g., exposure to cold air, allergens, or exercise) becomes augmented (CD, section 6.8). This condition is typically quantified by measuring the decrement in pulmonary function (e.g., spirometry or plethysmography) after inhalation exposure to specific (e.g., antigen, allergen) or nonspecific (e.g., methacholine, histamine) bronchoconstrictor stimuli. Exposure to O_3 causes an increase in nonspecific airway responsiveness as indicated by a reduction in the concentration of methacholine or histamine required to produce a given reduction in FEV_1 or increase in SRaw. Increased airway responsiveness is an important consequence of exposure to O_3 because its presence means that the airways are predisposed to narrowing on inhalation of various stimuli, such as specific allergens, cold air or SO_2 (CD, p. 8-21). Significant, clinically relevant decreases in pulmonary function have been observed in early phase allergen response in subjects with rhinitis after consecutive (4-day) exposure to 0.125 ppm O_3 (Holz et al., 2002). Similar increased airway responsiveness in asthmatics to house dust mite antigen 16 to 18 hrs after

exposure to a single dose of O_3 (0.16 ppm for 7.6 hrs) was observed. These observations suggest that O_3 exposure may be a clinically important factor that can exacerbate the response to ambient bronchoconstrictor substances in individuals with preexisting allergic asthma. Further, O_3 may have an immediate impact on asthmatics as well as contribute to effects that persist for longer periods (CD, p. 8-21).

An important aspect of increased airway responsiveness after O_3 exposure is that it represents a plausible link between O_3 exposure and increased hospital admissions. Kreit et al. (1989) found that O_3 can induce increased airway responsiveness in asthmatic subjects to O_3, who typically have increased airway responsiveness at baseline. A subsequent study (Jorres et al., 1996) suggested an increase in specific (i.e., allergen-induced) airway reactivity in subjects with allergic asthma, and to a lesser extent in subjects with allergic rhinitis after exposure to 0.25 ppm O_3 for 3 hrs; other studies (Molfino et al., 1991; Kehrl et al., 1999) reported similar results. According to one study (Folinsbee and Hazucha, 2000), changes in airway responsiveness after O_3 exposure resolve more slowly than changes in FEV_1 or respiratory symptoms. Other studies of repeated exposure to O_3 suggest that changes in airway responsiveness tend to be somewhat less affected by attenuation with consecutive exposures than changes in FEV_1 (Dimeo et al., 1981; Folinsbee et al., 1994; Gong et al., 1997a; Kulle et al., 1982) (CD, p. 6-31).

An extensive laboratory animal data base exploring the effects of acute, long-term, and repeated exposure, at rest, to O_3 indicates that induction of AHR occurs at relatively high ($\geq$1ppm) O_3 concentrations (p. 8-21). These studies provide clues to the roles of physiological and biochemical components involved in this process, but caution should be exercised in interpreting these results, as different mechanisms may be involved in mediating high- and low-dose responses. As observed in humans, the acute changes in AHR do not persist after long-term exposure of animals exposed to near-ambient concentrations of O_3, and attenuation has been reported. In addition, dosimetric adjustments potentially could be made to allow better estimation of levels that would be relevant to human exposure effect levels.

The CD concludes that O_3 exposure is linked with increased AHR (CD, section 6.8). Both human and animal studies indicate that AHR is not mechanistically associated with inflammation, but they do suggest a likely role for neuronal involvement (CD, p. 8-21). Increases in AHR do not appear to be strongly associated with decrements in lung function or increases in symptoms (CD, p. 6-31).

3.3.1.1.3 Respiratory Inflammation and Permeability

Based on evidence from the previous review, acute inflammatory responses in the lung have been observed subsequent to 6.6 hr O_3 exposures to the lowest tested level of 0.08 ppm in healthy adults engaged in moderately high exercise. Some studies suggest that inflammatory

responses may be detected in some individuals following O_3 exposures in the absence of O_3-induced pulmonary decrements in those subjects. Short-term exposures to O_3 also can cause increased permeability in the lungs of humans and experimental animals (CD, sections 5.2.3, 6.9, 7.2.5 and 8.4.3). Not only are the newer findings consistent with the previous review, but also there is better characterization of the physiological mechanisms by which O_3 causes these effects.

Lung inflammation and increased permeability, which are distinct events controlled by different mechanisms, are two well characterized effects of O_3 exposure observed in all species studied. Disruption of the lung barrier leads to leakage of serum proteins, influx of polymorphonuclear leukocytes (PMNs), release of bioactive mediators, and movement of compounds from the airspaces into the blood.

In the animal toxicological studies discussed in the CD (Chapter 5), the lowest O_3 concentration that induced inflammation in the mouse lung was 0.11 ppm for 24 hr exposures. Shorter exposures of 8 hours required concentrations of 0.26 ppm to induce epithelial permeability, although there was no effect on inflammation. The lowest O_3 concentration that affected epithelial permeability or inflammation in the rat was 0.5 ppm for a 3 hr exposure or 0.12 ppm for 6 hr (CD, p. 8-23). After acute exposures, the influence of the duration of exposure increases as the concentration of O_3 increases; however, dosimetric adjustments would need to be done before one can compare levels. The exact role of inflammation in causation of lung disease is not known; nor is the relationship between inflammation and lung function (CD, p. 5-23).

A number of human O_3-exposure studies have analyzed bronchoalveolar lavage (BAL) and nasal lavage (NL) fluids and cells for markers of inflammation and lung damage. These studies are summarized in the CD (Annex AX6, Tables AX6-12 and AX6-13). Increased lung inflammation is demonstrated by the presence of neutrophils (PMNs) found in BAL fluid in the lungs, which has long been accepted as a hallmark of inflammation. It is apparent, however, that inflammation within airway tissues may persist beyond the point that inflammatory cells are found in the BAL fluid. Soluble mediators of inflammation, such as cytokines and arachidonic acid metabolites have been measured in the BAL fluid of humans exposed to O_3. In addition to their role in inflammation, many of these compounds have bronchoconstrictive properties and may be involved in increased airway responsiveness following O_3 exposure (CD, p. 6-31, p. 8-22). An in vitro study of epithelial cells from nonatopic and atopic asthmatics exposed to 0.01 to 0.10 ppm O_3 showed significantly increased permeability compared to cells from normal persons. This indicates a potentially inherent susceptibility of cells from asthmatic individuals for O_3-induced permeability.

In the 1996 CD, assessment of human exposure studies indicated that a single, acute (1 to 4 hr) O_3 exposure ($\geq$ 0.08 to 0.1 ppm) of subjects engaged in moderate to heavy exercise could

induce a number of cellular and biochemical changes suggestive of pulmonary inflammation and lung permeability (CD, p. 8-22). These changes persisted for at least 18 hrs. Graham and Koren (1990) compared inflammatory mediators present in NL and BAL fluids of humans exposed to 0.4 ppm O_3 for 2 hrs and found similar increases in PMNs in both fluids, suggesting a qualitative correlation between inflammatory changes in the lower airways (BAL) and upper respiratory tract (NL). Acute airway inflammation was shown in Devlin et al. (1990) to occur among adults exposed to 0.08 ppm O_3 for 6.6 hr with exercise, and McBride et al. (1994) reported that asthmatic subjects were more sensitive than non-asthmatics to upper airway inflammation for O_3 exposures (0.24 ppm, 1.5 hr, with light IE) that did not affect pulmonary function (CD, p. 6-33).

The studies reporting inflammatory responses and markers of lung injury have clearly acknowledged that there is significant variation in response of subjects exposed, especially to 6.6 hour O_3 exposures at 0.08 and 0.10 ppm. To provide some perspective on the public health impact for these effects, we note that one study (Devlin et al., 1991, Figure 5) showed that roughly 10 to 50% of the 18 young healthy adult subjects experienced notable increases (i.e., $\geq$ 2 fold increase) in most of the inflammatory and cellular injury indicators analyzed, associated with 6.6-hour exposures at 0.08 ppm. Similar, although in some cases higher, fractions of the population of 10 healthy adults tested saw > 2 fold increases associated with 6.6-hour exposures to 0.10 ppm. The authors of this study suggest that "susceptible subpopulations such as the very young, elderly, and people with pulmonary impairment or disease may be even more affected" (Devlin et al., 1991).

Since 1996, a substantial number of human exposure studies have been published which have provided important new information on lung inflammation and epithelial permeability. Mudway and Kelly (2004) examined O_3-induced inflammatory responses and epithelial permeability with a meta-analysis of 21 controlled human exposure studies and showed that PMN influx in healthy subjects is associated with total O_3 dose (product of O_3 concentration, exposure duration, and minute ventilation) (CD, p. 6-34). Results of the analysis suggest that the time course for inflammatory responses (including recruitment of neutrophils and other soluble mediators) is not clearly established, but differential attenuation profiles for many of these parameters are evident (CD, p. 8-22).

A number of studies (Peden et al., 1997; Scannell et al., 1996; Hiltermann et al., 1999; Bosson et al., 2003) have provided evidence suggesting that asthmatics show greater inflammatory response than healthy subjects when exposed to similar O_3 levels (CD, section 6.9). Markers from BAL fluid following both 2-hr (Devlin et al., 1997) and 4-hr (Christian et al., 1998; Jorres et al., 2000) O_3 exposures repeated up to 5 days indicate that there is ongoing cellular damage irrespective of attenuation of some cellular inflammatory responses of the airways, pulmonary function, and symptom responses (CD, p. 8-22).

The CD (p. 8-24) concludes that interaction of O_3 with lipid constituents of epithelial lining fluid (ELF) and cell membranes and the induction of oxidative stress is implicated in injury and inflammation. Alterations in the expression of cytokines, chemokines, and adhesion molecules, indicative of an ongoing oxidative stress response, as well as injury repair and regeneration processes, have been reported in animal toxicology and human in vitro studies evaluating biochemical mediators implicated in injury and inflammation. While antioxidants in ELF confer some protection, O_3 reactivity is not eliminated at environmentally relevant exposures. Further, antioxidant reactivity with O_3 is both species-specific and dose-dependent (CD, p. 8-24).

3.3.1.1.4 Changes in Host Defense Capability

As discussed in the CD (sections 5.2.2, 6.9.6, and 8.4.2), short-term exposures to O_3 have been shown to impair host defense capabilities in both humans and experimental animals by depressing alveolar macrophage (AM) functions and by altering the mucociliary clearance of inhaled particles and microbes. Short-term O_3 exposures also interfere with the clearance process by accelerating clearance for low doses and slowing clearance for high doses. Animal toxicological studies have reported that acute O_3 exposures suppress alveolar phagocytosis and immune functions. Dysfunction of host defenses and subsequent increased susceptibility to bacterial lung infection in laboratory animals has been induced by short-term exposures to O_3 levels as low as 0.08 ppm (CD, p. 8-26).

Changes in antibacterial defenses are dependent on exposure regimens, species and strain of lab animals, species of bacteria, and age of the animals used. Acute O_3-induced suppression of alveolar phagocytosis and immune function in experimental animals appeared to be transient and attenuated with continuous or repeated exposures. Ozone exposure has also been shown to interfere with AM-mediated clearance in the respiratory region of the lung and with mucociliary clearance of the tracheobronchial airways. These interferences with clearance are dose dependent, with low doses accelerating clearance and high doses slowing the process (CD, p. 8-26).

A single controlled human exposure study (Devlin et al., 1991) reviewed in the 1996 CD reported that exposure to 0.08 to 0.10 ppm O_3 for 6.6 hrs (with moderate exercise) induced decrements in the ability of AMs to phagocytose microorganisms (CD, p. 8-26). Integrating the recent study results with evidence available in the 1996 CD, the CD concludes that available evidence indicates that short-term O_3 exposures have the potential to impair host defenses, primarily by interfering with AM function. Any impairment in AM function may lead to decreased clearance of microorganisms or nonviable particles. Compromised AM functions in

asthmatics may increase their susceptibility to other O_3 effects, the effects of particles, and respiratory infections (CD, p. 8-26).

3.3.1.1.5 Morphological Effects

The 1996 CD found that short-term O_3 exposures cause similar alterations in lung morphology in all laboratory animal species studied, including primates. Cells in the centriacinar region (CAR) of the lung (the segment between the last conducting airway and the gas exchange region) have been recognized as a primary target of O_3-induced damage (epithelial cell necrosis and remodeling of respiratory bronchioles), possibly because epithelium in this region receives the greatest dose of O_3 delivered to the lower respiratory tract. Following chronic O_3 exposure, structural changes have been observed in the CAR, the region typically affected in most chronic airway diseases of the human lung (CD, p. 8-24).

Ciliated cells in the nasal cavity and airways, as well as Type I cells in the gas-exchange region, are also identified as targets. While short-term O_3 exposures can cause epithelial cell profileration and fibrolitic changes in the CAR, these changes appear to be transient with recovery time after exposure, depending on species and O_3 dose. The potential impacts of repeated short-term and chronic morphological effects of O_3 exposure are discussed later in section 3.3.1.2.5.

Recent studies continue to show that short-term and sub-chronic exposures to O_3 cause similar alterations in lung structure in a variety of experimental animal species, at concentrations of 0.15 ppm in rats (12 hr/day for 6 weeks) and even lower concentrations in primates (8 hr/day for 90 days) (CD, section 5.2.4.). Recent work has shown that a topical anti-inflammatory corticosteroid can prevent these effects in nasal epithelia, while exposure to bacterial endotoxin can potentiate effects. Ozone-induced fibrotic changes in the CAR are maximal at 3 days of exposure and recover 3 days post-exposure with exposures of 0.2 ppm O_3 in rodents. One study has demonstrated variability of local O_3 dose and subsequent injury in the respiratory tract due to depletion of glutathione (GSH). The proximal respiratory bronchiole receives the most acute epithelial injury from exposures $\leq$ 1 ppm, while metabolic effects were greatest in the distal bronchioles and minor daughter airways (CD, p. 5-38).

Based on evidence from animal toxicological studies, short-term and sub-chronic exposures to O_3 can cause morphological changes in the respiratory systems, particularly in the CAR, of a number of laboratory animal species (CD, section 5.2.4).

3.3.1.1.6 Emergency Department Visits/Hospital Admissions for Respiratory Causes

The 1996 CD evaluated ED visits and hospital admissions as possible outcomes following exposure to O_3 (CD, section 7.3). The evidence was limited for ED visits, but results

of several studies generally indicated that short-term exposures to O_3 were associated with respiratory ED visits. The strongest and most consistent evidence, both below and above 0.12 ppm 1-hr max O_3, was found in the group of studies which investigated summertime daily hospital admissions for respiratory causes in different eastern North American cities. These studies were consistent in demonstrating that ambient O_3 levels were associated with increased hospital admissions and accounted for about one to three excess respiratory hospital admissions per million persons with each 100 ppb increase in 1-hr max O_3, with adjustment for possible confounding effects of temperature and copollutants. Overall, the 1996 CD concluded that there was strong evidence that ambient O_3 exposures can cause significant exacerbations of preexisting respiratory disease in the general public (CD, p. 7-66). Excess respiratory-related hospital admissions associated with O_3 exposures for the New York City area (based on Thurston et al., 1992) were included in the quantitative risk assessment in the prior review and are included in the current assessment along with estimates for respiratory-related hospital admissions in Cleveland, Detroit, and Los Angeles based on more recent studies (see Chapter 5). Significant uncertainties and the difficulty of obtaining reliable baseline incidence numbers resulted in ED visits not being used in the quantitative risk assessment conducted in the last O_3 NAAQS review.

In the past decade, a number of studies have examined the temporal pattern associations between O_3 exposures and ED visits for respiratory causes (CD, section 7.3.2). These studies are summarized in the CD (Table AX7-3, Chapter 7 Annex). Respiratory causes for ED visits include asthma, bronchitis, emphysema, pneumonia, and other upper and lower respiratory infections, such as influenza, but asthma visits typically dominate the daily incidence counts. Among studies with adequate controls for seasonal patterns, many reported at least one significant positive association involving O_3. These studies examined ED visits for total respiratory complaints (Delfino et al., 1997b, 1998b; Hernandez-Garduno et al., 1997; Ilabaca et al., 1999; Lin et al., 1999), asthma (Friedman et al., 2001; Jaffe et al., 2003; Stieb et al., 1996; Tenias et al., 1998; Tobias et al., 1999 ; Tolbert et al., 2000 ; Weisel et al., 2002), and COPD (Tenias et al., 2002).

Figure 7-8 (CD, p. 7-68) provides effect estimates for associations between ED visits for asthma and short-term O_3 exposures. In general, O_3 effect estimates from summer only analyses tended to be positive and larger compared to results from cool season or all year analyses (CD, p. 7-67). Several of the studies reported significant associations between O_3 concentrations and ED visits for respiratory causes. However, inconsistencies were observed which were at least partially attributable to differences in model specifications and analysis approach among various studies. For example, ambient O_3 concentrations, length of the study period, and statistical methods used to control confounding by seasonal patterns and copollutants appear to affect the observed O_3 effect on ED visits. Thus, the CD (p. 7-71) has concluded that stratified analyses by

season generally supported a positive association between O_3 concentrations and ED visits for asthma in the warm season.

Unscheduled hospital admissions occur in response to unanticipated disease exacerbations and are more likely to be affected by environmental factors, such as high O_3 levels. Thus, hospital admissions studies focus specifically on unscheduled admissions. Results of a fairly large number of these studies published during the past decade are summarized in Table AX7-4 (CD, Chapter 7 Annex). As a group, these hospital admissions studies tend to be larger geographically and temporally than the ED visit studies and provide results that are generally more consistent. The largest and most significant associations of respiratory hospital admissions with O_3 concentrations were observed using short lag periods, in particular for a 0-day lag (same day exposure) and a 1-day lag (previous day exposure). Most studies in the United States and Canada indicated positive, statistically significant associations between ambient O_3 concentrations and respiratory hospital admissions in the warm season, including studies with 98th percentile 8-hr maximum O_3 levels as low as about 50 ppb. However, not all studies found a statistically significant relationship with O_3, possibly because of very low ambient O_3 levels. Analyses for confounding using multipollutant regression models suggest that copollutants generally do not confound the association between O_3 and respiratory hospitalizations. Ozone effect estimates were robust to PM adjustment in all-year and warm-season only data.

Overall, the CD concludes that positive and robust associations were found between ambient O_3 concentrations and various respiratory disease hospitalization outcomes, when focusing particularly on results of warm-season analyses. Recent studies also generally supported a positive association between O_3 concentrations and ED visits for asthma during the warm season (CD, p. 7-175). These observations are strongly supported by the human clinical, animal toxicologic, and epidemiologic evidence for lung function decrements, increased respiratory symptoms, airway inflammation, and increased airway responsiveness. Taken together, the overall evidence supports a causal relationship between acute ambient O_3 exposures and increased respiratory morbidity outcomes resulting in increased ED visits and hospitalizations during the warm season (CD, p. 8-77).

3.3.1.1.7 Effects on Exercise Performance

The effects of O_3 exposure on exercise performance of healthy individuals have been investigated in a number of controlled exposure studies (CD, section 6.7). Several studies discussed in the 1996 CD reported that endurance exercise performance and VO_{2max} may be limited by acute exposure to O_3. Other studies found that significant reductions in maximal endurance exercise performance may occur in well-conditioned athletes while they perform CE (V_E > 80 L/min) for 1 hr at O_3 concentrations $\geq$ 0.18 ppm. There are no new studies available in

the CD. Thus, as in the 1996 CD, the CD concludes that reports from studies of O_3 exposure during high-intensity exercise indicate that breathing discomfort associated with maximal ventilation may be an important factor in limiting exercise performance in some, but not all, subjects (CD, p. 6-30).

3.3.1.2 Effects on the Respiratory System from Long-term Exposures

The 1996 CD concluded that there was insufficient evidence from the limited number of studies to determine whether long-term O_3 exposures resulted in chronic health effects at ambient levels observed in the U.S. However, the aggregate evidence suggested that O_3 exposure, along with other environmental factors, could be responsible for health effects in exposed populations (CD, section 7.5). Animal toxicological studies carried out in the 1980's and 1990's demonstrated that long-term exposures can result in a variety of morphological effects, including permanent changes in the small airways of the lungs, including remodeling of the distal airways and CAR and deposition of collagen, possibly representing fibrotic changes. These changes result from the damage and repair processes that occur with repeated exposure. Fibrotic changes were also found to persist after months of exposure providing a potential pathophysiologic basis for changes in airway function observed in children in some recent epidemiological studies. It appears that variable seasonal ambient patterns of exposure may be of greater concern than continuous daily exposures.

This section reviews studies published since 1996 in which health effects were assessed for O_3 exposures lasting from weeks to several years. Summaries of recent morphological effects studies of subchronic and chronic exposures are listed in Table AX5-10 (CD, Annex AX5). Summaries of recent morbidity effects epidemiological studies of long-term exposure are listed in Table AX7-6 (CD, Annex AX7).

3.3.1.2.1 Seasonal Ozone Effects on Lung Function

It is well documented in controlled human exposure and field studies that daily multi-hour exposures to O_3 produce transient declines in lung function; however, lung function effects of repeated exposures to O_3 over extended periods are not as well characterized. Several studies published since 1996 have investigated lung function changes over seasonal time periods (CD, section 7.5.3). One large, three-year study (Frischer et al., 1999) collected repeated lung function measurements in 1,150 young, Austrian school children in 9 communities and reported that there may be an association between developmental changes in lung function over the summer season and seasonal mean O_3 levels. Mean summertime 24-hr avg O_3 concentrations during the three summers was 34.8 ppb (SD 8.7). The number of days with half-hour maximum O_3 concentrations greater than 60 ppb ranged from 44 to 99 days across the 9 communities. Seasonal mean O_3 was associated with reduced lung function development. It was cautioned that

it was difficult to attribute the reported effects to O_3 alone independently of copollutants (CD, p. 7-113). A one-year extension of this study by Horak et al. (2002a,b) confirmed the results that seasonal mean O_3 levels may be related to a negative effect on increases in lung function in children. A study (Kopp et al., 2000) of 797 children in Austria and southwestern Germany reported smaller increases in lung function in children exposed to higher levels of ambient O_3 (mean O_3 concentration of 44 to 52 ppb) compared to children living in areas with lower ambient O_3 levels (25 to 33 ppb). Another Austrian study (Ihorst et al., 2000) of 2,153 young children found significantly lower FVC and FEV_1 increases associated with higher O_3 exposures in the summer but not in the winter. A pilot study (Kinney and Lippmann, 2000) of 72 young adult, military academy students provided results that are consistent with a seasonal decline in lung function that may be due, in part, to O_3 exposures. According to the CD (p. 7-114), these studies collectively indicate that seasonal O_3 exposure is associated with smaller growth-related increases in lung function in children than they would have experienced living in clean air and that there is some limited evidence that seasonal O_3 also may affect lung function in young adults, although uncertainty about the role of copollutants makes it difficult to attribute the effects to O_3 alone.

3.3.1.2.2 Reduced Baseline Lung Function and Respiratory Symptoms

Lung capacity grows during childhood and adolescence as body size increases, reaches a maximum during the twenties, and then begins to decline steadily and progressively with age. Long-term exposure to air pollution has long been thought to contribute to slower growth in lung capacity, diminished maximally attained capacity, and/or more rapid decline in lung capacity with age (CD, section 7.5.4). Toxicological findings evaluated in the 1996 CD demonstrated that repeated daily exposure of rats to an episodic profile of O_3 caused small, but significant, decrements in growth-related lung function that were consistent with early indicators of focal fibrogenesis in the proximal alveolar region, without overt fibrosis (CD, section 5.2.5.2). Because O_3 is a strong respiratory irritant and has been shown to cause inflammation and restructuring of the respiratory airways, it is plausible that long-term O_3 exposures might have a negative impact on baseline lung function, particularly during childhood when these exposures might have long-term risks. As noted in the current CD, however, no recent toxicological studies have been published on effects of chronic O_3 exposure.

Several epidemiological studies published since 1996 have examined the relationship between lung function development and long-term O_3 exposure. The most extensive and robust study of respiratory effects in relation to long-term air pollution exposures among children in the U.S. is the Children's Health Study carried out in 12 communities of southern California starting in 1993 (Avol et al., 2001; Gauderman et al., 2000, 2002, 2004a,b; Peters et al., 1999a,b). One

study (Peters et al., 1999a) examined the relationship between long-term O_3 exposures and self reports of respiratory symptoms and asthma in a cross sectional analysis and found a limited relationship between outcomes of current asthma, bronchitis, cough and wheeze and a 40 ppb increase in 1-hr max O_3 (CD, p. 7-115). Another analysis (Peters et al., 1999b) examined the relationship between lung function at baseline and levels of air pollution in the community and reported evidence that annual mean O_3 levels were associated with decreases in FVC, FEV_1, PEF and FEF_{25-75} (the latter two being statistically significant) among females but not males (CD, p. 7-116). In a separate study (Gauderman et al., 2000) of 4^{th}, 7^{th}, and 10^{th} grade students, a longitudinal analysis of lung function development over four years found no association with O_3 exposure. Subsequent studies by the same group (Gauderman et al., 2002, 2004a,b) led the authors to conclude that results provide little evidence that ambient O_3 at current levels is associated with chronic deficits in the rate of increase in growth-related lung function in children (CD, p. 7-116 to 7-118). Avol et al. (2001) examined children who had moved from participating communities in southern California to other states with improved air quality and found, with the exception of FEV_1, the O_3 effect estimates for all other spirometric parameters were negative, but the associations were not as strong as those observed for PM_{10} (CD, p. 7-116). Collectively, the results of these reports from the children's health cohorts provide little evidence for impact of long-term O_3 exposures on lung function development (CD, p. 7-122).

Evidence for a significant relationship between long-term O_3 exposures and decrements in maximally attained lung function was reported in a nationwide study of first year Yale students (CD, p. 7-120). Males had much larger effect estimates than females, which might reflect higher outdoor activity levels and correspondingly higher O_3 exposures during childhood. A similar study (Kunzli et al., 1997; Tager et al., 1998) of college freshmen at University of California at Berkeley also reported significant effects of long-term O_3 exposures on lung function (CD, p. 7-121). In a comparison of students whose city of origin was either Los Angeles or San Francisco, long-term O_3 exposures were associated with significant changes in mid- and end-expiratory flow measures, which could be considered early indicators for pathologic changes that might progress to COPD.

In summary, recent publications from the southern California children's cohort study provide no evidence for an association between long-term O_3 exposure and lung function development in children (CD, p. 7-118), while limited evidence available from studies of adults and college students suggest that long-term O_3 exposure may affect lung function or respiratory symptoms (CD, pp. 7-120, 7-121). Overall, the CD concluded that this body of evidence was inconclusive for effects of long-term O_3 exposure on respiratory symptoms or lung function (CD, p. 7-175).

3.3.1.2.3 Long-term O_3 Exposure and Respiratory Inflammation

As noted above in section 3.3.1.1.3 and in the CD (Chapter 6), chamber studies of exercising humans exposed to O_3 for 2 to 6.6 hrs have demonstrated inflammation in the lungs, including the alveolar region where gas exchange takes place. The potential long-term significance of short-term exposures to O_3 is that they can result in the release of reactive substances from inflammatory cells that can damage the sensitive cells lining the lungs. Over time repeated inflammation can lead to permanent lung damage and restructuring of the small airways and alveoli. Also, since inflammation is a hallmark characteristic of asthma, there is the possibility that O_3-induced inflammation may exacerbate existing asthma or contribute to the development of asthma in genetically predisposed individuals (CD, section 7.5.5).

For subchronic exposures of animals, permeability changes are transient (and species-dependent) and return to control levels even with continuing exposure. For long-term O_3 exposures, persistent O_3-induced inflammation plays an important role in alterations of lung structure and function. Significant remodeling of the epithelium and underlying connective tissues in distal airways have been reported in rats exposed to 0.25 ppm O_3 (12 hr/day for 6 weeks) and in monkeys exposed to 0.15 ppm O_3 (8 hr/day for 90 days)(CD, p. 8-23).

In one epidemiological field study (Kinney et al., 1996), BAL fluids were taken in the summer and winter from a group of joggers in New York and were compared for evidence of acute inflammation and of enhanced cell damage (CD, p. 7-122). The mean 1-hr max concentrations for a 3-month period were 58 ppb (max 110 ppb) in the summer and 32 ppb (max 64 ppb) in the winter. There was little evidence of an association between O_3 and acute inflammation in the summer BAL fluids compared to winter, but there was evidence of enhanced cell damage. This suggests that even though inflammation may diminish over the summer, cell damage may be continuing. A cross-sectional cohort study (Calderon-Garciduenas et al., 1995) conducted in Mexico City provides evidence of inflammation and genetic damage to cells in the nasal passages of children chronically exposed to O_3 and other air pollutants (CD, p. 7-123). In Mexico City, the 1-hr avg O_3 concentrations exceeded 120 ppb for 4.4 hr/day, on average. Significantly higher DNA damage was reported in children living in Mexico City compared to nonurban children and in older compared to younger children. Another marker of inflammation, urinary eosinophils, was analyzed in an Austrian school children study (Frischer et al., 2001), and it was reported that O_3 exposure (mean 30 day avg O_3 concentration before sample collection was 31.6 ppb) was significantly associated with eosinophil inflammation (CD, p. 7-122).

In assessing these studies, the CD (p. 7-123) concluded that specific attribution of these adverse respiratory and genotoxic effects to O_3 is difficult given the complex mixture in ambient air, although inflammatory changes like eosinophil levels observed in the Austrian study would be consistent with known effects of O_3.

3.3.1.2.4 Risk of Asthma Development

There have been a few studies investigating associations between long-term O_3 exposures and the onset of new cases of asthma (CD, section 7.5.6). The Adventist Health and Smog (AHSMOG) study cohort of 3,914 was drawn from nonsmoking, non-Hispanic white adult Seventh Day Adventists living in California (Greer et al., 1993; McDonnell et al., 1999). Subjects were surveyed in 1977, 1987, and 1992. During the ten-year follow-up in 1987, it was reported that the incidence of new asthma was 2.1% for males and 2.2% for females (Greer et al., 1993). A statistically significant relative risk of 3.12 (95% CI: 1.16, 5.85) per 10 ppb increase in annual mean O_3 was observed in males, compared to a nonsignificant relative risk of 0.94 (95% CI: 0.65, 1.34) in females. In the 15-year follow-up in 1992, it was reported that 3.2% of eligible males and 4.3% of eligible females had developed adult asthma (McDonnell et al., 1999). For males, the relative risk of developing asthma was 2.27 (95% CI: 1.03, 4.87) per 30 ppb increase in 8-hr average O_3, but there was no evidence of an association in females. The lack of an association in females does not necessarily mean there is no effect but may be due to differences in time-activity patterns in males and females, which could lead to greater misclassification of exposure in females. Consistency of results in the two studies with different follow-up times provides supportive evidence of an association between long-term O_3 exposure and asthma incidence in adult males; however, representativeness of this cohort to the general U.S. population may be limited (CD, p. 7-125).

In a similar study (McConnell et al., 2002) of incident asthma among children (ages 9 to 16 at enrollment), annual surveys of 3,535 children initially without asthma were used to identify new-onset asthma cases as part of the Children's Health Study. Six high-O_3 (75.4 ppb mean 1-hr max over four years) and six low-O_3 (50.1 ppb, mean 1-hr max) communities were identified where the children resided. There were 265 children who reported new-onset asthma during the follow-up period. Although asthma risk was no higher for all residents of the six high-O_3 communities versus the six low-O_3 communities, asthma risk was 3.3 times greater for children who played three or more sports as compared with children who played no sports within the high-O_3 communities. This association was absent in the communities with lower O_3 concentrations. No other pollutants were found to be associated with new-onset asthma (CD, p. 7-125).

Playing sports may result in extended outdoor activity and exposure occurring during periods when O_3 levels are higher. The sports activities would cause an increased ventilation rate, thus resulting in increased O_3 dose. It should be noted, however, that the results of the Children's Health Study (McConnell et al., 2002) were based on a small number (20 in high-O_3 areas and 9 in low- O_3 areas) of new-onset asthma cases among children who played three or

more sports (CD, p. 7-125). Future replication of these findings in other cohorts would help determine whether a causal interpretation is appropriate.

3.3.1.2.5 Morphological Effects

In animal toxicology studies, the progression of morphological effects reported during and after a chronic exposure in the range of 0.5 to 1.0 ppm O_3 is complex, with inflammation peaking over the first few days of exposure, then dropping, then plateauing, and finally, largely disappearing (CD, section 5.2.4.4). By contrast, fibrotic changes in the tissue increase very slowly over months of exposure, and, after exposure ceases, the changes sometimes persist or increase. Epithelial hyperplasia peaks soon after the inflammatory response but is usually maintained in both the nose and lungs with continuous exposure. Epithelial hyperplasia/metaplasia also does not return to pre-exposure levels after the end of exposure. Patterns of exposure in this same concentration range determine effects, with 18 months of daily exposure, causing less morphologic damage than exposures on alternating months. This is important as environmental O_3 exposure is typically seasonal. Long-term studies of Plopper and colleagues (Evans et al., 2003; Schelegle et al., 2003; Chen et al., 2003; Plopper and Fanucchi, 2000) investigated infant rhesus monkeys exposed to simulated, seasonal O_3 (0.5 ppm, 8 hrs/day for 5 days, every 14 days for 11 episodes) and demonstrated: 1) remodeling in the distal airways, 2) abnormalities in tracheal basement membrane; 3) eosinophil accumulation in conducting airways; and 4) decrements in airway innervation (CD, p. 5-45). As with other effects, these findings advance earlier information regarding possible injury-repair processes occurring with long-term O_3 exposures suggesting that these processes are only partially reversible and may progress following cessation of O_3 exposure. Further, these processes may lead to nonreversible structural damage to lung tissue; however, there is still too much uncertainty to quantitatively extrapolate these levels to human effect levels at this time (CD, p. 8-25).

3.3.1.2.6 Summary

In the past decade, important new longitudinal studies have examined the effect of chronic O_3 exposure on respiratory health outcomes. Evidence from recent long-term morbidity studies have suggested in some cases that chronic exposure to O_3 may be associated with seasonal declines in lung function or reduced lung function development, increases in inflammation, and development of asthma in children and adults. Seasonal decrements or smaller increases in lung function measures have been reported in several studies; however, it remains uncertain to what extent these changes are transient. While there is supportive evidence from animal studies involving chronic exposures, large uncertainties still remain as to whether current ambient levels and exposure patterns might cause these same effects in human

populations. The CD also concludes that epidemiological studies of new asthma development and longer-term lung function declines remain inconclusive at present (CD, p. 7-134).

3.3.1.3 Effects on the Cardiovascular System

At the time of the 1997 review, the possibility of O_3-induced cardiovascular effects was a largely unrecognized issue. Since then, a very limited body of evidence from animal, controlled human exposure and epidemiologic studies has emerged that provides some potential plausible mechanisms for how O_3 exposures might exert cardiovascular system effects, however much needs to be done to substantiate these effects. Possible mechanisms may involve O_3-induced secretions of vasoconstrictive substances and/or effects on neuronal reflexes that may result in increased arterial blood pressure and/or altered electrophysiologic control of heart rate or rhythm. Some animal toxicology studies have shown O_3-induced decreases in heart rate, mean arterial pressure, and core temperature. One controlled human exposure study that evaluated effects of O_3 exposure on cardiovascular health outcomes found no significant O_3-induced differences in ECG or blood pressure in healthy or hypertensive subjects but did observe a significant O_3-induced increase the alveolar-to-arterial PO_2 gradient and heart rate in both groups resulting in an overall increase in myocardial work and impairment in pulmonary gas exchange (Gong et al., 1998). In another controlled human exposure study, inhalation of a mixture of $PM_{2.5}$ and O_3 by healthy subjects increased brachial artery tone and reactivity (Brook et al., 2002).

The evidence from a few animal studies also includes potential direct effects such as O_3-induced release from lung epithelial cells of platelet activating factor (PAF) that may contribute to blood clot formation that would increase the risk of serious cardiovascular outcomes (e.g., heart attack, stroke, mortality). Also, interactions of O_3 with surfactant components in epithelial lining fluid of the lung may result in production of oxysterols and reactive oxygen species that may exhibit PAF-like activity contributing to clotting and also may exert cytotoxic effects on lung and heart muscle cells.

Epidemiologic panel and field studies that examined associations between O_3 and various cardiac physiologic endpoints have yielded limited evidence suggestive of a potential association between acute O_3 exposure and altered heart rate variability, ventricular arrhythmias, and incidence of heart attacks. A number of epidemiological studies have also reported associations between short-term exposures and hospitalization for cardiovascular diseases. As shown in Figure 7-13 of the CD, many of the studies reported negative or inconsistent associations. Some other studies, especially those that examined the relationship when O_3 exposures were higher, have found robust positive associations between O_3 and cardiovascular hospital admissions (CD, p. 7-82). For example, one study reported a positive association between O_3 and cardiovascular

hospital admissions in Toronto, Canada in a summer-only analysis (mean 1-hr max O_3 of 41.2 ppb) (Burnett et al., 1997b). The results were robust to adjustment for various PM indices, whereas the PM effects diminished when adjusting for gaseous pollutants. Other studies stratified their analysis by temperature, i.e., by warms days (≥ 20°C) versus cool days (< 20°C). Several analyses using warms days consistently produced positive associations.

The epidemiologic evidence for cardiovascular morbidity is much more mixed than for respiratory morbidity, with only one of several U.S./Canadian studies showing statistically significant positive associations of cardiovascular hospitalizations with warm-season O_3 concentrations. Most of the available European and Australian studies, all of which conducted all-year O_3 analyses, did not find an association between short-term O_3 concentrations and cardiovascular hospitalizations. Overall, the currently available evidence is inconclusive regarding an association between cardiovascular hospital admissions and ambient O_3 exposure (CD, p. 7-83).

Based on the evidence from animal toxicology, human controlled exposure, and epidemiologic studies, the CD concludes that this generally limited body of evidence is highly suggestive that O_3 can directly and/or indirectly contribute to cardiovascular-related morbidity, but that much needs to be done to more fully substantiate links between ambient O_3 exposures and adverse cardiovascular outcomes (CD, p. 8-77).

3.3.2 Premature Mortality

There were only a limited number of studies which examined the relationship between O_3 and mortality available for review in the 1996 CD. Some studies suggested that mortality was associated with short-term exposure to O_3, but conclusions could not be drawn regarding such associations (CD, p. 7-84). Numerous recent studies have provided new and more substantial evidence supporting such an association, as discussed below in section 3.3.2.1.

At the time of the last review, little epidemiological evidence was available on potential associations between long-term exposure to O_3 and mortality. Some recent studies have evaluated this relationship and provide limited, if any, evidence for an association between chronic O_3 exposure and mortality, as described in section 3.3.2.2.

3.3.2.1 Mortality and Short-term O_3 Exposure

The 1996 CD concluded that an association between daily mortality and O_3 concentration for areas with high O_3 levels (e.g., Los Angeles) was suggested. However, due to a very limited number of studies available at that time, there was insufficient evidence to conclude that the observed association was likely causal, and thus the possibility that O_3 exposure may be associated with mortality was not relied upon in the 1997 decision on the O_3 primary standard.

The 2006 CD includes results from numerous epidemiological analyses of the relationship between O_3 and mortality. Key findings are available from multi-city time-series studies that report associations between O_3 and mortality. These studies include analyses using data from 90 U.S. cities in the National Mortality, Morbidity and Air Pollution (NMMAPS) study (Dominici et al., 2003) and from 95 U.S. communities in an extension to the NMMAPS analyses (Bell et al., 2004). The analyses conducted by Huang et al. (2005) used a subset of 19 U.S. cities and focused primarily on cause-specific mortality associations during the warm season. An additional study (Schwartz, 2005) used case-crossover design and data from 14 U.S. cities to further investigate the influence of adjustment for weather variables in the O_3-mortality relationship (CD, p. 8-38). Finally, results are available from a European study, Air Pollution and Health: a European Approach (APHEA), using data from 23 cities (Gryparis et al., 2004) and 4 cities (Toulomi et al., 1997) (CD, p. 7-93).

The original 90-city NMMAPS analysis, with data from 1987 to 1994, was primarily focused on investigating effects of PM_{10} on mortality. A significant association was reported between mortality and 24-hr average O_3 concentrations during the warm season, but the association was not significant in analyses for the full year (Samet et al., 2000) (CD, Figure 7-21; p. 7-98). This is because the estimate using all available data was about half that for the summer-only data at a lag of 1-day. The extended NMMAPS analysis included data from 95 U.S. cities and included an additional 6 years of data, from 1987-2000 (Bell et al., 2004), and significant associations were reported between O_3 and mortality. The effect estimate for increased mortality was 0.5% per 24-hr average O_3 measured on the same day (20 ppb change; 95% PI: 0.24, 0.78), and 1.04% per 24-hr average O_3 in a 7-day distributed lag model (20 ppb change; 95% PI: 0.54, 1.55) (CD, p. 7-88). In analyses using only data from the warm season, the results were not significantly different from the full-year results; the effect estimate for increased mortality was 0.44% per 24-hr average O_3 measured on the same day (20 ppb change; 95% PI: 0.14, 0.74), and 0.78% per 24-hr average O_3 in a 7-day distributed lag model (20 ppb change; 95% PI: 0.26, 1.30). The authors also report that O_3-mortality associations were robust to adjustment for PM (CD, p. 7-100).

Using a subset of the NMMAPS data set, Huang et al. (2005) focused on associations between cardiopulmonary mortality and O_3 exposure (24-hr avg) during the summer season only. The authors report a 1.47% increase per 20 ppb change in O_3 concentration measured on the same day (95% PI: 0.54, 2.39) and a 2.52% increase per 20 ppb change in O_3 concentration using a 7-day distributed lag model (95% PI: 0.94, 4.10)(CD, p. 7-92). These findings suggest that the effect of O_3 on mortality is immediate but also persists for several days.

As discussed below in section 3.4, confounding by weather, especially temperature, is complicated by the fact that higher temperatures are associated with the increased photochemical

activities that are important for O_3 formation. Using a case-crossover study design, Schwartz (2005) assessed associations between daily maximum concentrations and mortality, matching case and control periods by temperature, and using data only from the warm season. The reported effect estimate of 0.92% change in mortality per 40 ppb O_3 (1-hr max, 95% PI: 0.06, 1.80) was similar to time-series analysis results with adjustment for temperature (0.76% per 40 ppb O_3, 95% PI, 0.13, 1.40), suggesting that associations between O_3 and mortality were robust to the different adjustment methods for temperature (CD, p. 7-93).

An initial publication from APHEA, a European multi-city study, reported statistically significant associations between daily maximum O_3 concentrations and mortality, with an effect estimate of a 4.5% increase in mortality per 40 ppb O_3 (95% CI: 1.6, 7.7) in four cities (Toulomi et al., 1997). An extended analysis was done using data from 23 cities throughout Europe (Gryparis et al., 2004). In this report, a positive but not statistically significant association was found between mortality and 1-hr daily maximum O_3 in a full year analysis (CD, p. 7-93). Gryparis et al. (2004) noted that there was a considerable seasonal difference in the O_3 effect on mortality; thus, the small effect for the all-year data might be attributable to inadequate adjustment for confounding by seasonality. Focusing on analyses using summer measurements, the authors report statistically significant associations with total mortality [1.8% increase per 30 ppb 8-hr O_3 (95% CI: 0.8, 2.9)], cardiovascular mortality [2.7% increase per 30 ppb 8-hr O_3 (95% CI: 1.2, 4.3)] and with respiratory mortality (6.8% increase per 30 ppb 8-hr O_3, 95% CI: 4.5, 9.2) (CD, p. 7-93, 7-99).

Two of the recent multi-city mortality studies (Bell et al., 2004; Gryparis et al., 2004) have also reported associations for multiple averaging times (CD, p. 8-38). Bell and colleagues (2004) reported associations between mortality and 1-hr daily max, 8-hr daily max and 24-hr avg O_3 concentrations. Effect estimates for associations with 1-hr O_3 was slightly larger than that reported for 8-hr O_3 concentrations, and both were slightly larger than the association with 24-hr avg O_3, but the effect estimates did not differ statistically. The APHEA study (Gryparis et al., 2004) also reported effect estimates that were slightly larger with 1-hr than with 8-hr O_3 concentrations, but not significantly so.

Numerous single-city analyses have also reported associations between mortality and short-term O_3 exposure, especially for those analyses using warm season data. As shown in Figure 7-21 of the CD, the results of recent publications show a pattern of positive, often statistically significant associations between short-term O_3 exposure and mortality during the warm season (CD, p. 7-97). For example, statistically significant associations were reported in southern California (Ostro, 1995), Philadelphia (Moolgavkar et al., 1995), Dallas (Gamble et al., 1998), and Vancouver (Vedal et al., 2003), as well as numerous studies conducted in other countries. However, no evidence of an association was seen in a study conducted in Pittsburgh

(Chock et al., 2000). In considering results from year-round analyses, there remains a pattern of positive results but the findings are less consistent. For example, statistically significant associations were reported in Philadelphia (Moolgavkar et al., 1995) and Dallas (Gamble et al., 1998), while positive but not statistically significant associations were reported in Detroit (Lippmann et al., 2000, reanalyzed in Ito, 2003), San Jose (Fairley, 1999, reanalyzed Fairley, 2003), and Atlanta (Klemm et al., 2004). No evidence for associations was reported in Los Angeles (Kinney et al., 1995), Coachella Valley (Ostro et al., 2003), and St. Louis and Eastern Tennessee (Dockery et al., 1992). In most single-city analyses, effect estimates were not substantially changed with adjustment for PM (CD Figure 7-22, p. 7-101).

In addition, several meta-analyses have been conducted on the relationship between O_3 and mortality. As described in section 7.4.4 of the CD, these analyses reported fairly consistent and positive combined effect estimates ranging from 1.5 to 2.5% increase in mortality for a standardized change in O_3 (CD, Figure 7-20, p. 7-95). Three recent meta-analyses evaluated potential sources of heterogeneity in O_3-mortality associations (Bell et al., 2005; Ito et al., 2005; Levy et al., 2005). The CD (p. 7-96) observes common findings across all three analyses, in that all reported that effect estimates were larger in warm season analyses, reanalysis of results using default GAM criteria did not change the effect estimates, and there was no strong evidence of confounding by PM (CD, p. 7-97). Bell et al. (2005) and Ito et al. (2005) both provided suggestive evidence of publication bias, but O_3-mortality associations remained after accounting for that potential bias. The CD (7-97) concludes that the "positive O_3 effects estimates, along with the sensitivity analyses in these three meta-analyses, provide evidence of a robust association between ambient O_3 and mortality."

Most of the single-pollutant model estimates from single-city studies range from 0.5 to 5% excess deaths per standardized increments. Corresponding summary estimates in large U.S. multi-city studies ranged between 0.5 to 1% with some studies noting heterogeneity across cities and studies (CD, p. 7-110).

In the CD (p. 7-101), Figure 7-22 shows the O_3 risk estimates with and without adjustment for PM indices using all-year data in studies that conducted two-pollutant analyses. Approximately half of the O_3 risk estimates increased slightly, whereas the other half decreased slightly with the inclusion of PM in the models. In general, the O_3-mortality risk estimates were robust to adjustment for PM in the models, with the exception of Los Angeles, CA data with PM_{10} (Kinney et al., 1995) and Mexico City data with TSP (Borja-Aburto et al., 1997). The U.S. 95 communities study (Bell et al., 2004) examined the sensitivity of acute O_3-mortality effects to potential confounding by PM_{10} (CD, 7-100). Restricting analysis to days when both O_3 and PM_{10} data were available, the community-specific O_3-mortality effect estimates as well as the national average results indicated that O_3 was robust to adjustment for PM_{10} (Bell et al., 2004).

Several O_3-mortality studies examined the effect of confounding by PM indices in different seasons (CD, p. 7-102, Figure 7-23). In analyses using all-year data and warm-season only data, O_3 effect estimates were once again fairly robust to adjustment for PM indices, with values showing both slight increases and decreases with the inclusion of PM in the model. In the analyses using cool season data only, the O_3 effect estimates all increased slightly with the adjustment of PM indices, although none reached statistical significance.

The three recent meta-analyses (Bell et al., 2005; Ito et al., 2005; Levy et al., 2005) all examined the influence of PM on O_3 risk estimates. No substantial influence was observed in any of these studies. In the analysis by Bell et al. (2005), the combined estimate without PM adjustment was 1.7% (95% PI: 1.10, 2.37) from 41 estimates, and the combined estimate with PM adjustment was 1.95% (95% PI: 1.06, 4.00) from 11 estimates per 20 ppb increase in 24-hr avg O_3. In the meta-analysis of 15 cities (Ito et al., 2005), the combined estimate was 1.6% (95% PI: 1.1, 2.2) and 1.5% (95% PI: 0.8, 2.2) per 20 ppb in 24-hr avg O_3 without and with PM adjustment, respectively (CD, p. 7-103). The additional time-series analysis of six cities by Ito et al. (2005) found that the influence of PM by season varied across alternative weather models but was never substantial. Levy et al. (2005) examined the regression relationships between O_3 and PM indices (PM_{10} and $PM_{2.5}$) with O_3-mortality effect estimates for all year and by season. Positive slopes, which might indicate potential confounding, were observed for $PM_{2.5}$ on O_3 effect estimates in the summer and all-year periods, but the relationships were weak. The effect of one causal variable (i.e., O_3) is expected to be overestimated when a second causal variable (e.g., PM) is excluded from the analysis, if the two variables are positively correlated and act in the same direction. However, the results from these meta-analyses, as well as several single- and multiple-city studies, indicate that copollutants generally do not appear to substantially confound the association between O_3 and mortality (CD, p. 7-103).

Finally, from those studies that included assessment of associations with specific causes of death, it appears that effect estimates for associations with cardiovascular mortality are larger than those for total mortality; effect estimates for respiratory mortality are less consistent in size, possibly due to reduced statistical power in this subcategory of mortality (CD, p. 7-108). The U.S. 95 communities study (1987-2000) analyzed O_3 effect estimates from cardiovascular and respiratory mortality. The analysis by Bell et al. (2005) used all available data, which included all-year data from 55 communities and warm-season only data from 40 communities. The national average estimate from the constrained distributed lag model was slightly greater for cardiopulmonary deaths than deaths from all causes, with an excess risk of 1.28% (95% PI: 0.62, 1.97) compared to 1.04% (95% PI: 0.54, 1.55) per 20 ppb increase in 24-hr avg O_3 in the preceding week.

One recent multi-city study (Bell et al., 2006) examined the shape of the concentration-response function for the O_3-mortality relationship in 98 U.S. urban communities for the period 1987 to 2000 specifically to evaluate whether a "safe" threshold level exists. Results from various analytic methods all indicated that any threshold would exist at very low concentrations, far below the level of the current O_3 NAAQS and other, lower international O_3 standards,[3] and nearing background levels. Notably, in a subset analysis using only days that were below the level of the current O_3 NAAQS, the O_3-mortality association remained statistically significant with only a small change in the size of the effect estimate. Further, in a subset analysis based on 24-hr average O_3 concentrations, the effect estimates declined and lost statistical significance only when the maximum daily average concentration included was $\leq$ 10 ppb (Bell et al., 2006, p. 14 and Figure 2), which corresponds to daily maximum 8-hr average concentrations in U.S. cities that are within the range of background concentrations. The authors conclude that "interventions to further reduce ozone pollution would benefit public health, even in regions that meet current regulatory standards and guidelines" (Bell et al., 2006, p. 3).

A related study (Huang et al., 2005) examined O_3 effects on cardiopulmonary mortality during the summers (June to September) of 1987 to 1994 in 19 large U.S. cities from the NMMAPS database. Figure 7-24 in the CD (p. 7-104), presents the Bayesian city-specific and overall average O_3 effect estimates for cardiopulmonary mortality per 20 ppb increase in 24-hr avg O_3 from a constrained 7-day distributed lag model. The O_3 effect estimate was 2.52% (95% PI: 0.94, 4.10) excess risk in cardiopulmonary mortality per 20 ppb increase in 24-hr avg O_3 in the preceding week for the combined analysis of all cities. For analyses of summer data, confounding of the O_3 effect by PM is of concern as daily variations in O_3 may be positively correlated to PM during the summer months. Huang et al. (2005) observed that when PM_{10} was included in the model, the O_3 effect estimate, on average, remained positive and significant. As PM_{10} measurements were available only every 1 to 6 days, only single-day lags were examined. At a 0-day lag, O_3 was associated with a 1.47% (95% PI: 0.54, 2.39) excess risk versus a 1.49% (95% PI: 0.66, 3.47) excess risk in cardiopulmonary mortality in the O_3-only model and after adjustment for PM_{10}, respectively. The slight sensitivity of the O_3 health effects to the inclusion of PM_{10} in the model may indicate a true confounding effect. However, as only the days with PM_{10} data available were included in the analysis, the lack of significance is likely attributable to higher statistical uncertainty due to the reduced data availability (CD, p. 7-105).

Figure 7-25 in the CD (p., 7-106), presents effect estimates for associations between O_3 and cardiovascular mortality for all-year and warm-season analyses. All studies, with the

[3] Other international 8-hr O_3 standards considered by Bell et al. (2006, Table 1) include the California standard of 70 ppb, the Canadian standard of 65 ppb, and the World Health Organization guideline and European Commission target value of approximately 61 ppb.

exception of Ponka et al. (1998), showed positive associations between O_3 and cardiovascular mortality (CD, p. 7-105). As with all-cause mortality, there appears to be heterogeneity in the effect estimates across studies. The cardiovascular mortality estimate from one meta-analysis (Bell et al., 2005) appears to be close to the mode of the effect estimates from the various studies, as shown in Figure 7-25, in the CD (p. 7-106). This is expected, given that many of these studies were also included in the meta-analysis. This study observed that the posterior mean estimate for cardiovascular causes (2.23% excess risk per 20 ppb increase in 24-hr avg O_3 from 25 estimates) was slightly larger than that for total mortality (1.75% excess risk from 41 estimates). However, since cardiovascular deaths account for the largest fraction (over 40%) of total deaths, it is not surprising that the risk estimates for cardiovascular mortality are somewhat similar to those from all-cause mortality. Overall, the cardiovascular mortality risk estimates in the current literature show consistently positive associations with some heterogeneity (most estimates fall within the range of 1 to 8% per 40 ppb increase in 1-hr avg O_3 (CD, p. 7-107).

Several studies observed that the risk estimates for the respiratory category were larger than the cardiovascular and total nonaccidental categories (Anderson et al., 1996; Gouveia and Fletcher, 2000; Gryparis et al., 2004; Zmirou et al., 1998). The apparent inconsistencies across studies may be due in part to the differences in model specifications, but they may also reflect the lower statistical power associated with the smaller daily counts of the respiratory category (usually accounting for less than 10% of total deaths) compared to the larger daily counts for the cardiovascular category (approximately 40 to 50% of total deaths). Thus, an examination of the differences in risk estimates across specific causes requires a large population and/or a long period of data collection.

In summary, several single-city studies observed positive associations of ambient O_3 concentrations with total nonaccidental and cardiopulmonary mortality. The CD finds that the results from U.S. multi-city time-series studies provide the strongest evidence to date for O_3 effects on acute mortality. Recent meta-analyses also indicate positive risk estimates that are unlikely to be confounded by PM; however, future work is needed to better understand the influence of model specifications on the risk coefficient (CD, p. 7-175). A meta-analysis that examined specific causes of mortality found that the cardiovascular mortality risk estimates were higher than those for total mortality. For cardiovascular mortality, the CD (Figure 7-25, p. 7-106) suggests that effect estimates are consistently positive and more likely to be larger and statistically significant in warm season analyses. The findings regarding the effect size for respiratory mortality have been less consistent, possibly because of lower statistical power in this subcategory of mortality. The CD (p. 8-78) concludes that these findings are highly suggestive that short-term O_3 exposure directly or indirectly contribute to non-accidental and cardiopulmonary-related mortality, but additional research is needed to more fully establish underlying mechanisms by which such effects occur.

3.3.2.2 Mortality and Long-term O_3 Exposure

Little evidence was available in the last review on the potential for associations between mortality and long-term exposure to O_3. In the Harvard Six City prospective cohort analysis, the authors report that mortality was not associated with long-term exposure to O_3 (Dockery et al., 1993). The authors note that the range of O_3 concentrations across the six cities was small (19.7 to 28.0 ppb in average 24-hr concentrations over the 7-year study period), which may have limited the power of the study to detect associations between mortality and O_3 levels (CD, p. 7-127).

As discussed in section 7.5.8 of the CD, in this review there are results available from three prospective cohort studies: the American Cancer Society (ACS) study, the Adventist Health and Smog (AHSMOG) study, and the U.S. Veterans Cohort study. In addition, a major reanalysis report includes evaluation of data from the Harvard Six City cohort study (Krewski et al., 2000). This reanalysis also includes additional evaluation of data from the initial ACS cohort study report that had only reported results of associations between mortality and long-term exposure to fine particles and sulfates (Pope et al., 1995).[4]

In this reanalysis of data from the previous Harvard Six City prospective cohort study, the investigators replicated and validated the findings of the original studies, and the report included additional quantitative results beyond those available in the original report (Krewski et al., 2000). In the reanalysis of data from the Harvard Six Cities study, the effect estimate for the association between long-term O_3 concentrations (8.3 ppb between the highest and lowest concentrations in the cities) and mortality was negative and nearly statistically significant (relative risk = 0.87, 95% CI: 0.76, 1.00).

The ACS study is based on health data from a large prospective cohort of approximately 500,000 adults and air quality data from about 150 U.S. cities. The initial report (Pope et al., 1995) focused on associations with fine particles and sulfates, for which significant associations had been reported in the earlier Harvard Six Cities study (Dockery et al., 1993). As part of the major reanalysis of these data, results for associations with other air pollutants were also reported, and the authors report that no significant associations were found between O_3 and all cause mortality (95% CI: 0.96-1.07). A significant association was reported for cardiopulmonary mortality (relative risk=1.08, 95% CI: 1.01, 1.16) (Krewski et al., 2000, p. 174). For some specifications of O_3 exposure in the ACS study, there was an effect in the warm quarter, as there was in the reanalysis of the Harvard Six Cities study.

[4] This reanalysis report and the original prospective cohort study findings are discussed in more detail in section 8.2.3 in *Air Quality Criteria for Particulate Matter* (EPA, 2004).

The ACS II study (Pope et al., 2002) reported results of associations with an extended data base; the mortality records for the cohort had been updated to include 16 years of follow-up (compared with 8 years in the first report) and more recent air quality data were included in the analyses. Results are presented for full-year and summer season analyses, and show no evidence for a significant association between long-term exposure to O_3 and mortality. As shown in Figure 7-27 of the CD, the effect estimates are not statistically significant for associations between long-term O_3 exposure and all-cause, cardiopulmonary, and lung cancer mortality (CD, p. 7-128) in all year analyses. However, in the summer season, marginally significant associations were observed for cardiopulmonary mortality.

The Adventist Health and Smog (AHSMOG) cohort includes about 6,000 adults living in California. In two studies from this cohort, a significant association has been reported between long-term O_3 exposure and increased risk of lung cancer mortality among males only (Beeson et al., 1998; Abbey et al., 1999). No significant associations were reported between long-term O_3 exposure and mortality from all causes or cardiopulmonary causes. Due to the small numbers of lung cancer deaths (12 for males, 18 for females) and the precision of the effect estimate (i.e., the wide confidence intervals), the CD raised concerns about the plausibility of the reported association with lung cancer (CD, p. 7-130).

The U.S. Veterans Cohort study (Lipfert et al., 2000b, 2003) of approximately 50,000 middle-aged males diagnosed with hypertension, reported some positive associations between mortality and peak O_3 exposures (95^{th} percentile level for several years of data). The study included numerous analyses using subsets of exposure and mortality follow-up periods which spanned the years 1960 to 1996. In the results of analyses using deaths and O_3 exposure estimates concurrently across the study period, there were positive, statistically significant associations between peak O_3 and mortality, with a 9.4% excess risk (95% CI: 0.4, 18.4) per mean 95% percentile O_3 (CD, p. 7-129).

Thus, the results from all-year analyses in the Harvard Six Cities and ACS cohorts provide no evidence for associations between long-term O_3 exposure and mortality, though the warm-season results in the reanalysis of the ACS cohort study suggest a potential association. Imprecise and inconclusive associations were reported in analyses for the AHSMOG cohort study. Significant associations between long-term O_3 exposure and mortality were only reported for the Veterans cohort study; however, this study used an indicator of peak O_3 concentrations and the cohort is also a rather specific subgroup of the U.S. population. Overall, the CD concludes that consistent associations have not been reported between long-term O_3 exposure and all-cause, cardiopulmonary or lung cancer mortality (CD, p. 7-130).

3.3.3 Ozone Effects on UV-B Flux

The CD (Chapter 10) provides a thorough analysis of the current understanding of the relationship between reducing tropospheric O_3 concentrations and the potential impact these reductions might have on increasing UV-B surface fluxes and indirectly contributing to increased UV-B related health effects. It is clear that there are many factors that influence UV-B radiation penetration to the earth's surface, including cloud cover, surface albedo, PM concentration and composition, and gas phase pollution. A risk assessment of UV-B related health effects would need to take into account human habits, such as outdoor activities, dress and skin care. However, little is known about the impact of these factors on individual exposure to UV-B, and detailed information does not exist regarding type (e.g., peak or cumulative) and time period (e.g., childhood, lifetime, current) of exposure, wavelength dependency of biological responses, and interindividual variability in UV-B resistance. In fact there have been recent reports indicating the necessity of UV-B in producing vitamin D, suggesting that increased risks of human disease due to slight excess UV-B exposure may be offset by the benefits of enhanced vitamin D production. However, as with other impacts of UV-B on human health, this beneficial effect of UV-B radiation has not been studied in sufficient detail to allow for a credible health benefits or risk assessment. The CD (p. 10-38) concluded that the effects of changes in surface-level O_3 concentrations on UV-induced health effects cannot be critically assessed given the significant uncertainties summarized above.

3.3.4 Summary

The CD (Chapters 4-8) summarizes and assesses substantial new evidence which builds upon what was previously known about the health effects of O_3. The new information supports previous findings that short-term O_3 is associated with lung function decrements and respiratory symptoms, as well as numerous more subtle effects on the respiratory system such as morphological changes and altered host defense mechanisms. Short-term O_3 exposure has also been associated with hospital admissions for respiratory causes in numerous new studies that further confirm the findings evaluated in the 1996 CD. The CD reports that warm-season studies show evidence for positive and robust associations between ambient O_3 concentrations and respiratory hospital admissions, asthma ED visits, and respiratory symptoms and lung function effects in asthmatic children (CD, p. 7-175).

Some new studies have suggested associations between increased incidence of asthma or reduced lung function and long-term exposure to elevated ambient O_3 levels. The findings of this small group of studies are inconsistent, however, and the CD concludes that the evidence for this group of associations is inconclusive (CD, p. 7-175).

A new body of studies has suggested associations between short-term O_3 exposure and effects on the cardiovascular system, including changes in heart rate variability, cardiac arrhythmia, incidence of MI and hospitalization for cardiovascular diseases. The CD finds this body of evidence to be limited but supportive of potential effects of O_3 on the cardiovascular system (CD, p. 8-77).

A major area where new information presented in the CD has significantly expanded our knowledge on health effects is evidence of an elevated risk of mortality associated with acute exposure to O_3, especially in the summer or warm season when O_3 levels are typically high. Results from recent large U.S. multicity time-series studies and meta-analyses provide the strongest evidence for associations between short-term O_3 exposure and mortality (CD, p. 7-175). The risk estimates shown are consistent across studies and robust to control for potential confounders. This overall body of evidence is highly suggestive that O_3 directly or indirectly contributes to nonaccidental and cardiopulmonary-related mortality, but additional research is needed to more fully establish underlying mechanisms by which such effects occur (CD, p. 8-78).

3.4 ASSESSMENT OF EVIDENCE FROM EPIDEMIOLOGICAL STUDIES

In Chapter 8, the CD assesses the new health evidence, integrating findings from experimental (e.g., toxicological, dosimetric and controlled human exposure) and epidemiological studies, to make judgments about the extent to which causal inferences can be made about observed associations between health endpoints and exposure to O_3. Section 8.4.4.3 of the CD indicates that *strength* of the O_3 effects (including the magnitude and precision of reported O_3 effect estimates and their statistical significance), *robustness* of epidemiological associations (i.e., stability in the effect estimates after considering a number of factors), and *consistency* of effects associations (looking across results of multiple- and single-city studies conducted by different investigators in different places and times) are all important in forming judgments as to the likely causal significance of observed associations (CD, p. 8-40).

In evaluating the evidence from epidemiological studies in sections 7.1.3 and 8.4.4.3, the CD focuses on well-recognized criteria, including: (1) the *strength* of reported associations, including the magnitude and precision of reported effect estimates and their statistical significance; (2) the *robustness* of reported associations, or stability in the effect estimates after considering factors such as alternative models and model specification, potential confounding by co-pollutants, and issues related to the consequences of exposure measurement error; and (3) the *consistency* of the effects associations as observed by looking across results of multiple- and single-city studies conducted by different investigators in different places and times (CD, p. 8-40). Integrating more broadly across epidemiological and experimental evidence, the CD also

focuses on the *coherence* and *plausibility* of observed O_3-related health effects to reach judgments about causality (CD, section 8.6).

Subsequent to the final CD being published, CASAC sent a letter to the Administrator (Henderson, 2006b) providing additional advice on some key issues in order to inform specifically the preparation of this draft Staff Paper specifically and the review of the O_3 NAAQS in general. The issues related to assessment of epidemiological studies are addressed in this section and include the general issue of the utility of time-series epidemiological studies in assessing the risks from exposure to O_3 and other criteria pollutants, as well as related issues about exposure measurement error in O_3 mortality time-series studies and O_3 as a surrogate for the broader mix of photochemical oxidant pollution in time-series studies. Implications of these issues for staff conclusions about the adequacy of the current O_3 NAAQS and the identification of options for consideration will be considered in Chapter 6.

The following discussion summarizes the conclusions and judgments from the CD's summary of epidemiologic evidence and integrative assessment, focusing in particular on discussions of strength, robustness, and consistency in the epidemiological evidence; judgments in the CD about coherence and plausibility are summarized below in section 3.5. This section also addresses issues related to lag periods between O_3 ambient exposure levels and health outcomes, the nature of O_3-health effect concentration-response relationships, and the assessment of air pollutant mixtures containing O_3 in time-series epidemiological studies.

3.4.1 Strength of Associations

The strength of associations most directly refers to the magnitude of the reported relative risk estimates. Taking a broader view, the CD draws upon the criteria summarized in a recent report from the U.S. Surgeon General, which define strength of an association as "the magnitude of the association and its statistical strength" which includes assessment of both effect estimate size and precision, which is related to the statistical power of the study (CDC, 2004). In general, when associations are strong in terms of yielding large relative risk estimates, it is less likely that the association could be completely accounted for by a potential confounder or some other source of bias (CDC, 2004). With associations that yield small relative risk estimates it is especially important to consider potential confounding and other factors in assessing causality.

Effect estimates between O_3 and many health outcomes are generally small in size and could thus be characterized as weak. For example, effect estimates for associations with mortality generally range from 0.5 to 5% increases per 40 ppb increase in 1-hr max O_3 or equivalent, whereas associations for hospitalization range up to 50% increases per standardized O_3 increment. The CD particularly notes that there are several multicity studies for associations between short-term O_3 exposure and mortality or morbidity that, although small in size, have

great precision due to the statistical power of the studies (CD, p.8-40). That is, the associations were strong enough to have been reliably measured by the studies such that many of the associations can be distinguished from the null hypothesis with statistical confidence.

3.4.2 Robustness of Associations

Factors considered in assessing *robustness* include impact of exposure error, potential confounding by copollutants, and alternative models and model specifications, as evaluated in the CD (sections 7.1.3 and 8.4.4.3) and discussed below.

3.4.2.1 Exposure Error

In time-series epidemiologic studies, concentrations measured at centrally-located ambient monitoring stations are generally used to represent a community's exposure to ambient O_3. For time-series studies and panel studies that use these ambient concentrations, the emphasis is on the temporal (e.g., daily or hourly) changes in ambient O_3. In other cohort or cross-sectional studies, air quality data averaged over a period of months to years are used as indicators of a community's long-term exposure to ambient O_3 and other pollutants. In both types of analyses, exposure error is an important consideration, as actual exposures to individuals in the population will vary across the community.

In considering exposure error, it should be noted that total personal exposure can be partitioned into two types of sources, ambient and nonambient. As described in the CD, there are few sources of O_3 exposure for most people other than ambient air; potential indoor sources of O_3 include office equipment, air cleaners, and small electric motors (CD, p. 7-6). Sheppard (2005) notes that nonambient source exposures typically vary across individuals, but the community averages do not vary across communities. In addition, nonambient exposures are not likely to have strong temporal correlations. In contrast, ambient concentrations across individuals should be highly correlated, as they tend to vary over time similarly for everyone because of changes in source generation, weather, and season. The independence of ambient and non-ambient exposure sources has important implications. Sheppard et al. (2005) observes that when ambient and nonambient sources are independent, exposure variation due to nonambient source exposures behaves like Berkson measurement error and does not bias the effect estimates.

Ozone concentrations measured at central ambient monitoring sites may explain, at least partially, the variance in individual exposures to ambient O_3; however, this relationship is influenced by various factors related to building ventilation practices and personal behaviors. Further, the pattern of exposure misclassification error and the influence of confounders may differ across the outcomes of interest as well as in susceptible populations. As discussed in the CD Section 3.9, only a limited number of studies have examined the relationship between ambient O_3 concentrations and personal exposures to ambient O_3. One of the strongest

predictors of the relationship between ambient concentrations and personal exposures appears to be time spent outdoors. The strongest relationships were observed in outdoor workers (Brauer and Brook, 1995, 1997; O'Neill et al., 2003). For example, Brauer and Brook (1995, 1997) observed that in farmers who worked 6-14 hours outdoors each day, the personal to ambient O_3 concentration ratio was 0.96, with a Spearman correlation coefficient of 0.64. Statistically significant correlations between ambient concentrations and personal exposures were also observed for children, who likely spend more time outdoors in the warm season (Linn et al., 1996; Xue et al., 2005).

There is some concern about the extent to which ambient concentrations are representative of personal O_3 exposures of another particularly susceptible group of individuals, the debilitated elderly, and what impact that may have on mortality and hospitalization time-series studies. Those who suffer from chronic cardiovascular or respiratory conditions may tend to protect themselves more from environmental threats by reducing their exposure to both O_3 and its confounders, such as high temperature and PM, than those who are healthy. The correlation between ambient concentrations and personal exposure measurements in older adults (mean age 75 years) has been examined by Sarnat et al. (2001, 2005). These studies by Sarnat et al. also included children and COPD patients, and only results for the combined populations are reported. The first study conducted in Baltimore, MD observed no relationship between ambient concentrations and personal exposures in both the summer and the winter. However, the second study conducted in Boston, MA, found statistically significant associations between ambient O_3 concentrations and personal exposures to O_3. The regression coefficient was larger in the summer ($\beta = 0.27$), compared to the winter ($\beta = 0.04$), suggesting once again that time spent outdoors had a large influence on the relationship between ambient concentrations and personal exposures. Eight of 29 subjects had personal-ambient O_3 correlations greater than 0.8 during the summer.

Collectively, these studies observed that the daily averaged personal O_3 exposures from the population were well correlated with ambient O_3 concentrations despite the substantial variability that existed among the personal measurements. Averaging likely removes the noise associated with other sources of variation. These studies provide supportive evidence that ambient O_3 concentrations from central monitors may serve as valid surrogate measures for mean personal exposures experienced by the population, which is of most relevance for time-series studies. A better understanding of the relationship between ambient concentrations and personal exposures, as well as of the other factors that affect relationship will improve the interpretation of concentration-population health response associations observed.

The CD discusses the potential influence of exposure error on epidemiologic study results in Section 7.1.3.1. Zeger et al. (2000) outlined the three components to exposure measurement

error: (1) the use of average population rather than individual exposure data; (2) the difference between average personal ambient exposure and ambient concentrations at central monitoring sites; and (3) the difference between true and measured ambient concentrations (CD, p. 7-7). These components are expected to have different effects, with the first and third likely not causing bias in a particular direction ("nondifferential error") but increasing the standard error, while the second component may result in downward bias, or attenuation of the risk estimate (CD, pp. 7-7 to 7-8). Ambient exposure can be assumed to be the product of the ambient concentration and an attenuation factor (i.e., building filter). Panel studies and time-series studies that use ambient concentrations instead of personal exposure measurements will estimate a health risk that is attenuated by that factor. Navidi et al. (1999) used data from a children's cohort study to compare effect estimates from a simulated "true" exposure level to results of analyses from O_3 exposures determined by several methods. The results indicated that the use of O_3 exposures from personal sampling or microenvironmental approaches is associated with nondifferential error in O_3 effect estimates, compared with effect estimates from "true" exposures. However, O_3 exposures based on the use of ambient monitoring data overestimates the individual's O_3 exposure and thus generally results in O_3 effect estimates that are biased downward (CD, p. 7-8). Similarly, Zidek (1997) observed that a statistical analysis must balance bias and imprecision (error variance). For example, in a reanalysis of a study by Burnett et al. (1994) on the acute respiratory effects of ambient air pollution, Zidek et al. (1998) reported that accounting for measurement, as well as making a few additional changes to the analysis, resulted in qualitatively similar conclusions, but the effects estimates were considerably larger in magnitude (CD, p. 7-8).

A simulation study by Sheppard et al. (2005) also considered attenuation of the risk based on personal behavior, their microenvironment, and the qualities of the pollutant in time-series studies. Of particular interest is their finding that significant variation in nonambient exposure or in ambient source exposure that is independent of ambient concentration does not further bias the effect estimate. In other words, risk estimates were not further attenuated in time-series studies even when the correlations between personal exposures and ambient concentrations were weak.

In addition to overestimation of exposure and the resulting underestimation of effects, the use of ambient O_3 concentrations may obscure the presence of thresholds in epidemiologic studies (CD p. 7-9). Brauer et al. (2002) concluded that surrogate measures of exposure, such as those from centrally located ambient monitors, that were not highly correlated with personal exposures obscured the presence of thresholds in epidemiologic studies at the population level, even if a common threshold exists for individuals within the population.

Existing epidemiologic models may not fully take into consideration all of the biologically relevant exposure history or reflect the complexities of all of the underlying

biological processes. As discussed in the CD, Section 3.9, using ambient concentrations to determine exposure generally overestimates true personal O_3 exposures by approximately 2- to 4-fold in available studies, resulting in attenuated risk estimates. The implication is that the effects being estimated occur at fairly low exposures and the potency of O_3 is greater than these effects estimates indicate. As very few studies evaluating O_3 health effects with personal O_3 exposure measurements exist in the literature, effect estimates determined from ambient O_3 concentrations must be evaluated and used with caution to assess the health risks of O_3. Until more data on personal O_3 exposure becomes available, the use of routinely monitored ambient O_3 concentrations as a surrogate for personal exposures is not generally expected to change the principal conclusions from O_3 epidemiologic studies. Therefore, population health risk estimates derived using ambient O_3 levels from currently available observational studies, with appropriate caveats about personal exposure considerations, remain useful.

The CD recommends caution in the quantitative use of effect estimates calculated using ambient O_3 concentrations as they may lead to underestimation of the potency of O_3. However, staff observes that the use of these risk estimates for comparing relative risk reductions between alternative ambient O_3 standards considered in the risk assessment is less likely to suffer from this concern. In addition, as discussed in Chapter 5, staff has conducted an exposure assessment in conjunction with a portion of the health risk assessment that incorporates estimated population exposures in developing risk estimates for health outcomes based on controlled human exposure studies.

3.4.2.2 Confounding by Copollutants

Confounding occurs when a health effect that is caused by one risk factor is attributed to another variable that is correlated with the causal risk factor; epidemiological analyses attempt to adjust or control for potential confounders. Copollutants (e.g., PM, CO, SO_2 and NO_2) can meet the criteria for potential confounding in O_3-health associations if they are potential risk factors for the health effect under study and are correlated with O_3. Effect modifiers include variables that may influence the health response to the pollutant exposure (e.g., co-pollutants, individual susceptibility, smoking or age). Both are important considerations for evaluating effects in a mixture of pollutants, but for confounding, the emphasis is on controlling or adjusting for potential confounders in estimating the effects of one pollutant, while the emphasis for effect modification is on identifying and assessing the effects for different modifiers.

The CD observes that O_3 is generally not highly correlated with other criteria pollutants (e.g., PM_{10}, CO, SO_2 and NO_2), but may be more highly correlated with secondary fine particles, especially during the summer months (CD, p. 7-148). In addition, the correlation between O_3 and other pollutants may vary across seasons, since O_3 concentrations are generally higher in the

summer months. For example, positive associations are observed between O_3 and pollutants such as fine particles during the warmer months, but negative correlations may be observed between O_3 and these pollutants during the cooler months (CD, p. 7-17). Thus, the CD pays particular attention to the results of season-specific analyses and studies that assess effects of PM in potential confounding of O_3-health relationships in its discussions in section 7.6.4.

Multipollutant models are commonly used to assess potential confounding in epidemiological studies. As discussed in the CD, the limitations to the use of multipollutant models include the difficulty in interpreting results where the copollutants are highly colinear, or where correlations between pollutants change by season (CD, p. 7-150). This is particularly the situation where O_3 and a copollutant, such as sulfates, are formed under the same atmospheric condition; in such cases multipollutant models would produce unstable and possibly misleading results (CD, p. 7-152).

For mortality, the results from numerous multi-city and single-city studies are shown in Figure 7-22 of the CD. These results indicate that O_3-mortality associations do not appear to be substantially changed in multipollutant models including PM_{10} or $PM_{2.5}$ (CD, p. 7-101). Focusing on results of warm season analyses, Figure 7-23 of the CD shows effect estimates for O_3-mortality associations that are fairly robust to adjustment for PM in multipollutant models (CD, p. 7-102). In general, based on results from several single- and multiple-city studies, and on recent meta-analyses, the CD (p. 7-103) concludes that "copollutants generally do not appear to substantially confound the association between O_3 and mortality."

Similarly, multipollutant models are presented for associations between short-term O_3 exposures and respiratory hospitalization in Figure 7-12 of the CD; the CD concludes that copollutants generally do not confound the relationship between O_3 and respiratory hospitalization (CD, p. 7-79 to 7-80). Multipollutant models were not used as commonly in studies of relationships between respiratory symptoms or lung function with O_3, but the CD reports that results of available analyses indicate that such associations generally were robust to adjustment for $PM_{2.5}$ (CD, p. 7-154). For various co-pollutant models, in a large multicity study of asthmatic children (Mortimer et al., 2002), the O_3 effect was attenuated, but there was still a positive association. In Gent et al. (2003), effects of O_3, but not $PM_{2.5}$, remained statistically significant and even increased in magnitude in two-pollutant models (CD, p. 7-53).

Considering this body of studies, the CD concludes: "Multipollultant regression analyses indicated that O_3 risk estimates, in general, were not sensitive to the inclusion of copollutants, including $PM_{2.5}$ and sulfate. These results suggest that the effects of O_3 on respiratory health outcomes appear to be robust and independent of the effects of other copollutants (CD, p. 7-154)." We use the results of single-pollutant model results in presentation of results in this chapter and in quantitative risk assessments conducted as part of this review (see Chapter 5) for

purposes of comparing results from different studies. However, we also include the use of multi-pollutant model results in presenting risk estimates, when available, to more completely characterize the quantitative health risks associated with ambient O_3 levels.

3.4.2.3 Model Specification

The CD observes that one challenge of time-series epidemiological analysis is assessing the relationship between O_3 and health outcomes while avoiding bias due to confounding by other time-varying factors, particularly seasonal trends and weather variables (CD, p. 7-14). These variables are of particular interest because O_3 concentrations have a well-characterized seasonal pattern (see Chapter 2) and are also highly correlated with changes in temperature. Thus it can be difficult to distinguish whether effects are associated with O_3 or with seasonal or weather variables in statistical analyses.

Section 7.1.3.4 of the CD discusses statistical modeling approaches that have been used to adjust for time-varying factors, highlighting a series of analyses that were done in a Health Effects Institute-funded reanalysis of numerous time-series studies. While the focus of these reanalyses was on associations with PM, a number of investigators also examined the sensitivity of O_3 coefficients to the extent of adjustment for temporal trends and weather factors. In addition, several recent studies, including U.S. multi-city studies (Bell et al., 2005; Huang et al., 2005; Schwartz et al., 2005) and a meta-analysis study (Ito et al., 2005), evaluated the effect of model specification on O_3-mortality associations. As discussed in the CD (section 7.6.3.1), these studies generally report that associations reported with O_3 are not substantially changed with alternative modeling strategies for adjusting for temporal trends and meteorologic effects. In the meta-analysis by Ito et al. (2005), a separate multicity analysis was presented that found that alternative adjustments for weather resulted in up to 2-fold difference in the O_3 effect estimate. However, significant confounding can occur when strong seasonal cycles are present, suggesting that season-specific results are more generally robust than year-round results in such cases. The CD concludes that "seasonal dependence of O_3-mortality effects complicates interpretation of O_3 risk estimates calculated from year-round data without adequate adjustment of temporal trends" (CD, p. 7-99), and that more work is needed in this area to reduce the uncertainty involved in the epidemiologic interpretation of O_3 effect estimates (CD, p. 7-141).

A number of epidemiological studies have conducted season-specific analyses, as discussed in section 7.6.3.2 of the CD. As observed above in section 3.3, such studies have generally reported stronger and more precise effect estimates for O_3 associations in the warm season than in analyses conducted in the cool seasons or over the full year. For assessing relationships between O_3 and health outcomes, the CD highlights several reasons to focus on warm season analyses: (1) the seasonal nature of O_3 concentrations; (2) the relationship between

O_3 formation and temperature; (3) correlations between other pollutants, particularly fine particles, and O_3 variations across seasons in some areas; and (4) factors affecting exposure to ambient O_3, such as air conditioning use, varies seasonally in most areas of the U.S.. We have therefore focused on epidemiological findings from warm season analyses, where available, for qualitative assessments and for the quantitative risk assessment discussed in Chapter 5.

3.4.3 Consistency

Consistency refers to the persistent finding of an association between exposure and outcome in multiple studies of adequate power in different persons, places, circumstances and times (CDC, 2004). In considering results from multicity studies and single-city studies in different areas, the CD observes general consistency in effects of short-term O_3 exposure on mortality, respiratory hospitalization and other respiratory health outcomes (CD, p. 8-41). The variations in effects that are observed may be attributable to differences in relative personal exposure to O_3, as well as varying concentrations and composition of copollutants present in different regions. Thus, the CD concludes that "consideration of consistency or heterogeneity of effects is appropriately understood as an evaluation of the similarity or general concordance of results, rather than an expectation of finding quantitative results with a very narrow range" (CD, p.8-41).

3.4.4 Lag Structure in Short-term Exposure Studies

In the short-term exposure epidemiological studies, many investigators have tested associations for a range of lag periods between the health outcome and O_3 concentration (see CD, sections 7.1.3.3). The CD observes that the selection of an appropriate lag period can depend on the health outcome under study. For example, if cough is resulting from the irritant action of O_3, that would be expected to occur with a short lag time; however, exacerbation of asthma through an inflammatory response might occur up to several days after initial exposure (CD, p. 7-12). For both mortality and respiratory hospital admissions, the CD reports that most significant associations between O_3 and mortality were observed with O_3 measured on the same day or a 1-day lag period in studies using individual lag periods (CD, p. 7-14). In U.S. multi-city studies, larger effect estimate sizes were reported for the O_3-mortality relationship with the distributed lag structure (CD, p. 7-88). Field studies of lung function or respiratory symptoms reported associations with O_3 across a range of lag periods from exposure on the same day to exposures averaged over several days (CD, sections 7.2.3 and 7.2.4). Cardiovascular effects appeared to be associated with O_3 at shorter lag periods; cardiovascular health outcomes such as changes in cardiac autonomic control were associated with O_3 measured on the same day (CD, section 7.2.7.1). In addition, Peters et al. (2001) reported a positive but not statistically

significant association between myocardial infarction onset and O_3 with very short lag times of 1- to 4 hr (CD, p. 7-64).

In focusing on an effect estimate reported for any individual lag period, the CD observes that it is important to consider the pattern of results across the series of lag periods. If there is an apparent pattern of results across the different lags, then selecting the single-day lag with the largest effect from a series of positive associations is still likely to underestimate the overall effect size, since single-day lag effect estimates do not fully capture the risk that may be distributed over adjacent or other days (CD, p. 7-13). However, if the reported effect estimates vary substantially across lag periods, any result for a single day may well be biased (CD, p. 7-14). If the effect of O_3 on health outcomes persists over several days, distributed lag model results may provide more accurate effect estimates for quantitative assessment than an effect estimate for a single lag period (CD, p. 7-12). Conversely, if the underlying O_3-health relationship is truly an acute effect, then a distributed lag model would likely result in a reduced effect estimate size that may underestimate the effect (CD, p. 7-12).

On this basis, the CD focuses on effect estimates from models using 0- or 1-day lag periods, with some consideration of multi-day lag effects (CD, p. 7-14). For quantitative assessments, we conclude that it is appropriate to use results from lag period analyses consistent with those reported in the CD, focusing on single day lag periods of 0-1 days for associations with mortality or respiratory hospitalization, depending on availability of results (CD, p. 7-14). When available, distributed lag model results also have been used in the quantitative risk assessment. However, for those few studies that show inconsistent patterns, the use of single-day lag results is not appropriate for inclusion in the quantitative assessment.

3.4.5 Concentration-Response Relationships and Potential Thresholds

It has been recognized that it is reasonable to expect that there likely are biological thresholds for different health effects in individuals or groups of individuals with similar innate characteristics and health status. For O_3 exposure, individual thresholds would presumably vary substantially from person to person due to individual differences in genetic susceptibility, pre-existing disease conditions and possibly individual risk factors such as diet or exercise levels (and could even vary from one time to another for a given person). Thus, it would be difficult to detect a distinct threshold at the population level, below which no individual would experience a given effect, especially if some members of a population are unusually sensitive even down to very low concentrations (U.S. EPA, 2004, p. 9-43, 9-44).

Some studies have tested associations between O_3 and health outcomes after removal of days with higher O_3 levels from the data set; such analyses do not necessarily indicate the presence or absence of a threshold, but provide some information on whether the relationship is

found using only lower-concentration data. For example, using data from 95 U.S. cities, Bell et al. (2004) found that the effect estimate for an association between short-term O_3 exposure and mortality was little changed when days exceeding 60 ppb (24-hr average) were excluded in the analysis. Bell et al. (2006) found no difference in estimated effect even when all days with 24-hr O_3 concentrations <20 ppb were excluded (CD, p. 8-43). Using data from 8 U.S. cities, Mortimer and colleagues (2002) also reported that associations between O_3 and both lung function and respiratory symptoms remained statistically significant and of the same or greater magnitude in effect size when concentrations greater than 80 ppb (8-hr avg) were excluded (CD, p. 7-46). Several single-city studies are also summarized in section 7.6.5 of the CD that report similar findings of associations that remain or are increased in magnitude and statistical significance when data at the upper end of the concentration range are removed.

Other time-series epidemiological studies have used statistical modeling approaches to evaluate whether thresholds exist in associations between short-term O_3 exposure and mortality. As discussed in section 7.6.5 of the CD, one European multi-city study included evaluation of the shape of the concentration-response curve, and observed no deviation from a linear function across the range of O_3 measurements from the study (Gryparis et al., 2004; CD p. 7-154). Several single-city studies also observed a monotonic increase in associations between O_3 and morbidity that suggest that no population threshold exists (CD, p. 7-159).

On the other hand, a study in Korea used several different modeling approaches and reported that a threshold model provided the best fit for the data. The results suggested a potential threshold level of about 45 ppb (1-hr maximum concentration; < 35 ppb, 8-hr avg) for an association between mortality and short-term O_3 exposure during the summer months (Kim et al., 2004; CD, p. 8-43). The authors reported larger effect estimates for the association for data above the potential threshold level, suggesting that an O_3-mortality association might be underestimated in the non-threshold model. A threshold analysis recently reported by Bell et al. (2006) for 98 U.S. communities, including the same 95 communities in Bell et al. (2004), indicated that if a population threshold existed for mortality, it would likely fall below a 24-h average O_3 concentration of 15 ppb (< 25 ppb, 8-hr avg). In addition, Burnett and colleagues (1997) plotted the relationships between air pollutant concentrations and both respiratory and cardiovascular hospitalization, and it appears in these results that the associations with O_3 are found in the concentration range above about 30 ppb (1-hr maximum; < 25 ppb, 8-hr avg).

Vedal and colleagues (2003) reported a significant association between O_3 and mortality in British Columbia where O_3 concentrations were quite low (mean concentration of 27.3 ppb). The authors did not specifically test for threshold levels, but the fact that the association was found in an area with such low O_3 concentrations suggests that any potential threshold level would be quite low in this data set.

In summary, the CD finds that, taken together, the available evidence from clinical and epidemiological studies suggests that no clear conclusion can now be reached with regard to possible threshold levels for O_3-related effects (CD, p. 8-44). We also recognize that the available epidemiological evidence neither supports nor refutes the existence of thresholds at the population level for effects such as increased hospital admissions and premature mortality. There are limitations in epidemiological studies that make discerning thresholds in populations difficult, including low data density in the lower concentration ranges, the possible influence of exposure measurement error, and interindividual differences in susceptibility to O_3-related effects in populations. We recognize, however, the possibility that thresholds for individuals may exist in reported associations at fairly low levels within the range of air quality observed in the studies but not be detectable as population thresholds in epidemiological analyses. Based on the CD's conclusions, we judge that there is insufficient evidence to support use of potential threshold levels in quantitative risk assessments and that it is appropriate to estimate risks within the range of air quality concentrations down to estimated policy-relevant background level.

3.4.6 Health Effects of Pollutant Mixtures Containing O_3

The potential for O_3-related enhancements of PM formation, particle uptake, and exacerbation of PM-induced cardiovascular effects underscores the importance of considering contributions of O_3 interactions with other often co-occurring air pollutants to health effects due to O_3-containing pollutant mixes. Chapters 4, 5, and 6 of the CD provide a discussion of experimental studies that evaluate interactions of O_3 with other co-occurring pollutants. Some examples of important pollutant mixture effects noted there are highlighted below.

In Chapter 4, the CD noted some important interactive effects of coexposures to O_3, and NO_2 and SO_2, two other common gaseous copollutants found in ambient air mixes. A study by Rigas et al. (1997) showed that continuous exposure of healthy human adults to SO_2 or to NO_2 increased inhaled bolus O_3 absorption, while continuous exposure to O_3 alone decreased bolus absorption of O_3. This suggests enhancement of O_3 uptake by NO_2 or SO_2 coexposure in ambient air mixes. Another study by Jenkins et al. (1999) showed that asthmatics exhibited enhanced airway responsiveness to house dust mite following exposures to O_3, NO_2, and the combination of the two gases (CD, Chapter 6). Spirometric responses, however, were impaired only by O_3 and O_3+NO_2 at higher concentrations. On the other hand, animal toxicology studies (CD, Chapter 5) that evaluated exposures to O_3 in mixture with NO_2, formaldehyde, and PM demonstrated additive, synergistic or antagonistic effects, depending on the exposure regimen and the specific health endpoints evaluated.

Several studies have demonstrated the enhancement by O_3 exposure of various respiratory responses of sensitive individuals to allergens. For example, Peden et al. (1995) showed O_3-

induced increased response to nasal allergen challenge among allergic asthmatic subjects, and Michelson et al. (1999) showed promotion by 0.4 ppm O_3 exposure of inflammatory cell influx in response to nasal allergen challenge in asymptomatic dust-mite sensitive asthmatics. In addition, Jörres et al. (1996) demonstrated enhancement by 0.25 ppm O_3 exposure of airway responsiveness in mildly allergic asthmatics that was increased in response to an individual's historical allergen (grass and birch pollen, house dust mite, animal dander). These results were further extended by Holz et al. (2002) who showed that repeated daily exposure to 0.125 ppm O_3 for 4 days exacerbated lung function decrements (e.g., decreased FEV_1) in response to bronchial allergen challenges among subjects with preexisting allergic airway disease, with or without asthma (see Chapter 6 of the CD). This suggests that O_3 exposure can place allergic people who do not have asthma, as well as people who do have asthma, at increased risk for allergic respiratory effects. Consistent with and supporting the above findings are animal toxicology studies reviewed in detail by Harkema and Wagner (2005), which indicate that (a) O_3-induced epithelial and inflammatory responses in laboratory rodents are markedly enhanced by coexposure to inhaled biogenic substances (e.g., bacterial endotoxin or ovalbumin, an experimental aeroallergen) and (b) adverse airway effects of biogenic substances can be exacerbated by coexposure to O_3.

Also of much note is a newly emerging literature which indicates that O_3 can modify the biological potency of certain types of ambient PM, as shown by experimental tests. For example, as described in the CD, Section 5.4.2, the reaction of diesel PM with 0.1 ppm O_3 for 48 hr increased the potency (compared to non-exposed or air-exposed diesel PM) to induce neutrophil influx, total protein, and LDH in lung lavage fluid in response to intratracheal PM instillation in rats (Madden et al., 2000). However, the potency of carbon black particles was not enhanced by exposure to O_3, suggesting that O_3 reaction with organic components of the diesel PM were responsible for the observed increased diesel PM effects.

Potential interaction of O_3 with fine PM in aged rats was examined by Kleinman et al. (2000). In this study the effects of fine PM containing two common toxic constituents, ammonium bisulfate (ABS, 0.3 μm 70 $\mu g/m^3$) and elemental carbon (C, 0.3 μm 50 $\mu g/m^3$) and a mixture (ABS + C) with 0.2 ppm O_3 was evaluated on aged rat lung structure and macrophage function. Exposures of O_3, elemental carbon or ABS alone did not cause significant lung injury, lung tissue collagen content or respiratory burst activity. On the other hand, mixtures (ABS + C + O_3) caused significant lung injury as assessed by increased cell proliferation response in lung epithelial and interstitial cells, loss of lung tissue collagen and increase in respiratory burst and phagocytic activity.

The majority of toxicological studies discussed in the CD evaluated effects of individual pollutants or simple mixtures of the constituents of urban smog mixtures, and these toxicology

studies may not fully explain epidemiologic findings that have increasingly shown ambient O_3, other gaseous pollutants, and/or PM to be associated with various health effects at relatively low concentrations. In a recent report, Sexton et al. (2004) utilized "smog chambers", i.e., environmental irradiation chambers to generate synthetic photochemical oxidants mixtures similar to urban smog, and studied the toxicity of such mixtures on the inflammatory response of A549 cells in an in vitro exposure system. In this preliminary study, the authors found the simulated urban photochemical oxidant mixture generated with the addition of O_3 to have enhanced toxicity (as assessed by the expression of IL-8 mRNA). Additional toxicology studies using similar realistic air pollution smog mixtures in the future may provide more relevant biological understanding for the potential interactions that occur in the ambient air among various pollutants.

The body of epidemiological studies discussed in this Staff Paper emphasizes the role of O_3 acting autonomously, from a statistical sense, in association with a variety of adverse respiratory and cardiovascular effects. Despite a variety of plausible mechanisms, there exists a general consensus suggesting that O_3, either directly or through initiation, interferes with basic cellular oxidation processes responsible for inflammation, reduced antioxidant capacity, atherosclerosis and other effects. Reasoning that O_3 influences cellular chemistry through basic oxidative properties (as opposed to a unique chemical interaction), other reactive oxidizing species (ROS) in the atmosphere acting either independently or in combination with O_3 may also contribute to a number of adverse respiratory and cardiovascular health effects. Consequently, the role of O_3 should be considered more broadly as O_3 behaves as a generator of numerous oxidizing species in the atmosphere.

All of the above types of interactive effects of O_3 with other co-occurring gaseous and nongaseous viable and nonviable PM components of ambient air mixes argue for not only being concerned about direct effects of O_3 acting alone, but also the need for viewing O_3 as a surrogate indicator for air pollution mixes which may enhance risk of adverse effects due to O_3 acting in combination with other pollutants. Viewed from this perspective, those epidemiologic findings of morbidity and mortality associations, with ambient O_3 concentrations extending to concentrations below 0.08 ppm, become more understandable and plausible.

3.5 BIOLOGICAL PLAUSIBILITY AND COHERENCE OF EVIDENCE

This section summarizes material contained in section 8.4.3 and section 8.6 of the CD, which integrates epidemiological studies with mechanistic information from controlled human exposure studies and animal toxicological studies to draw conclusions regarding the coherence of evidence and biological plausibility of O_3-related health effects. For its assessment, the CD's discussion draws from epidemiological evidence on a range of relevant health endpoints (from

cardiopulmonary and physiological changes to morbidity and mortality) and assessment of available toxicological and biochemical evidence on potential plausible causal relationships for the observed epidemiological associations (CD, p. 8-45).

3.5.1 Animal-to-Human Extrapolation Issues

Table 3-1 (Table 8-1, CD, p. 8-29) summarizes physiological and biochemical observations which represent the knowledge base available from studies in humans and animals that support conclusions drawn about biological alterations that cause acute O_3-induced health effects. Table 3-1 was based upon experimental data (contained in CD Chapters 5 and 6, as well as the chapter annexes), which used environmentally relevant exposure regimens. Although most of the acute O_3-induced biological alterations are transient and attenuate over time, this does not mean that injury at the cellular and tissue level does not continue. Most inflammatory markers (e.g., PMN influx) attenuate after 5 days of exposure, but markers of cell damage (e.g., LDH enzyme activity) do not attenuate and continue to increase. The time-line for resolution of many of the physiological and biological parameters presented in Figure 3-2 (Figure 8-3, CD, p. 8-30) differ for healthy human subjects and those with underlying cardiopulmonary diseases. The CD also notes that alterations in acute O_3-induced cellular and molecular changes observed in human airway epithelium evolve over time, depicted in Figure 3-3 (Figure 8-4, CD, p. 8-31), and that the knowledge of this profile is important in assessing biological plausibility to integrate across evidence of various health endpoints.

The similarities in physiological, biochemical and pathological processes between humans and many animal species are due to the high level of genome sequence homology that exists across species (CD, p. 8-28). It is this homology that supports the use of knowledge gained on initiation, progression, and treatment regimes for disease processes across species, especially on the acute O_3-induced effects in the respiratory tracts of humans and various animal species, as depicted in CD Table 3-1 and Figures 3-2 and 3-3. The similarities observed in human and rat respiratory system effects (e.g., in measures of lung function, ventilatory response, host defense), attenuation, and at higher levels of cellular organization (e.g., neutrophilic inflammation, macrophage phagocytosis processes) lend support to animal-to-human extrapolation. This is particularly important in collecting information that would not be possible to gather in human exposure or epidemiological studies but may corroborate data from both types of studies.

Table 3-1. Acute O_3-induced Physiological and Biochemical Changes in Human and Animals

Physiological/Biochemical Alterations	Human Exposure Studies [1,2]	Animal Toxicology Studies [3,4]
Pulmonary Function:	↓ FEV_1 ↑ Frequency of breathing (rapid, shallow) ↓ FVC (cough, breathing discomfort, throat irritation, wheezing) Mild bronchoconstriction	↑ Frequency of breathing (rapid, shallow) ↓ FVC
Airway Responsiveness:	↑ (neuronal involvement) Change in lung resistance	↑ (vagal mediation) Change in lung resistance
Inflammation:	Yes ↑ inflammatory mediators	Yes ↑ inflammatory mediators
Reactive Oxygen Species:	↑	↑
Host Defense:	↑ particle clearance ↑ permeability ↓ AM phagocytosis	↑ particle clearance ↑ permeability ↓ clearance of bacteria ↑ severity of infection ↑ mortality & morbidity
Lung Injury: Morphology:	Yes	Yes
Susceptibility:	Age, Interindividual variability Disease status Polymorphism in certain genes being recognized	Species-specific differences Genetic basis for susceptibility indicated
Cardiovascular Changes:	Impairment in arterial O_2 transfer Ventilation-perfusion mismatch (suggesting potential arterial vasoconstriction) ↑ rate pressure product[5] ↑ myocardial work[5]	Heart rate ↓ core body temperature ↑ atrial natriuretic factor Role for platelet activity factor (PAF) indicated Increased pulmonary vascular resistance

[1] Controlled chamber exposure studies in human volunteers were carried out for a duration of 1 to 6.6 h with O_3 concentration in the range of 0.08-0.40 ppm with intermittent exercise.
[2] Data on some of the biochemical parameters were obtained from in vitro studies using cells recovered from BALF.
[3] Responses were observed in animal toxicology studies with exposure for a duration of 2 to 72 h with O_3 concentration in the range of 0.1 to 2.0 ppm.
[4] Various species (mice, rat, guinea pigs and rabbit) and strains.
[5] In hypertensive subjects.

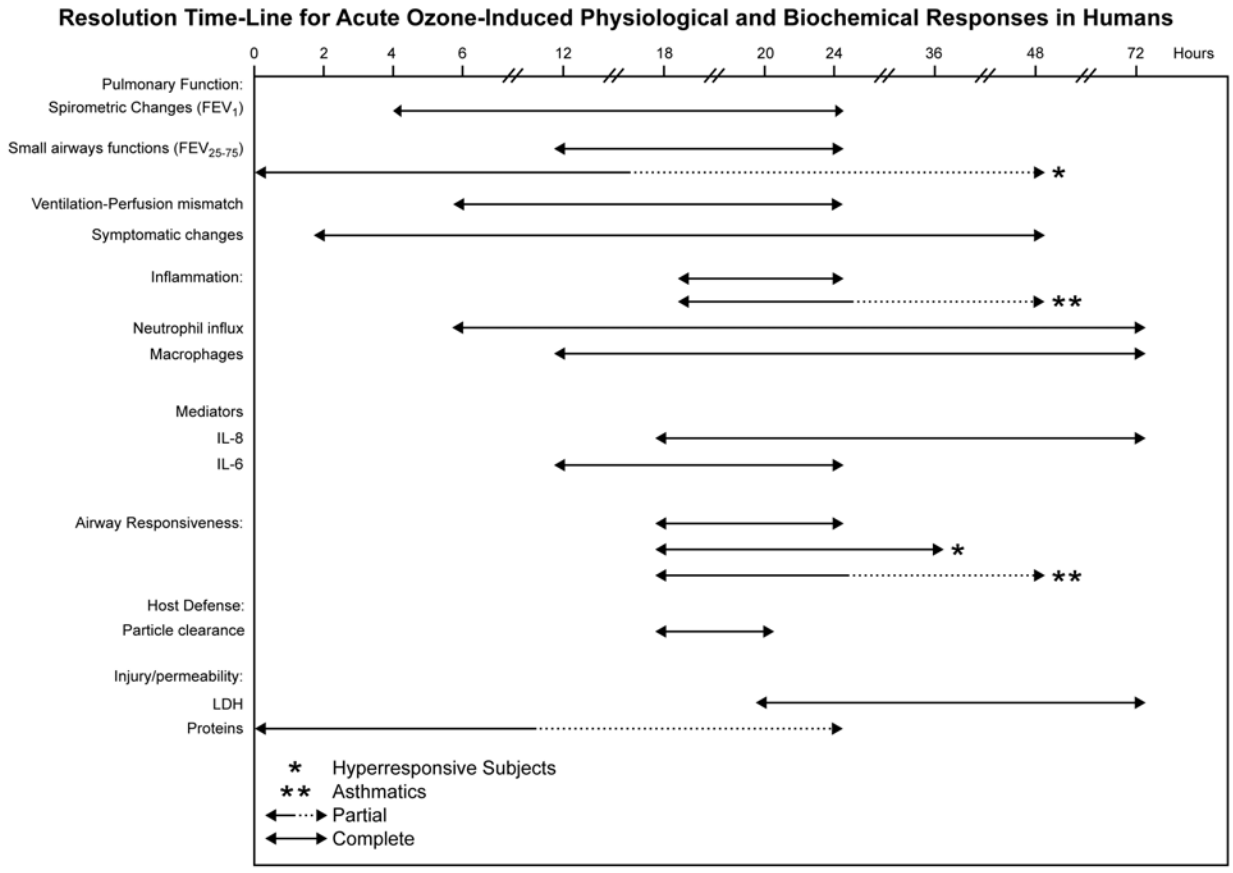


Figure 3-2. Resolution time-line for the respiratory, physiological, and biochemical parameters are derived from studies reported in the CD, Chapter 6 and Chapter 6 Annex.

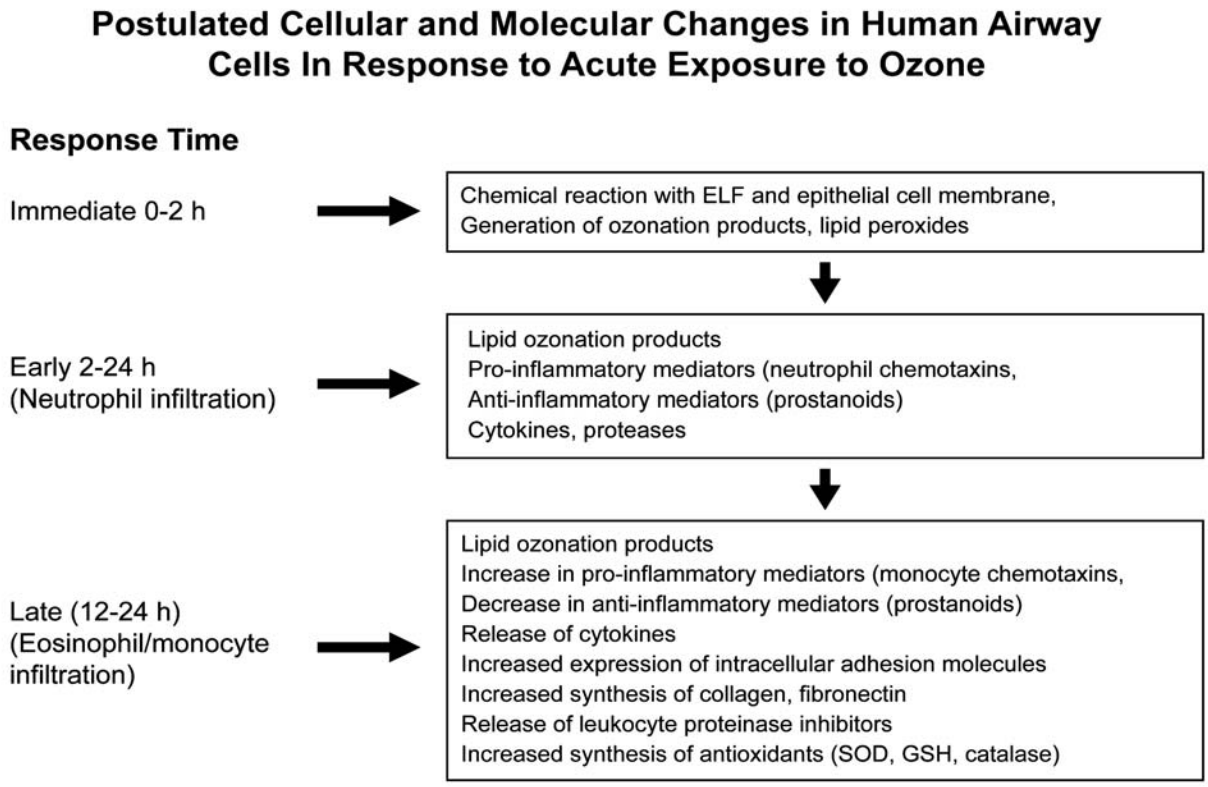


Figure 3-3. Acute (1-8 h) O_3 exposure-induced cellular and molecular changes and timelines for their resolution depicted here are derived from the data reported in Leikauf et al. (1995) and Mudway and Kelly (2000)3-4.Acute (1-8 h) O_3 exposure-induced cellular and molecular changes and timelines for their resolution depicted here are derived from the data reported in Leikauf et al. (1995) and Mudway and Kelly (2000).

Quantitative extrapolation requires a combination of dosimetry, end point homology, and species sensitivity. Although uncertainties continue to exist, animal-to-human extrapolation can be done for a number of health endpoints with sufficient accuracy to be useful in evaluating the potential for human health effects. For example, the amount of protein in lavage fluid shows a striking relationship when interspecies dosimetric adjustments are applied to the individual species and exposure studies. One study (Hatch et al., 1994) of inflammatory markers suggests that a 2 ppm O_3 exposure in sedentary rats approximates a 0.4 ppm exposure in exercising humans (i.e., if one considers the dosimetry, the sensitivities of rats and humans are consistent). This supports the use of some animal data collected at higher O_3 exposures to help understand molecular changes in acutely exposed humans (CD, 8-31). Also of importance are the chronic exposure studies (12 to 24 months) reporting lesions in animals caused by long-term O_3 exposures that may analogously occur in humans with long-term (months, years) exposure to relatively high levels of O_3. However, specific exposure patterns of O_3 concentrations that could produce comparable alterations in human lungs remain to be substantiated (CD, p. 8-32).

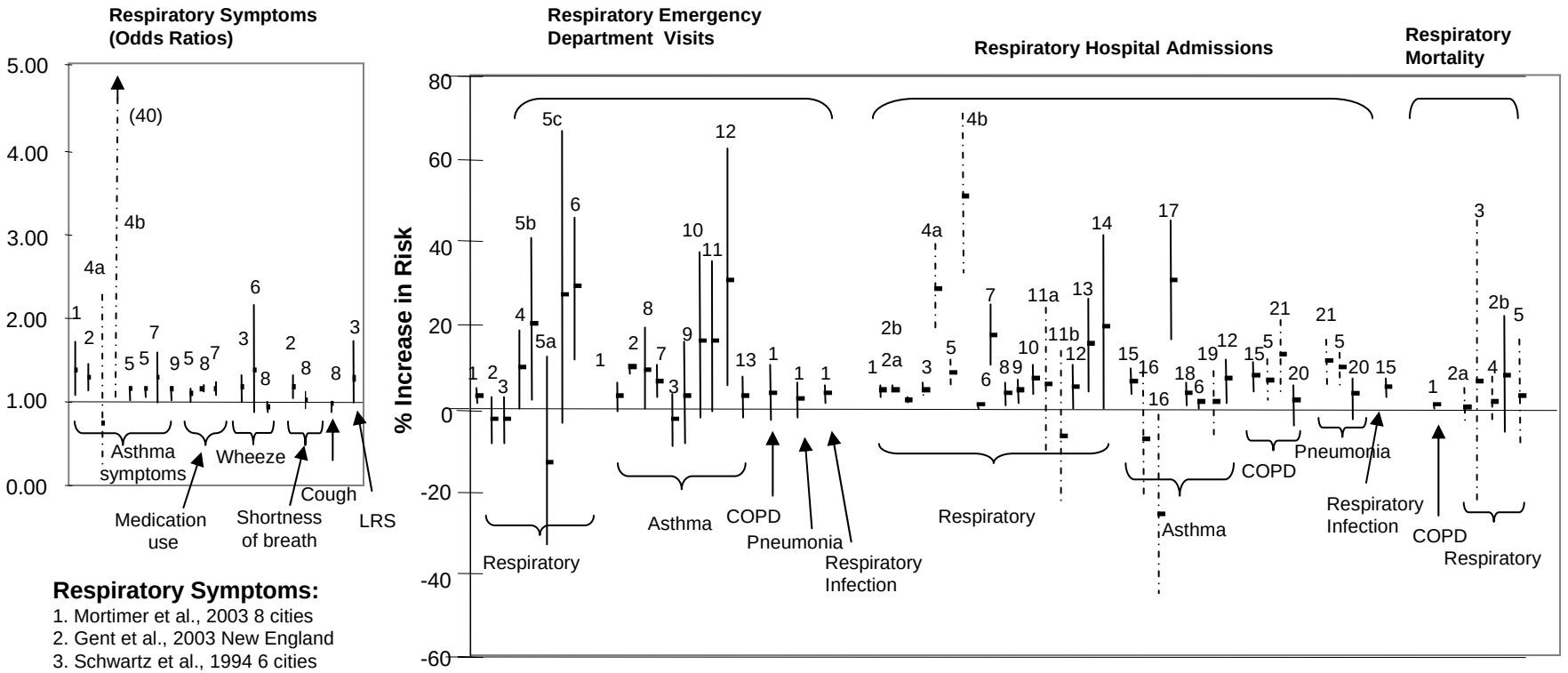
3.5.2 Coherence and Plausibility of Short-term Effects on the Respiratory System

Acute respiratory morbidity effects that have been associated with short-term exposure to O_3 include such health endpoints as decrements in lung function, increased airway responsiveness, airway inflammation, epithelial injury, immune system effects, ED visits for respiratory diseases, and hospitalization due to respiratory illness

Recent epidemiological studies have supported evidence available in the previous O_3 NAAQS review on associations between ambient O_3 exposure and decline in lung function for children. Earlier observations that children and asthmatic individuals are particularly susceptible to ambient O_3 are supported by a meta-analysis (Kinney et al., 1996) of summer camp studies. The CD (p. 8-34) concludes that exposure to ambient O_3 has a significant effect on lung function and is associated with increased respiratory symptoms and medication use, particularly in asthmatics.

Short-term exposure to O_3 has also been associated with more severe morbidity endpoints, such as ED visits and hospital admissions for respiratory cases, including specific respiratory illness (e.g., asthma) (CD, sections 7.3.2 and 7.3.3). In addition, a few epidemiological studies have reported positive associations between short-term O_3 exposure and respiratory mortality, though the associations are not generally statistically significant (CD, p. 7-108).

Considering the evidence from epidemiological studies, the results described above provide evidence for coherence in O_3-related effects on the respiratory system. Effect estimates from U.S. and Canadian studies are shown in Figure 3-4, where it can be seen that



Respiratory Symptoms:

1. Mortimer et al., 2003 8 cities
2. Gent et al., 2003 New England
3. Schwartz et al., 1994 6 cities
4. Delfino et al., 2003 San Diego (a=score>1; b=score>2)
5. Ross et al., 2002 East Moline
6. Neas et al., 1995 Uniontown
7. Delfino et al., 1998 San Diego
8. Ostro et al., 2001 S. California
9. Thurston et al., 1997 Connecticut

Emergency Department Visits:

1. Peel et al., 2005 Atlanta
2. Wilson et al., 2005 Portland NH
3. Wilson et al., 2005 Manchester NH
4. Stieb et al., 1996 St. John Canada
5. Jones et al., 1995 Baton Rouge (a=0-17 yo; b=18-60 yo; c=>60 yo)
6. Delfino et al., 1997 Montreal
7. Tolbert et al., 2000 Atlanta
8. Jaffe et al., 2003 3 Ohio cities
9. Jaffe et al., 2003 Cleveland
10. Jaffe et al., 2003 Columbus
11. Jaffe et al., 2003 Cincinnati
12. Friedman et al., 2001 Atlanta
13. Zhu et al., 2003 Atlanta

Hospital Admissions:

1. Burnett et al., 1994 Toronto
2. Gwynn et al., 2001 New York City (a=nonwhite; b=white)
3. Gwynn et al., 2000 Buffalo
4. Yang et al., 2003 Vancouver (a=<3 yo; b=65+ yo)
5. Moolgavkar et al., Minneapolis/St. Paul
6. Thurston et al., 1992 New York City
7. Burnett et al., 1997 Toronto
8. Schwartz et al., 1996 Cleveland
9. Delfino et al., 1994 Montreal
10. Burnett et al., 1997 16 Canadian cities
11. Luginaah et al., 2003 Windsor (a=males; b=females)
12. Thurston et al., 1992 Buffalo
13. Thurston et al., 1994 Toronto
14. Schwartz et al., 1996 Spokane
15. Burnett et al., 1999 Toronto
16. Lin et al., 2003 Windsor
17. Burnett et al., 2001Toronto
18. Sheppard et al., 2003 Seattle
19. Nauenberg and Basu, 1999 Los Angeles
20. Ito, 2003 Detroit
21. Schwartz et al., 1994 Detroit

Respiratory Mortality:

1. Moolgavkar, 2003 Cook County
2. Ito, 2003 Detroit (a=1986-1990; b=1992-1994)
3. Vedal et al., 2003 Vancouver
4. Villeneuve et al., 2003 Vancouver
5. Ostro et al., 2003, Coachella Valley

Figure 3-4. Effect estimates (with 95% confidence intervals) for associations between short-term ozone exposure and respiratory health outcomes.

Effect estimates expressed as odds ratios for associations with respiratory symptoms and % increase for other outcomes, per standardized increments: 20 ppb for 24-hr O_3, 30 ppb for 8-hr O_3, and 40 ppb for 1-hr O_3, presented in order of decreasing statistical power from left to right in each category. Dotted line (blue) indicates all year analyses; solid line (red) indicates warm season results. LRS=lower respiratory symptoms; COPD=chronic obstructive pulmonary disease

mostly positive associations have been reported with respiratory effects ranging from respiratory symptoms, such as cough or wheeze, to hospitalization for various respiratory diseases, and there is suggestive evidence for associations with respiratory mortality. Many of the reported associations are statistically significant.

Considering also evidence from toxicological, chamber, and field studies, the CD (section 8.6) discusses biological plausibility and coherence of evidence for acute O_3-induced respiratory health effects. Inhalation of O_3 for several hours while subjects are physically active can elicit both acute adverse pathophysiological changes and subjective respiratory tract symptoms (CD, section 8.4.2). Acute pulmonary responses observed in healthy humans exposed to O_3 at ambient concentrations include: decreased inspiratory capacity; mild bronchoconstriction; rapid, shallow breathing during exercise; subjective symptoms of tracheobronchial airway irritation, including cough and pain on deep inspiration; decreases in measures of lung function (e.g., FVC and FEV_1); and increased airway resistance (SR_{aw}). The severity of symptoms and magnitude of response depends on inhaled dose, individual O_3 sensitivity, and the degree of attenuation or enhancement of response resulting from previous O_3 exposures. Lung function studies of several animal species acutely exposed to relatively low O_3 levels (0.25 to 0.4 ppm) show responses similar to those observed in humans, including increased breathing frequency, decreased tidal volume, increased resistance, and decreased FVC. Alterations in breathing pattern return to normal within hours of exposure, and attenuation in functional responses following repeated O_3 exposures is similar to those observed in humans.

Physiological and biochemical alterations investigated in controlled human exposure and animal toxicology studies tend to support certain hypotheses of underlying pathological mechanisms which lead to the development of respiratory-related effects reported in epidemiology studies (e.g., increased hospitalization and medication use). Some of these are: (a) decrements in lung function, (b) bronchoconstriction, (c) increased airway responsiveness, (d) airway inflammation, (e) epithelial injury, (f) immune system activation, (g) host defense impairment, and sensitivity of individuals, such as age, genetic susceptibility, and the degree of attenuation present due to prior exposures. The time sequence, magnitude, and overlap of these complex events, both in terms of development and recovery (as depicted in Figures 3-2 and 3-3), illustrate the inherent difficulty of interpreting the biological plausibility of O_3-induced cardiopulmonary health effects (CD, p. 8-48).

The interaction of O_3 with airway epithelial cell membranes and epithelial lining fluid (ELF) to form lipid ozonation products and ROS is supported by numerous human, animal and in vitro studies. Ozonation products and ROS initiate a cascade of events that lead to oxidative stress, injury, inflammation, airway epithelial damage and increased epithelial damage and increased alveolar permeability to vascular fluids. Repeated respiratory inflammation can lead to

a chronic inflammatory state with altered lung structure and lung function and may lead to chronic respiratory diseases such as fibrosis and emphysema (CD, section 8.6.2). Continued respiratory inflammation also can alter the ability to respond to infectious agents, allergens and toxins. Acute inflammatory responses to O_3 are well documented, and lung injury can become apparent within 3 hr after exposure in humans. Ozone-induced lung injury and subsequent disruption of the airway epithelial barrier has been implicated in increased mucociliary clearance of particles in human subjects.

Taken together, the CD concludes that the evidence from experimental human and animal toxicology studies indicates that acute O_3 exposure is causally associated with respiratory system effects, including O_3-induced pulmonary function decrements, respiratory symptoms, lung inflammation, and increased lung permeability, airway hyperresponsiveness, increased uptake of nonviable and viable particles, and consequent increased susceptibility to PM-related toxic effects and respiratory infections (CD, p. 8-48).

3.5.3 Coherence and Plausibility of Effects on the Cardiovascular System

There is very limited experimental evidence of animals and humans that has evaluated possible mechanisms or physiological pathways by which acute O_3 exposures may induce cardiovascular system effects. Ozone induces lung injury, inflammation, and impaired mucociliary clearance, with a host of associated biochemical changes all leading to increased lung epithelial permeability. As discussed in section 3.2.1.3, the generation of lipid ozonation products and reactive oxygen species in lung tissues can influence pulmonary hemodynamics, and ultimately the cardiovascular system.

Other potential mechanisms by which O_3 exposure may be associated with cardiovascular disease outcomes have been described. Laboratory animals exposed to relatively high O_3 concentrations ($\geq$ 0.5 ppm) demonstrate tissue edema in the heart and lungs. Ozone-induced changes in heart rate, edema of heart tissue, and increased tissue and serum levels of ANF found with 8-h 0.5 ppm O_3 exposure in animal toxicology studies (Vesely et al., 1994a,b,c) also raise the possibility of potential cardiovascular effects of acute ambient O_3 exposures.

Animal toxicology studies have found both transient and persistent ventilatory responses with and without progressive decreases in heart rate (Arito et al., 1997). Observations of O_3-induced vasoconstriction in a controlled human exposure study by Brook et al. (2002) suggests another possible mechanism for O_3-related exacerbations of preexisting cardiovascular disease. One controlled human study (Gong et al., 1998) evaluated potential cardiovascular health effects of O_3 exposure. The overall results did not indicate acute cardiovascular effects of O_3 in either the hypertensive or control subjects. The authors observed an increase in rate-pressure product and heart rate, a decrement for FEV_1, and a >10 mm Hg increase in the alveolar/arterial pressure

difference for O_2 following O_3 exposure. The mechanism for the decrease in arterial oxygen (O_2) tension study could be due to an O_3-induced ventilation-perfusion mismatch. Foster et al. (1993) demonstrated that even in relatively young healthy adults, O_3 exposure can cause ventilation to shift away from the well-perfused basal lung. This effect of O_3 on ventilation distribution may persist beyond 24-hr post-exposure (Foster et al., 1997). These findings suggest that O_3 may exert cardiovascular effects indirectly by impairing alveolar-arterial O_2 transfer and potentially reducing O_2 supply to the myocardium. Ozone exposure may increase myocardial work and impair pulmonary gas exchange to a degree that could perhaps be clinically important in persons with significant preexisting cardiovascular impairment.

As noted in section 3.3.1.3, a limited number of new epidemiological studies have reported associations between short-term O_3 exposure and effects on the cardiovascular system. Among these studies, three were population-based and involved relatively large cohorts. Two studies, the ARIC (Liao at al., 2004) and the NAS (Parks et al., 2005) evaluated associations between O_3 and HRV. The other study, MONICA (Ruidavets et al., 2005) evaluated the association between O_3 levels and the relative risk of MI. Such studies may offer more informative results based on their large subject-pool and design. Results from these three studies were suggestive of an association between O_3 exposure and the cardiovascular endpoints studies. In other recent studies on incidence of myocardial infarction and some more subtle cardiovascular health endpoints, such as changes in heart rate variability or cardiac arrhythmia, some but not all studies reported associations with short-term exposure to O_3 (CD, section 7.2.7.1). From these studies, the CD concludes that the "current evidence is rather limited but suggestive of a potential effect on HRV, ventricular arrhythmias, and MI incidence" (CD, p. 7-65).

An increasing number of studies have evaluated the association between O_3 exposure and cardiovascular hospital admissions. As shown in Figure 7-13 and discussed in section 7.3.4 of the CD, many reported negative or inconsistent associations, whereas other studies, especially those that examined the relationship when O_3 exposures were higher, have found positive and robust associations between O_3 and cardiovascular hospital admissions. The CD finds that the overall evidence from these studies remains inconclusive regarding the effect of O_3 on cardiovascular hospitalizations (CD, p. 7-83).

The CD notes that the suggestive positive epidemiologic findings of O_3 exposure on cardiac autonomic control, including effects on HRV, ventricular arrhythmias and MI, and reported associations between O_3 exposure and cardiovascular hospitalizations in the warm season gain credibility and scientific support from the results of experimental animal toxicology and human clinical studies, which are indicative of plausible pathways by which O_3 may exert cardiovascular effects (CD, Section 8.6.1).

3.5.4 Coherence and Plausibility of Effects Related to Long-Term O_3 Exposure

Human chamber studies can not evaluate effects of long-term exposures to O_3; there is some evidence available from toxicological studies. While early animal toxicology studies of long-term O_3 exposures were conducted using continuous exposures, more recent studies have focused on exposures which mimic diurnal and seasonal patterns and more realistic O_3 exposure levels (CD, p. 8-50). Studies of monkeys that compared these two exposure scenarios found increased airway pathology only with the latter design. Persistent and irreversible effects reported in chronic animal toxicology studies suggest that additional complementary human data are needed from epidemiologic studies (CD, p. 8-50).

A long-term study of infant rhesus monkeys exposed to simulated seasonal O_3 (0.5 ppm , 8 hr/day for 5 days every 14 days for 11 episodes) reported remodeling of the distal airways, abnormalities in tracheal basement membrane, accumulation of eosinophils in conducting airways, and decrements in airway innervation. Another long-term exposure study of monkeys exposed to 0.61 ppm O_3 for a year and studies of rats exposed for 20 months (0.5-1.0 ppm O_3 for 6 hr/day) reported increased deposition of collagen and thickening of the CAR, suggestive of irreversible long-term O_3 impacts on the lungs. Although some earlier seasonal exposure studies of rats reported small, but significant, decrements in lung function consistent with focal fibrogenesis in the proximal alveolar region, other chronic exposure studies with exposures of 0.5 to 1.0 ppm O_3 report epithelial hyperplasia that disappears in a few days.

At this time, there is limited evidence from human studies for long-term O_3-induced effects on lung function. As discussed in section 8.6.2 of the CD, previous epidemiological studies have provided only inconclusive evidence for either mortality or morbidity effects of long-term O_3 exposure. The CD observes that the inconsistency in findings may be due to a lack of precise exposure information, the possibility of selection bias, and the difficulty of controlling for confounders (CD, p. 8-50). Several new longitudinal epidemiology studies have evaluated associations between long-term O_3 exposures and morbidity and mortality and suggest that these long-term exposures may be related to changes in lung function in children; however, little evidence is available to support a relationship between chronic O_3 exposure and mortality or lung cancer incidence (CD, p. 8-50).

The CD (p. 8-51) concludes that evidence from animal toxicology studies strongly suggests that chronic O_3 exposure is capable of damaging the distal airways and proximal alveoli, resulting in lung tissue remodeling leading to apparent irreversible changes. Such structural changes and compromised pulmonary function caused by persistent inflammation may exacerbate the progression and development of chronic lung disease. Together with the limited evidence available from epidemiological studies, these findings offer some insight into potential biological mechanisms for suggested associations between long-term or seasonal exposures to O_3

and reduced lung function development in children which have been observed in epidemiologic studies (CD, p. 8-51).

3.5.5 Coherence and Plausibility of Short-Term Mortality-Related Health Endpoints

An extensive epidemiological literature on air pollution related mortality risk estimates from the U.S., Canada, and Europe is discussed in the CD (sections 7.4 and 8.6.3). These single- and multi-city mortality studies coupled with meta-analyses generally indicate associations between acute O_3 exposure and elevated risk for all-cause mortality, even after adjustment for the influence of season and PM. Several single-city studies that specifically evaluated the relationship between O_3 exposure and cardiopulmonary mortality also reported results suggestive of a positive association (CD, p. 8-51). These mortality studies suggest a pattern of effects for causality that have biologically plausible explanations, but our knowledge regarding potential underlying mechanisms is very limited at this time and requires further research. Most of the physiological and biochemical parameters investigated in human and animal studies suggest that O_3-induced biochemical effects are relatively transient and attenuate over time. The CD (p. 8-52) hypothesizes a generic pathway of O_3-induced lung damage, potentially involving oxidative lung damage with subsequent inflammation and/or decline in lung function leading to respiratory distress in some sensitive population groups (e.g., asthmatics), or other plausible pathways noted below that may lead to O_3-related contributions to cardiovascular effects that ultimately increase risk of mortality.

The third National Health and Nutrition Examination Follow-up data analysis indicates that about 20% of the adult population has reduced FEV_1 values, suggesting impaired lung function in some portion of the population. Most of these individuals have COPD, asthma or fibrotic lung disease (Manino et al., 2003), which are associated with persistent low-grade inflammation. Furthermore, patients with COPD are at increased risk for cardiovascular disease. Also, lung disease with underlying inflammation may be linked to low-grade systemic inflammation associated with atherosclerosis, independent of cigarette smoking (CD, p. 8-52). Lung function decrements in persons with cardiopulmonary disease have been associated with inflammatory markers, such as C-reactive protein (CRP) in the blood. At a population level it has been found that individuals with the lowest FEV_1 values have the highest levels of CRP, and those with the highest FEV_1 values have the lowest CRP levels (Manino et al., 2003; Sin and Man, 2003). This complex series of physiological and biochemical reactions following O_3 exposure may tilt the biological homeostasis mechanisms which could lead to adverse health effects in people with compromised cardiopulmonary systems.

Of much interest are several other types of newly available data that support reasonable hypotheses that may help to explain the findings of O_3-related increases in cardiovascular mortality observed in some epidemiological studies. These include the direct effect of O_3 on increasing PAF in lung tissue that can then enter the general circulation and possibly contribute to increased risk of blood clot formation and the consequent increased risk of MI, cerebrovascular events (stroke), or associated cardiovascular-related mortality. Ozone reactions with cholesterol in lung surfactant to form epoxides and oxysterols that are cytotoxic to lung and heart muscles and that contribute to atherosclerotic plaque formation in arterial walls represent another potential pathway. Stimulation of airway irritant receptors may lead to increases in tissue and serum levels of ANF, changes in heart rate, and edema of heart tissue. A few new field and panel studies of human adults have reported associations between ambient O_3 concentrations and changes in cardiac autonomic control (e.g., HRV, ventricular arrhythmias, and MI). These represent plausible pathways that may lead to O_3-related contributions to cardiovascular effects that ultimately increase the risk of mortality.

In addition, O_3-induced increases in lung permeability allow more ready entry for inhaled PM into the blood stream, and O_3 exposure may increase the risk of PM-related cardiovascular effects. Furthermore, increased ambient O_3 levels contribute to ultrafine PM formation in the ambient air and indoor environments. Thus, the contributions of elevated ambient O_3 concentrations to ultrafine PM formation and human exposure, along with the enhanced uptake of inhaled fine particles, consequently may contribute to exacerbation of PM-induced cardiovascular effects in addition to those more directly induced by O_3 (CD, p. 8-53).

3.6 OZONE-RELATED IMPACTS ON PUBLIC HEALTH

The following discussion draws from section 8.7 of the CD to characterize factors which modify responsiveness to O_3, subpopulations potentially at risk for O_3-related health effects, and potential public health impacts associated with exposure to ambient O_3. Providing appropriate protection of public health requires that a distinction be made between those effects that are considered adverse health effects and those that are not adverse. What constitutes an adverse health effect depends not only on the type and magnitude of effect but also on the population group being affected. While some changes in healthy individuals would not be considered adverse, similar changes in susceptible individuals would be seen as adverse. In order to estimate the potential public health impact, it is important to consider both the susceptible subpopulations for O_3 exposure and the definition of adversity for O_3 health effects.

3.6.1 Factors that Modify Responsiveness to Ozone

There are numerous factors that can modify individual responsiveness to O_3. These include: influence of physical activity; age; gender and hormonal influences; racial, ethnic and socioeconomic status (SES) factors; environmental factors; and oxidant-antioxidant balance. These factors are discussed in more detail in section 6.5 of the CD.

It is well established that physical activity increases an individual's minute ventilation and will thus increase the dose of O_3 inhaled (CD, section 6.5.4). Increased physical activity results in deeper penetration of O_3 into more distal regions of the lungs, which are more sensitive to acute O_3 response and injury. This will result in greater lung function decrements for acute exposures of individuals during increased physical activity. Research has shown that respiratory effects are observed at lower O_3 concentrations if the level of exertion is increased and/or duration of exposure and exertion are extended. Predicted O_3-induced decrements in lung function have been shown to be a function of exposure duration and exercise level for healthy, young adults (McDonnell et al., 1997).

Most of the studies investigating the influence of age have used lung function decrements and symptoms as measures of response. For healthy adults, lung function and symptom responses to O_3 decline as age increases. The rate of decline in O_3 responsiveness appears greater in those 18 to 35 years old compared to those 35 to 55 years old, while there is very little change after age 55. In one study (Seal et al., 1996) analyzing a large data set, a 5.4% decrement in FEV_1 was estimated for 20 year old individuals exposed to 0.12 ppm O_3, whereas similar exposure of 35 year old individuals were estimated to have a 2.6% decrement. While healthy children tend not to report respiratory symptoms when exposed to low levels of O_3, for subjects 18 to 36 years old symptom responses induced by O_3 tend to decrease with increasing age (McDonnell et al., 1999).

Limited evidence of gender differences in response to O_3 exposure has suggested that females may be predisposed to a greater susceptibility to O_3. Lower plasma and NL fluid levels of the most prevalent antioxidant, uric acid, in females relative to males may be a contributing factor (Housley et al., 1996). Consequently, reduced removal of O_3 in the upper airways may promote deeper penetration. However, most of the evidence on gender differences appears to be equivocal, with one study (Hazucha et al., 2003) suggesting that physiological responses of young healthy males and females may be comparable (CD, section 6.5.2).

A few studies have suggested that ethnic minorities might be more responsive to O_3 than Caucasian population groups (CD, section 6.5.3). This may be more the result of a lack of adequate health care and socioeconomic status than any differences in sensitivity to O_3. The limited data available, which have investigated the influence of race, ethnic or other related factors on responsiveness to O_3, prevent drawing any clear conclusions at this time.

Few human studies have examined the potential influence of environmental factors such as the sensitivity of individuals who voluntarily smoke tobacco (i.e., smokers) and the effect of high temperatures. New controlled human exposure studies have confirmed that smokers are less responsive to O_3 than nonsmokers; however, time course of development and recovery of these effects, as well as reproducibility, was not different from nonsmokers (CD, section 6.5.5). Influence of ambient temperature on pulmonary effects induced by O_3 has been studied very little, but additive effects of heat and O_3 exposure have been reported.

Antioxidants, which scavenge free radicals and limit lipid peroxidation in the ELF, are the first line of defense against oxidative stress. Ozone exposure leads to absorption of O_3 in the ELF with subsequent depletion of ELF antioxidant level in the nasal ELF, but concentration and antioxidant enzyme activity in ELF or plasma do not appear related to O_3 responsiveness (CD, section 6.5.6). Controlled studies of dietary antioxidant supplements have shown some protective effects on lung function decrements but not on symptoms and airway inflammatory responses. Dietary antioxidant supplements have provided some protection to asthmatics by attenuating post-exposure airway hyperresponsiveness. Animal studies have also supported the protective effects of ELF antioxidants.

3.6.2 Susceptible Population Groups

Several characteristics that may increase the extent to which a population group shows sensitivity to O_3 have been discussed in the CD, in the sections on clinical studies in Chapter 6, epidemiological studies in Chapter 7, and in the integrated assessment in Chapter 8; this section will draw on all of these. The characteristics that likely increase susceptibility to O_3 are based on: (1) activity patterns; (2) lung disease; (3) age; and (4) biological responsiveness to O_3. Other groups that might have enhanced sensitivity to O_3, but for which there is currently very little evidence, include: people with heart disease; groups based on race, gender and socioeconomic status; and those with nutritional deficiencies.

3.6.2.1 Active People

A large group of individuals at risk from O_3 exposure consists of outdoor workers and children, adolescents, and adults who engage in outdoor activities involving exertion or exercise during summer daylight hours when ambient O_3 concentrations tend to be higher. This conclusion is based on a large number of controlled-exposure human studies and several epidemiologic field/panel studies which have been conducted with healthy children and adults and those with preexisting respiratory diseases (CD, sections 6.2 and 6.3). The controlled human exposure studies show a clear O_3 exposure-response relationship with increasing spirometric and symptomatic response as exercise level increases. Furthermore, O_3-induced response increases as time of exposure increases. Studies of outdoor workers and others who

participate in outdoor activities indicate that extended exposures to O_3 at elevated exertion levels can produce marked effects on lung function.

The effects of O_3 on the respiratory health of outdoor workers and others who participate in outdoor activities have been investigated in several recent epidemiologic studies. These individuals may experience increased vulnerability for O_3 health effects, because they are typically exposed to high doses of O_3 as they spend long hours outdoors often at elevated exertion levels. In a group of berry pickers in Fraser Valley, Canada, decrements in lung function (~5% decrease in FEV_1 per 40 ppb increase in 1-hr max O_3) were associated with acute exposure to relatively low concentrations of O_3 (Brauer et al., 1996). The mean ambient 1-hr max O_3 was 40.3 ppb (SD 15.2) over the study period of June to August 1993. The berry pickers worked outdoors for an average of 11 hr at elevated heart rates (on average, 36% higher than resting levels). These results indicate that extended exposures to O_3 at elevated exertion levels can produce marked effects on lung function among outdoor workers.

Höppe et al. (1995) examined forestry workers for O_3-related changes in pulmonary function in Munich, Germany. Ventilation rates, estimated from their average activity levels, were elevated. When comparisons were made between high O_3 days (mean ½-hr max O_3 of 64 ppb) and low O_3 days (mean ½-hr max O_3 of 32 ppb), 59% of the forestry workers experienced a notable decrement in lung function (i.e., at least a 20% increase in specific airway resistance or at least a 10% decrease in FEV_1, FVC, or PEF) on high O_3 days. None experienced improved lung function. This study also examined athletes following a 2-hr outdoor training period in the afternoon yielding a ventilation rate double the estimate for the forestry workers. Though a significant association between ambient O_3 levels and decrements in FEV_1 was observed overall, a smaller percentage of the athletes (14%) experienced a notable decrement in lung function on high O_3 days compared to the forestry workers; and 19% of the athletes actually showed an improvement.

A large field study by Korrick et al. (1998) examined the effects of multi-hour O_3 exposures (on average, 8 hr) on adults hiking outdoors on Mount Washington, in NH. The mean of the hourly O_3 concentrations during the hike was 40 ppb (range 21-74). After the hike, all subjects combined experienced a relatively small mean decline in FEV_1 (1.5% decrease per 30 ppb increase in mean hourly O_3 concentrations) during the hike. Ozone-related changes in lung function parameters were estimated. Stratifying the data by hiking duration indicated that individuals who hiked 8 to 12 hr experienced a 2-fold greater decline in FEV_1 versus those only hiking 2 to 8 hr.

Results from the above field studies are consistent with those from earlier summer camp studies (Avol et al., 1990; Higgins et al., 1990; Raizenne et al., 1987, 1989; Spektor et al., 1988,

1991), which also observed strong associations between acute O_3 exposure and decrements in lung function among children who spent long hours outdoors. In a recent analysis by the Southern California Children's Health Study, a total of 3,535 initially nonasthmatic children (ages 9 to 16 years at enrollment) were followed for up to 5 years to identify new-onset asthma cases associated with higher long-term ambient O_3 concentrations (McConnell et al., 2002). Communities were stratified by pollution levels, with six high-O_3 communities (mean 1-hr max O_3 of 75.4 ppb [SD 6.8] over four years) and six low-O_3 communities (mean 50.1 ppb [SD 11.0]). In the combined analysis using all children, asthma risk was not found to be higher for residents of the six high-O_3 communities versus those from the six low-O_3 communities. However, within the high-O_3 communities, asthma risk was more than 3 times greater for children who played three or more sports versus those who played no sports, an association not observed in the low-O_3 communities. Therefore, among children repeatedly exposed to higher O_3 levels, increased exertion outdoors (and resulting increased O_3 dose) was associated with excess asthma risk.

These field studies with subjects at elevated exertion levels support the extensive evidence derived from controlled human exposure studies. The majority of human chamber studies have examined the effects of O_3 exposure in subjects performing continuous or intermittent exercise for variable periods of time. Significant O_3-induced respiratory responses have been observed in clinical studies of exercising individuals. The epidemiologic studies discussed above also indicate that prolonged exposure periods, combined with elevated levels of exertion or exercise, may magnify O_3 effects on lung function. Thus, outdoor workers and others who participate in higher exertion activities outdoors during the time of day when high peak O_3 concentrations occur appear to be particularly vulnerable to O_3 effects on respiratory health. Although these studies show a wide variability of response and sensitivity among subjects and the factors contributing to this variability continue to be incompletely understood, the effect of increased exertion is consistent. It should be noted that this wide variability of response and sensitivity among subjects may be in part due to the wide range of other highly reactive photochemical oxidants coexisting with O_3 in the ambient air.

3.6.2.2 People with Lung Disease

People with preexisting pulmonary disease are likely to be among those at increased risk from O_3 exposure. Altered physiological, morphological and biochemical states typical of respiratory diseases like asthma, COPD and chronic bronchitis may render people sensitive to additional oxidative burden induced by O_3 exposure. The new results from controlled exposure and epidemiologic studies continue to indicate that asthmatics are a sensitive subpopulation for O_3 health effects.

A number of epidemiological studies have been conducted using asthmatic study populations. The majority of epidemiological panel studies that evaluated respiratory symptoms and medication use related to O_3 exposures focused on children. These studies suggest that O_3 exposure may be associated with increased respiratory symptoms and medication use in children with asthma. Other reported effects include respiratory symptoms, lung function decrements, and ED visits, as discussed in the CD (section 7.6.7.1). Strong evidence from a large multi-city study (Mortimer et al., 2002), along with support from several single-city studies suggest that O_3 exposure may be associated with increased respiratory symptoms and medication use in children with asthma. With regard to ambient O_3 levels and increased hospital admissions and ED visits for asthma and other respiratory causes, strong and consistent evidence establishes a correlation between O_3 exposure and increased exacerbations of preexisting respiratory disease for 1-hr maximum O_3 concentrations <0.12 ppm. Several hospital admission and ED visit studies in the U.S. (Peel et al., 2005), Canada (Burnett et al., 1997a; Anderson et al., 1997), and Europe (Anderson et al., 1997) have reported positive associations between increase in O_3 and increased risk of ED visits and hospital admissions, especially during the warm season.

Several clinical studies reviewed in the 1996 CD on atopic and asthmatic subjects had suggested but not clearly demonstrated enhanced responsiveness to acute O_3 exposure compared to healthy subjects. The majority of the newer studies reviewed in Chapter 6 of the CD indicate that asthmatics are as sensitive as, if not more sensitive than, normal subjects in manifesting induced pulmonary function decrements.

Ozone-induced increases in neutrophils, protein, and IL-8 were found to be significantly higher in the BAL fluid from asthmatics compared to healthy subjects, suggesting mechanisms for the increased sensitivity of asthmatics. Similarly, subjects with allergic asthma exhibited increased airway responsiveness to inhaled allergens upon acute O_3 exposure. Asthmatics present a differential response profile for cellular, molecular, and biochemical parameters (CD, Figure 8-1) that are altered in response to acute O_3 exposure. Increases in O_3-induced nonspecific airway responsiveness incidence and duration could have important clinical implications for asthmatics.

Bronchial constriction following provocation with allergens presents a two-phase response. The early response is mediated by release of histamine and leukotrienes that leads to contraction of smooth muscle cells in the bronchi, narrowing the lumen and decreasing the airflow. In asthmatics, these mediators also cause accumulation of eosinophils, followed by production of mucus and a late-phase bronchial constriction and reduced airflow. Holz et al. (2002) reported an early phase response in subjects with rhinitis after a consecutive 4-day exposure to 0.125 ppm O_3 that resulted in a clinically relevant (>20%) decrease in FEV_1. Allergen challenge in mild asthmatics 24 hr before exposure to 0.27 ppm O_3 for 2 hr resulted in

significantly increased eosinophil counts in samples of respiratory tract lining fluid, obtained by sputum induction, compared to results in healthy subjects (Vagaggini et al., 2002). Epithelial cells from mucosal biopsies of allergic asthmatics indicated significant increases in the expression of IL-5, IL-8 and GM-CSF, suggesting increased neutrophilic inflammation compared to healthy subjects (Bosson et al., 2003).

Several human exposure studies have shown differences between asthmatics and healthy human subjects with regard to PMN influx in BAL fluid. In vitro studies (Schierhorn et al., 1999) of nasal mucosal biopsies from atopic and nonatopic subjects exposed to 0.1 ppm O_3 found significant differences in release of IL-4, IL-6, IL-8, and TNF-α. Another study by Schierhorn et al. (2002) found significant differences in the O_3-induced release of the neuropeptides neurokinin A and substance P for allergic patients in comparison to nonallergic controls, suggesting increased activation of sensory nerves by O_3 in the allergic tissues. Another study by Bayram et al. (2002) using in vitro culture of bronchial epithelial cells recovered from atopic and nonatopic asthmatics also found significant increases in epithelial permeability in response to O_3 exposure. In addition, some controlled human O_3 exposure studies in asthmatics (Hiltermann et al., 1999; Scannell et al., 1996) reported increased secretion of IL-8, suggesting increased neutrophilic inflammation. Two studies (Jörres et al., 1996; Holz et al., 2002) observed increased airway responsiveness to repeated daily O_3 exposure to bronchial allergen challenge in subjects with preexisting allergic airway disease.

Newly available reports from controlled human exposure studies (see Chapter 6 in the CD) utilized subjects with preexisting cardiopulmonary diseases such as COPD, asthma, allergic rhinitis, and hypertension. The data generated from these studies that evaluated pulmonary function changes in spirometry did not find clear differences between filtered air and O_3 exposure in COPD subjects. However, the new data on airway responsiveness, inflammation, and various molecular markers of inflammation and bronchoconstriction indicate that people with atopic asthma and allergic rhinitis comprise susceptible groups for O_3-induced adverse health effects.

Although controlled human exposure studies have not found evidence of larger spirometric changes in people with COPD relative to healthy subjects, this may be due to the fact that most people with COPD are older adults who would not be expected to have such changes based on their age. However, in Section 8.7.1, the CD notes that new epidemiological evidence indicates that people with COPD may be more likely to experience other effects, including emergency room visits, hospital admissions, or premature mortality. For example, results from an analysis of five European cities indicated strong and consistent O_3 effects on unscheduled respiratory hospital admissions, including COPD (Anderson et al., 1997). Also, an analysis of a 9-year data set for the whole population of the Netherlands provided risk estimates for more specific causes of mortality, including COPD (Hoek et al., 2000, 2001; reanalysis Hoek, 2003); a

positive, but nonsignificant, excess risk of COPD-related mortality was found to be associated with short-term O_3 concentrations. Moreover, as indicated by Gong et al. (1998), the effects of O_3 exposure on alveolar-arterial oxygen gradients may be more pronounced in patients with preexisting obstructive lung diseases. Relative to healthy elderly subjects, COPD patients have reduced gas exchange and low SaO_2. Any inflammatory or edematous responses due to O_3 delivered to the well-ventilated regions of the COPD lung could further inhibit gas exchange and reduce oxygen saturation. In addition, O_3-induced vasoconstriction could also acutely induce pulmonary hypertension. Inducing pulmonary vasoconstriction and hypertension in these patients would perhaps worsen their condition, especially if their right ventricular function was already compromised (CD, Section 6.10).

3.6.2.3 Children and Older Adults

Supporting evidence exists for heterogeneity in the effects of O_3 by age. As discussed in section 6.5.1 of the CD, children, adolescents, and young adults (<18 yrs of age) appear, on average, to have nearly equivalent spirometric responses to O_3, but have greater responses than middle-aged and older adults when exposed to comparable O_3 doses. Symptomatic responses to O_3 exposure, however, do not appear to occur in healthy children, but are observed in asthmatic children, particularly those who use maintenance medications. For adults (>17 yrs of age) symptoms gradually decrease with increasing age. In contrast to young adults, the diminished symptomatic responses in children and symptomatic and spirometric responses in the elderly may put them at an increased risk for continued exposure.

As described in the section 7.6.7.2 of the CD, many epidemiological field studies focused on the effect of O_3 on the respiratory health of school children. In general, children experienced decrements in pulmonary function parameters, including PEF, FEV_1, and FVC. Increases in respiratory symptoms and asthma medication use were also observed in asthmatic children. In one German study, children with and without asthma were found to be particularly susceptible to O_3 effects on lung function. Approximately 20% of the children, both with and without asthma, experienced a greater than 10% change in FEV_1, compared to only 5% of the elderly population and athletes (Höppe et al., 2003).

The American Academy of Pediatrics (2004) notes that children and infants are among the population groups most susceptible to many air pollutants, including O_3. This is in part because their lungs are still developing. For example, eighty percent of alveoli are formed after birth, and changes in lung development continue through adolescence (Dietert et al., 2000). Children are also likely to spend more time outdoors than adults do, which results in increased exposure to air pollutants (Wiley et al., 1991a,b). Moreover, children have high minute

ventilation rates and high levels of physical activity which also increases their dose (Plunkett et al., 1992).

Several mortality studies have investigated age-related differences in O_3 effects. Among the studies that observed positive associations between O_3 and mortality, a comparison of all age or younger age (≤65 years of age) O_3-mortality effect estimates to that of the elderly population (>65 years) indicates that, in general, the elderly population is more susceptible to O_3 mortality effects (Borja-Aburto et al., 1997; Bremner et al., 1999; Gouveia and Fletcher, 2000; O'Neill et al., 2004; Simpson et al., 1997; Sartor et al., 1995; Sunyer et al., 2002). For example, a study by Gouveia and Fletcher (2000) examined the O_3-mortality effect by age in São Paulo, Brazil. Among all ages, O_3 was associated with a 0.6% excess risk in all cause mortality per 40 ppb increase in 1-hr max O_3. In comparison, in the elderly population, the O_3-mortality risk estimate was nearly threefold greater, at 1.7%. Similarly, a Mexico City study found that O_3-mortality effect estimates were 1.3% and 2.8% per 20 ppb increase in 24-hr average O_3 concentration in all ages and the elderly, respectively (O'Neill et al., 2004).

The meta-analysis by Bell et al. (2005) found a larger effect estimate for the elderly (2.92% per 20 ppb increase in 24-hr average O_3) than for all ages (1.75%). In the large U.S. 95 communities study (Bell et al., 2004), effect estimates were slightly higher for those aged 65 to 74 years, 1.40% excess risk per 20 ppb increase in 24-hr average O_3, compared to individuals less than 65 years and 75 years or greater, 1.00% and 1.04%, respectively, using a constrained distributed 7-day lag model. Bell et al. (2004) note that despite similar effects estimates, the absolute effect of O_3 is substantially greater in the elderly population due to the higher underlying mortality rates, which lead to a larger number of extra deaths for the elderly compared to the general population. The CD concludes that the elderly population (>65 years of age) appear to be at greater risk of O_3-related mortality and hospitalizations compared to all ages or younger populations (CD, p. 7-177).

The CD notes that, collectively, there is supporting evidence of age-related differences in susceptibility to O_3 lung function effects. The elderly population (>65 years of age) appear to be at increased risk of O_3-related mortality and hospitalizations, and children (<18 years of age) experience other potentially adverse respiratory health outcomes with increased O_3 exposure (CD, section 7.6.7.2).

3.6.2.4 People with Increased Responsiveness to Ozone

New animal toxicology studies using various strains of mice and rats have identified O_3-sensitive and resistant strains and illustrated the importance of genetic background in determining O_3 susceptibility (CD, section 8.7.4). Using subacute low exposure regimen (0.3 ppm O_3, 48h) studies on inbred strains that have been designated as inflammation prone or

resistant, Kleeberger et al., (1997) identified the pro-inflammatory cytokine gene, *Tnf*-α, as a susceptibility gene. Further characterization of this model indicated a role for TNF receptors (TNFR1, TNFR2) in O_3-induced pulmonary epithelial injury and inflammation (Cho et al., 2001). Studies on five inbred strains of mouse with differing response to O_3 exposure (acute high dose or low dose continuous exposure for 3 days), reported a protective role for clara cell secretory protein (CCSP) against O_3-induced oxidative damage (Broeckaert et al., 2003; Wattiez et al., 2003). The role for these genes and/or their orthologs in human susceptibility to O_3 exposure is yet to be examined.

Apart from age at the time of exposure, controlled human exposure studies have also indicated a high degree of interindividual variability in some of the pulmonary physiological parameters. Recent studies by David et al. (2003) and Romieu et al. (2004) reported a role for genetic polymorphism in antioxidant enzymes and genes involved in inflammation to modulate pulmonary function and inflammatory responses to O_3 exposure. Similar to mouse studies referred above, polymorphism in *Tnf*-α has been implicated in O_3-induced lung function changes in healthy, mild asthmatics and individuals with rhinitis. These observations suggest a potential role for these markers in the innate susceptibility to O_3, however, the validity of these markers and their relevance in the context of prediction to population studies needs additional experimentation.

Biochemical and molecular parameters extensively evaluated in these experiments were used to identify specific loci on the chromosomes and, in some cases, to relate the differential expression of specific genes to biochemical and physiological differences observed among these species. Utilizing O_3-sensitive and O_3-resistant species, it has been possible to identify the involvement of AHR and inflammation processes in O_3 susceptibility. However, most of these studies were carried out using relatively high doses of O_3, making the relevance of these studies questionable in human health effects assessment. The molecular parameters identified in these studies may serve as useful biomarkers with the availability of suitable technologies and, ultimately, can likely be integrated with epidemiological studies. Interindividual differences in O_3 responsiveness have been observed across a spectrum of symptoms and lung function responses do not yet allow identification of important underlying factors, except a significant role for age.

3.6.2.5 Other Population Groups

There is limited, new evidence supporting associations between short-term O_3 exposures and a range of effects on the cardiovascular system. Some but not all, epidemiological studies have reported associations between short-term O_3 exposures and the incidence of myocardial infarction and more subtle cardiovascular health endpoints, such as changes in heart rate

variability and cardiac arrhythmia. Others have reported associations with hospitalization or ED visits for cardiovascular diseases, although the results across the studies are not consistent. Studies also report associations between short-term O_3 exposure and mortality from cardiovascular or cardiopulmonary causes. The CD concludes that current cardiac physiologic effects evidence from some field studies is rather limited but supportive of a potential effect of short-term O_3 exposure and HRV, cardiac arrhythmia, and MI incidence (CD, p. 7-65). In the CD's evaluation of studies of hospital admissions for cardiovascular disease (CD, section 7.3.4), it is concluded that evidence from this growing group of studies is generally inconclusive regarding an association with O_3 in studies conducted during the warm season (CD, p. 7-83). This body of evidence suggests that people with heart disease may be at increased risk from short-term exposures to O_3; however, more evidence is needed to conclude that people with heart disease are a susceptible population.

Other groups that might have enhanced sensitivity to O_3, but for which there is currently very little evidence, include groups based on race, gender and socioeconomic status, and those with nutritional deficiencies, as discussed in section 3.6.1 which presents factors which modify responsiveness to O_3.

3.6.3 What Constitutes an Adverse Health Impact from Ozone Exposure?

In making judgments as to when various O_3-related effects become regarded as adverse to the health of individuals, in previous NAAQS reviews staff has relied upon the guidelines published by the American Thoracic Society (ATS) and the advice of CASAC. While recognizing that perceptions of "medical significance" and "normal activity" may differ among physicians, lung physiologists and experimental subjects, the ATS (1985) defined adverse respiratory health effects as "medically significant physiologic changes generally evidenced by one or more of the following: (1) interference with the normal activity of the affected person or persons, (2) episodic respiratory illness, (3) incapacitating illness, (4) permanent respiratory injury, and/or (5) progressive respiratory dysfunction."

During the 1997 review, it was concluded that there was evidence of causal associations from controlled human exposure studies for effects in the first of these five ATS-defined categories, evidence of statistically significant associations from epidemiological studies for effects in the second and third categories, and evidence from animal toxicology studies, which could be extrapolated to humans only with a significant degree of uncertainty, for the last two categories. For the current review, the evidence of O_3-related effects is stronger across all the categories. For ethical reasons, clear causal evidence from controlled human exposure studies still covers only effects in the first category. However, for this review there are results from epidemiological studies, upon which to base judgments about adversity, for effects in all of the

categories. Statistically significant and robust associations have been reported in epidemiology studies falling into the second and third categories. These more serious effects include respiratory illness that may require medication (e.g., asthma), but not necessarily hospitalization, as well as respiratory hospital admissions and ED visits for respiratory causes. Less conclusive, but still positive associations have been reported for school absences and cardiovascular hospital admissions. Human health effects for which associations have been suggested through evidence from epidemiological and animal toxicology studies, but have not been conclusively demonstrated still fall primarily into the last two categories. In the last review of the O_3 standard, evidence for these more serious effects came from studies of effects in laboratory animals. Evidence from animal studies evaluated in this CD strongly suggests that O_3 is capable of damaging the distal airways and proximal alveoli, resulting in lung tissue remodeling leading to apparently irreversible changes. Recent advancements of dosimetry modeling also provide a better basis for extrapolation from animals to humans. Information from epidemiological studies provides supporting, but limited evidence of irreversible respiratory effects in humans (as described in section 6.3.3.2 below). Moreover, the CD concludes that the findings from single-city and multi-city time-series epidemiology studies and meta-analyses of these epidemiology studies support a likely causal association between short-term O_3 exposure and mortality particularly in the warm season.

While O_3 has been associated with effects that are clearly adverse, application of these guidelines, in particular to the least serious category of effects related to ambient O_3 exposures, involves judgments about which medical experts on the CASAC panel and public commenters have in the past expressed diverse views. To help frame such judgments, we have defined gradations of individual functional responses (e.g., decrements in FEV_1 and airway responsiveness) and symptomatic responses (e.g., cough, chest pain, wheeze), together with judgments as to the potential impact on individuals experiencing varying degrees of severity of these responses, that have been used in previous NAAQS reviews. These gradations and impacts are summarized in Tables 3-2 and 3-3.

For active healthy people, moderate levels of functional responses (e.g., FEV_1 decrements of $\geq$10% but < 20%, lasting up to 24 hrs) and/or moderate symptomatic responses (e.g., frequent spontaneous cough, marked discomfort on exercise or deep breath, lasting up to 24 hrs) would likely interfere with normal activity for relatively few sensitive individuals; whereas large functional responses (e.g., FEV_1 decrements $\geq$ 20%, lasting longer than 24 hrs) and/or severe symptomatic responses (e.g., persistent uncontrollable cough, severe discomfort on exercise or deep breath, lasting longer than 24 hrs) would likely interfere with normal activities

Table 3-2. Gradation of Individual Responses to Short-Term Ozone Exposure in Healthy Persons[1]

Functional Response	**None**	**Small**	**Moderate**	**Large**
FEV_1	Within normal range (±3%)	Decrements of 3 to ≤10%	Decrements of >10 but <20%	Decrements of ≥20%
Nonspecific airway responsiveness[2]	Within normal range	Increases of <100%	Increases of ≤300%	Increases of >300%
Duration of response	None	<4 hrs	>4 hrs but ≤24 hrs	>24 hrs
Symptom Response	**Normal**	**Mild**	**Moderate**	**Severe**
Cough	Infrequent cough	Cough with deep breath	Frequent spontaneous cough	Persistent uncontrollable cough
Chest pain	None	Discomfort just noticeable on exercise or deep breath	Marked discomfort on exercise or deep breath	Severe discomfort on exercise or deep breath
Duration of response	None	<4 hrs	>4 hrs but ≤24 hrs	>24 hrs
Impact of Responses	**Normal**	**Normal**	**Mild**	**Moderate**
Interference with normal activity	None	None	A few sensitive individuals choose to limit activity	Many sensitive individuals choose to limit activity

[1] This table is reproduced from the 1996 O_3 AQCD (Table 9-1, page 9-24) (U.S. Environmental Protection Agency, 1996).
[2] An increase in nonspecific airway responsiveness of 100% is equivalent to a 50% decrease in PD_{20} or PD_{100}.

Table 3-3. Gradation of Individual Responses to Short-Term Ozone Exposure in Persons with Impaired Respiratory Systems[3]

Functional Response	**None**	**Small**	**Moderate**	**Large**
FEV_1 change	Decrements of <3%	Decrements of 3 to ≤10%	Decrements of >10 but <20%	Decrements of ≥20%
Nonspecific airway responsiveness[4]	Within normal range	Increases of <100%	Increases of ≤300%	Increases of >300%
Airway resistance (SRaw)	Within normal range (±20%)	SRaw increased <100%	SRaw increased up to 200% or up to 15 cm H2O/s	SRaw increased >200% or more than 15 cm H2O/s
Duration of response	None	<4 hr	>4 hr but ≤24 hr	>24 hr
Symptom Response	**Normal**	**Mild**	**Moderate**	**Severe**
Wheeze	None	With otherwise normal breathing	With shortness of breath	Persistent with shortness of breath
Cough	Infrequent cough	Cough with deep breath	Frequent spontaneous cough	Persistent uncontrollable cough
Chest pain	None	Discomfort just noticeable on exercise or deep breath	Marked discomfort on exercise or deep breath	Severe discomfort on exercise or deep breath
Duration of response	None	< 4 hr	>4 hr but ≤24 hr	>24 hr
Impact of Responses	**Normal**	**Mild**	**Moderate**	**Severe**
Interference with normal activity	None	Few individuals choose to limit activity	Many individuals choose to limit activity	Most individuals choose to limit activity
Medical treatment	No change	Normal medication as needed	Increased frequency of medication use or additional medication	Physician or emergency room visit

[3] This table is reproduced from the 1996 O_3 AQCD (Table 9-1, page 9-25) (U.S. Environmental Protection Agency, 1996).

[4] An increase in nonspecific airway responsiveness of 100% is equivalent to a 50% decrease in PD_{20} or PD_{100}.

for many sensitive individuals and therefore would be considered adverse under ATS guidelines. For the purpose of estimating potentially adverse lung function decrements in active healthy people, the CASAC indicated that a focus on the mid to upper end of the range of moderate levels of functional responses is most appropriate (e.g., FEV_1 decrements ≥ 15% but < 20%)[5]. However, for people with lung disease, even moderate functional (e.g., FEV_1 decrements ≥ 10% but < 20%, lasting up to 24 hr) or symptomatic responses (e.g., frequent spontaneous cough, marked discomfort on exercise or with deep breath, wheeze accompanied by shortness of breath, lasting up to 24 hr) would likely interfere with normal activity for many individuals, and would likely result in additional and more frequent use of medication. For people with lung disease, large functional responses (e.g., FEV_1 decrements ≥ 20%, lasting longer than 24 hrs) and/or severe symptomatic responses (e.g., persistent uncontrollable cough, severe discomfort on exercise or deep breath, persistent wheeze accompanied by shortness of breath, lasting longer than 24 hrs) would likely interfere with normal activity for most individuals and would increase the likelihood that these individuals would seek medical treatment. For the purpose of estimating potentially adverse lung function decrements in people with lung disease, the CASAC indicated that a focus on the lower end of the range of moderate levels of functional responses is most appropriate (e.g., FEV_1 decrements ≥ 10%).

In judging the extent to which these impacts represent effects that should be regarded as adverse to the health status of individuals, an additional factor that has been considered in previous NAAQS reviews is whether such effects are experienced repeatedly during the course of a year or only on a single occasion. While some experts would judge single occurrences of moderate responses to be a "nuisance," especially for healthy individuals, a more general consensus view of the adversity of such moderate responses emerges as the frequency of occurrence increases. Thus it has been judged that repeated occurrences of moderate responses, even in otherwise healthy individuals, may be considered to be adverse since they could well set the stage for more serious illness (61 FR 65723). The CASAC panel in the last review expressed a consensus view that these "criteria for the determination of an adverse physiological response were reasonable" (Wolff, 1995).

In 2000, the American Thoracic Society (ATS) published an official statement on "What Constitutes an Adverse Health Effect of Air Pollution?" (ATS, 2000), which updated its earlier guidance (ATS, 1985). The revised guidance was intended to address new investigative approaches used to identify the effects of air pollution, and to reflect the concern for the impacts of air pollution on specific groups that had been expressed through the environmental justice movement.

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[5] Transcript of CASAC meeting, day 8/24/06, page 149.

The new guidance builds upon and expands the 1985 definition of adversity in several ways. There is an increased focus on quality of life measures as indicators of adversity. There is also a more specific consideration of population risk. Exposure to air pollution that increases the risk of an adverse effect to the entire population is adverse, even though it may not increase the risk of any individual to an unacceptable level. For example, a population of asthmatics could have a distribution of lung function such that no individual has a level associated with significant impairment. Exposure to air pollution could shift the distribution to lower levels that still do not bring any individual to a level that is associated with clinically relevant effects. However, this would be considered to be adverse because individuals within the population would have diminished reserve function, and therefore would be at increased risk if affected by another agent.

Of the various effects of O_3 exposure that have been studied, many would meet the ATS definition of adversity. Such effects include, for example, any detectible level of permanent lung function loss attributable to air pollution, including both reductions in lung growth or acceleration of the age-related decline of lung function; exacerbations of disease in individuals with chronic cardiopulmonary diseases; reversible loss of lung function in combination with the presence of symptoms; as well as more serious effects such as those requiring medical care including hospitalization and, obviously, mortality.

As discussed above, relatively small, reversible declines in lung function parameters may be of questionable significance in healthy people. However, a 5 to 15 % change in FEV_1 is considered to have clinical importance to asthma morbidity (ATS 1991; Lebowitz et al. 1987; Lippmann, 1988). This is in line with the view expressed by the CASAC that a focus on the lower end of the range of moderate levels of functional responses is most appropriate (e.g., FEV_1 decrements $\geq$ 10%) to estimate the risk of potentially adverse lung function responses in people with lung disease. The National Institutes of Health (1997) has stated that a PEF below 80% of a person's personal best indicates a need for continued medication use in asthmatics. In Mortimer et al. (2002), O_3 was associated with increased incidence of $\geq$ 10% declines in morning PEF as well as morning symptoms, indicating that O_3 exposure may have clinically significant effects on asthmatic children.

Reflecting new investigative approaches, the ATS statement describes the potential usefulness of research into the genetic basis for disease, including responses to environmental agents that will provide insights into the mechanistic basis for susceptibility, and provide markers of risk status. Likewise biomarkers, that are indicators of exposure, effect or susceptibility, may someday be useful in defining the point at which a response should be equated with an adverse effect. Based on concern for segments of the population that may be disproportionately exposed to environmental contaminants, or have other factors that may

increase susceptibility (e.g., genetic or nutritional factors), there was a call for increased research in these areas.

Overall, the new guidance does not fundamentally change the approach previously taken to define adversity, nor does it suggest a need at this time to change the structure or content of the tables describing gradation of severity and adversity of effects in Tables 3-2 or 3-3 above.

3.6.4 Estimation of Potential Numbers of People in At-Risk Susceptible Population Groups in the United States

Although O_3-related health risk estimates may appear to be numerically small, their significance from an overall public health perspective is affected by the large numbers of individuals in potential risk groups. Several subpopulations may be identified as having increased susceptibility or vulnerability to adverse health effects from O_3, including: older adults, children, individuals with preexisting pulmonary disease, and those with higher exposure levels, such as outdoor workers.

One consideration in the assessment of potential public health impacts is the size of various population groups that may be at increased risk for health effects associated with O_3-related air pollution exposure. Table 8-4 in the CD summarizes information on the prevalence of chronic respiratory conditions in the U.S. population in 2002 and 2003 (Dey and Bloom, 2005; Lethbridge-Çejku et al., 2004). Individuals with preexisting cardiopulmonary disease constitute a fairly large proportion of the population, with tens of millions of people included in each disease category. Of most concern here are those individuals with preexisting respiratory conditions, with approximately 11% of U.S. adults and 13% of children having been diagnosed with asthma and 6% of adults having COPD (chronic bronchitis and/or emphysema). Table 8-5 in the CD provides further information on the number of various specific respiratory conditions per 100 persons by age among the U.S. population during the mid-1990s. Asthma prevalence tends to be higher in children than adults.

In addition, subpopulations based on age group also comprise substantial segments of the population that may be potentially at risk for O_3-related health impacts. Based on U.S. census data from 2003, about 26% of the U.S. population are under 18 years of age and 12% are 65 years of age or older. Hence, large proportions of the U.S. population are included in age groups that are considered likely to have increased susceptibility and vulnerability for health effects from ambient O_3 exposure.

The health statistics data illustrate what is known as the "pyramid" of effects. At the top of the pyramid, there are approximately 2.5 millions deaths from all causes per year in the U.S. population, with about 100,000 deaths from chronic lower respiratory diseases (Kochanek et al., 2004). For respiratory health diseases, there are nearly 4 million hospital discharges per year

(DeFrances et al., 2005), 14 million ED visits (McCaig and Burt, 2005), 112 million ambulatory care visits (Woodwell and Cherry, 2004), and an estimated 700 million restricted activity days per year due to respiratory conditions (Adams et al., 1999). Combining small risk estimates with relatively large baseline levels of health outcomes can result in quite large public health impacts. Thus, even a small percentage reduction in O_3 health impacts on cardiopulmonary diseases would reflect a large number of avoided cases.

Another key input for public health impact assessment is the range of concentration response functions for various health outcomes. Epidemiologic studies have reported associations between short-term exposure to O_3 with mortality, hospitalizations for pulmonary diseases, ED visits for asthma, reduced lung function, and incidence of respiratory symptoms. Effect estimates for morbidity responses to short-term changes in O_3 tend to be larger and more variable in magnitude than those for mortality.

In addition to attribution of risks for various health outcomes related to O_3 and other copollutants, important considerations in assessing the impact of O_3 on public health include the size of population groups at risk, as well as the concentration-response relationship and potential identification of threshold levels. Taken together, based on the above information, it can be concluded that exposure to ambient O_3 likely has a significant impact on public health in the U.S.

3.7 SUMMARY AND CONCLUSIONS FOR OZONE HEALTH EFFECTS

Based on dosimetric, experimental, and epidemiological evidence assessed in the 1996 CD, a set of findings and conclusions were drawn regarding potential health effects of O_3 exposure as of 1996. These conclusions are integrated into the Summary and Conclusions for Ozone Health Effects in the 2006 CD (section 8.8). The revised CD will be referred to as the "2006 CD" in this section in order to be more easily distinguished from the "1996 CD." Section 8.8 of the 2006 CD also has summarized the main conclusions derived from the integrated analysis of animal toxicology (2006 CD, Chapter 5), human experimental (2006 CD, Chapter 6) and epidemiological (2006 CD, Chapter 7) studies that evaluated evidence of health effects associated with short-term, prolonged, and long-term exposures to O_3 alone or in combination with other pollutants commonly found in the ambient air. This section summarizes conclusions drawn from section 8.8 of the 2006 CD with respect to the health effects associated with exposure to O_3 that are most relevant to our assessment of the adequacy of the current primary O_3 standard and the identification of options to consider concerning potential alternative standards to protect public health with an adequate margin of safety.

3.7.1 Respiratory Morbidity Effects of Short-term Exposures to Ozone

In the 1996 CD, it was concluded from assessment of controlled human exposure studies that short-term O_3 exposures to O_3 concentrations of $\geq$ 0.08 ppm for 6.6 to 8 hr under moderate exertion and $\geq$ 0.12 ppm for 1 hr under heavy exertion cause decrements in lung function in children and increased lung function and respiratory symptoms in healthy adults and asthmatic individuals exposed (2006 CD, p. 8-73). Lung inflammatory responses have been observed in healthy human adults following 6.6 hr O_3 exposures as low as 0.08 ppm (2006 CD, p. 8-75). Changes in lung function, respiratory symptoms, and lung inflammatory responses occur as a function of exposure concentration, duration, and level of exertion. Such experimentally demonstrated effects were consistent with and helped support the plausibility of epidemiological findings assessed in the 1996 CD regarding daily hospital admissions and ED visits for respiratory causes.

The 1996 CD concluded that group mean data from numerous controlled human exposure and field studies of healthy subjects (18 to 45 years of age) exposed for 1 to 3 hr indicate that, in general, statistically significant pulmonary function decrements beyond the range of normal measurement variability (e.g., 3 to 5% for FEV_1) occur in subjects exposed:

- at >0.12 ppm O_3 after very heavy exercise (competitive running).
- at >0.18 ppm O_3 after heavy exercise (easy jogging),
- at >0.30 ppm O_3 after moderate exercise (brisk walking),
- at >0.37 ppm O_3 after light exercise (slow walking), and
- at >0.50 ppm O_3 when at rest.

Small group mean changes (e.g., <5%) in FEV_1 have been observed in healthy young adults at levels as low as 0.12 ppm O_3 for 1 to 3 hr exposure periods. Also, lung function decrements have been observed in children and adolescents at concentrations of 0.12 and 0.14 ppm O_3 with heavy exercise. Some individuals within a study may experience FEV_1 decrements in excess of 15% under these conditions, even when group mean decrements are less than 5%.

For exposures of healthy, young adult subjects performing moderate exercise during longer duration exposures (6 to 8 hr), 5% group mean decrements in FEV_1 were observed at

- 0.08 ppm after O_3 5.6 hr,
- 0.10 ppm after O_3 4.6 hr, and
- 0.12 ppm after O_3 3 hr.

For these same subjects, 10% group mean FEV_1 decrements were observed at 0.12 ppm O_3 after 5.6 and 6.6 hr. As in the shorter duration studies, some individuals experience changes larger than those represented by group mean changes.

The 2006 CD (section 8.8) concludes that newer meta-analyses confirmed interindividual differences in lung function decrements reported in the 1996 CD. Age-specific differences in lung function responses were also observed. Spirometric responses (i.e., decrements in lung function) in healthy adults exposed to near ambient O_3 levels typically resolve to near baseline within 4-6 hr. Meta-analyses of four controlled human exposure studies (two new and two assessed in the 1996 CD) reporting the effects of prolonged (6.6 hr) exposures to 0.08 ppm O_3 during moderate exertion on lung function in young healthy adults (M=90, F=30; mean age 23 years) indicate an absolute FEV_1 decrease of 6%, whereas FEV_1 increased by 1% following fresh air exposures. Newer studies from Adams (2002, 2006), as illustrated earlier in Figure 3-1B, demonstrate notable interindividual variability for O_3 exposure concentrations of 0.04, 0.06 and 0.08 ppm. Following a continuous exposure to 0.08 ppm O_3 during intermittent, moderate exertion, the group mean FEV_1 decrement (corrected for filtered air) was 6%, but 23 % of subjects had FEV_1 decrements of 10% or more. Following exposure to 0.06 ppm O_3, the group mean FEV_1 decrement was less than 3%, but 7% of subjects had greater than 10% FEV_1 decrements (2006 CD, p. 8-18). However, as discussed in Section 3.3.1.1.1, we note that the pre- and post-exposure data presented in the Adams (2006) study show a small (< 3%) group mean FEV_1 decrement following the 6.6-hr exposure at 0.06 ppm, which may be statistically significantly different from filtered air responses.

A few controlled human exposure studies (Adams, 2003; 2006; Hazucha et al., 1992) investigated a triangular exposure pattern at O_3 concentrations that had 6.6 to 8-hr averages between 0.08 and 0.12 ppm in order to more closely mimic typical ambient O_3 exposure patterns. Greater overall FEV_1 decrements were observed with triangular exposures compared to the constant or square-wave exposures. Furthermore, peak FEV_1 decrements observed during triangular exposures were greater than those observed during square-wave patterns. At a lower average O_3 concentration of 0.06 ppm, no temporal (i.e., hour by hour responses) differences were observed in FEV_1 decrements between square-wave and triangular exposure patterns. There was, however, a statistically significant effect of the 0.06 ppm triangular exposure on total respiratory symptoms following 5.6 and 6.6 h of exposure that was not observed for the 0.06 ppm square-wave exposure protocol. Results of these studies suggest the potential for somewhat greater effects on lung function in ambient O_3 exposure scenarios that typically involve gradually increasing daily exposure up to a peak in the late afternoon and a subsequent gradual decline (2006 CD, p. 8-19). The quantitative risk assessment, discussed below in Chapter 5, provides

estimates of the percentages of school age children likely to experience FEV_1 decrements greater than or equal to 10, 15, and 20% after 8-hr exposures to O_3 while engaged in moderate exertion.

Decrements in lung function associated with ambient O_3 levels have also been found in children attending summer camps in southern Ontario, Canada, in the northeastern U.S., and in southern California (2006 CD, p. 8-74). The U.S. multicities study by Mortimer et al. (2002) observed an association between acute O_3 exposure and the incidence of a $\geq$10% decrement in morning PEF in asthmatic children. Meta-analyses indicate that a 0.50-mL decrease in FEV_1 is associated with a 1 ppb increase in O_3 concentration. For preadolescent children exposed to 120 ppb (0.12 ppm) ambient O_3, this amounts to an average decrement of 2.4 to 3.0% in FEV_1. Similar responses are reported for exercising children and adolescents exposed to O_3 in ambient air or O_3 in purified air for 1-2 hours.

The 1996 CD concluded that an increase in the incidence of cough has been reported at O_3 concentrations as low as 0.12 ppm in healthy adults during 1 to 3 hr of exposure with very heavy exercise. Other respiratory symptoms, such as pain on deep inspiration, shortness of breath, and lower respiratory scores (i.e., a combination of several symptoms), have been observed at 0.16 ppm to 0.18 ppm O_3, 1-hr average, with heavy and very heavy exertion. Respiratory symptoms also have been observed following exposure to 0.08, 0.10 and 0.12 ppm O_3 for 6.6 hr with moderate exertion levels. Also, increases in nonspecific airway responsiveness in healthy adults at rest have been observed after 1 to 3 hr of exposures to 0.40 ppm but not to 0.20 ppm O_3; during very heavy exertion, these increases were observed at concentrations as low as 0.18 ppm but not at 0.12 ppm O_3. Increases in nonspecific airway responsiveness during the 6.6 hr exposures with moderate levels of exertion have been observed at 0.08, 0.10 and 0.12 ppm O_3.

The majority of asthma panel studies evaluated the associations of ambient O_3 with lung function and respiratory symptoms in asthmatic children. Results obtained from these studies show some inconsistencies, with some indicating significant positive associations and other smaller studies not finding such effects. Overall, however, the multicity study by Mortimer et al. (2002) and several credible single-city studies (e.g., Gent et al., 2003) indicate a fairly robust association between ambient O_3 concentrations and increased respiratory symptoms in moderate to severe asthmatic children (2006 CD, p. 8-35).

The 2006 CD (p. 8-75) concludes that lung inflammatory responses have been observed in healthy human adults following 6.6 hr O_3 exposures as low as 0.08 ppm. These responses have been found even in the absence of O_3-induced lung function decrements for some individuals. Attenuation of most inflammatory markers occurs with repeated exposures over several days, but none of the several markers of lung injury and permeability show attenuation, which is indicative of continued lung tissue damage during repeated exposure. Laboratory animal studies have reported that 1 to 3 hr O_3 exposures as low as 0.1 to 0.5 ppm can cause (1)

lung inflammatory responses (e.g., increased ROS and inflammatory cytokines, influx of PMNs, and activation of AMs); (2) damage to epithelial airway tissues, (3) increases in permeability of both lung endothelium and epithelium, and (4) increases in susceptibility to infectious diseases due to modulation of lung host defenses. Consistent with the above results of human and animal experimental studies, there is limited epidemiologic evidence of an association between acute ambient O_3 exposure (1-hr max of about 0.1 ppm) and airway inflammation in children, all of which taken together is indicative of a causal role for O_3 in inflammatory responses in the airways (2006 CD, p. 8-76). See Table 3.4 for a summary of short-term health effects of O_3 based on clinical studies assessed in both the 1996 CD and 2006 CD.

The 1996 CD concluded that increased O_3 levels are associated with increased hospital admissions and ED visits for respiratory causes. Analyses from data in the northeastern U.S. suggested that O_3 air pollution is associated with a substantial portion of all summertime respiratory hospital visits and admissions. The 2006 CD concludes (CD, p. 8-36) that a large multi-city and several single-city studies have indicated a positive association between increased O_3 levels (especially during the warm season) and increased risk for respiratory hospital admissions and asthma ED visits.

3.7.2 Cardiovascular Morbidity Effects of Short-term Exposures to Ozone

One health endpoint that was unrecognized in the 1996 CD, but is addressed in the 2006 CD, is O_3-induced cardiovascular effects. Newly available evidence has emerged since 1996 which provides considerable plausibility for how O_3 could exert cardiovascular effects (2006 CD, p. 8-77). Examples of such O_3-induced cardiovascular effects include: (1) O_3-induced release from lung epithelial cells of PAF that may contribute to blood clot formation that would increase the risk of serious cardiovascular outcomes (e.g., heart attack, stroke, mortality); (2) interactions of O_3 with surfactant components in ELF of the lung resulting in production of oxysterols and ROS that may exhibit PAF-like activity contributing to clotting and/or exerting cytotoxic effects on lung and heart cells; (3) possible mechanisms that may involve O_3-induced secretions of vasoconstrictive substances and/or effects on neuronal reflexes that may result in increased arterial blood pressure and/or altered electrophysiologic of heart rate or rhythm; (4) associations between O_3 and various cardiac physiologic endpoints suggesting a potential relationship between O_3 exposure and altered HRV, ventricular arrhythmias, and incidence of MI; and (5) positive associations during the warm season only between ambient O_3 concentrations and cardiovascular hospitalizations. While the only controlled human exposure study that evaluated effects of O_3 exposure on the cardiovascular system found no O_3-induced differences in ECG or blood pressure in healthy or hypertensive subjects, the study did report overall increases in myocardial work and heart rate, and impairment in pulmonary gas exchange.

Table 3-4. Summary of Ozone-Induced Respiratory Health Effects from Clinical Studies[6]

Health Effect	Exercise Level	Prolonged Exposure	Short-term Exposure	Lowest Ozone Effect Level
Pulmonary Function Decrements	Moderate	6.6 hr		0.06 ppm
	Moderate	6.6 hr		0.08 ppm
	Moderate	4.6 hr		0.10 ppm
	Moderate	3.0 hr		0.12 ppm
	Competitive		1 hr	0.12-0.14 ppm
	Very Heavy		1-3 hr	0.16 ppm
	Heavy		1-3 hr	0.18 ppm
	Moderate		1-3 hr	0.30 ppm
	Light		1-3 hr	0.37 ppm
	At rest		1-3 hr	0.50 ppm
Increased Respiratory Symptoms	Moderate	6.6 hr		0.06 ppm
	Moderate	6.6 hr		0.08 ppm
	Very Heavy		1-3 hr	0.12 ppm
Airway Responsiveness	Moderate	6.6 hr		0.08 ppm
	Very Heavy		1-3 hr	0.18 ppm
	At rest		1-3 hr	0.40 ppm
Respiratory Inflammation	Moderate	6.6 hr		0.08 ppm
	Very Heavy		1-3 hr	0.20 ppm
Changes in Host Defenses	Moderate	6.6 hr		0.08 ppm
Decreased Exercise Performance	Competitive		1 hr	0.18 ppm

Also, animal toxicological studies have reported O_3-induced decreases in heart rate, mean arterial pressure and core temperature. Overall, the 2006 CD (p. 8-77) concludes that this generally limited body of evidence is highly suggestive that O_3 directly and/or indirectly contributes to cardiovascular-related morbidity, but much remains to be done to more fully substantiate links between short-term ambient O_3 exposures and adverse cardiovascular effects.

3.7.3 Mortality-Related Effects of Short-term Exposures to Ozone

The 1996 CD concluded that an association between daily mortality and O_3 concentration for areas with high O_3 levels (e.g., Los Angeles) was suggested. However, due to a very limited number of studies available at that time, there was insufficient evidence to conclude that the observed association was likely causal. Since 1996, new data are available from large multicity

[6] Information contained in this table is based on scientific data assessed in Chapters 6 and 8 of the 2006 CD.

studies conducted in the U.S. and Europe, and several single-city studies conducted all over the world, as well as from several meta-analyses that have combined information from multiple studies. The majority of these studies suggest an elevated risk of total nonaccidental mortality associated with acute exposure to O_3, especially in the summer or warm season when O_3 levels are typically high, with somewhat larger effect estimate sizes for associations with cardiovascular mortality (2006 CD, p. 7-175). The 2006 CD finds that the results from U.S. multicity time-series studies provide the strongest evidence to-date for associations between short-term O_3 exposure and mortality. These studies, along with recent meta-analyses, showed consistent effect estimates that are unlikely to be confounded by PM, though the 2006 CD observes that future work is needed to better understand the influence of model specifications on the effect estimates (2006 CD, p. 7-175). For cardiovascular mortality, the 2006 CD reports that effect estimates are consistently positive, falling in the range of 1 to 8% increases per 40 ppb in 1-hr max O_3 (2006 CD, p. 7-107). Overall, the 2006 CD concludes that the majority of these findings suggest an elevated risk of all-cause mortality associated with short-term O_3 exposure, especially in the summer or warm season when O_3 levels are typically high. Slightly greater effects were observed for cardiovascular mortality (2006 CD, p. 7-175).

3.7.4 Health Effects of Repeated Short-term Exposures to Ozone

The 1996 CD drew several conclusions regarding repeated short-term O_3 exposures (2006 CD, p. 8-15). Partial or complete attenuation is observed for some of the O_3-induced responses after more than 2 days of exposure. After 5 days of exposure, lung function changes return to control levels with the greatest changes usually occurring on the second day, but the attenuation was reversed after 7 to 10 days without O_3 exposure. Most inflammatory markers (e.g., PMN influx) attenuate after 5 days of exposure, but markers of cell damage (e.g., LDH enzyme activity) do not attenuate and continue to increase. Recovery of some inflammatory markers occurred a week to 10 days after exposure ceased, but some responses were not normal after 20 days. Animal studies suggest underlying cell damage continues throughout the attenuation process. Also, attenuation may alter normal distribution of O_3 within the lungs, allowing more O_3 to reach sensitive regions, possibly affecting lung defenses. Newer studies assessed in the 2006 CD (p. 8-74 and 8-75) supported all of these conclusions in addition to which it was concluded that repeated daily, multi-hour exposure to lower concentrations of O_3 (0.125 ppm for 4 days) causes an increased response to bronchial allergen challenge in subjects with preexisting allergic airway disease, with or without asthma. In these subjects, changes in airway responsiveness after O_3 exposure appear to be resolved more slowly than changes in FEV_1 or respiratory symptoms.

3.7.5 Confidence in Various Health Outcomes Associated with Short-term Exposures to Ozone

In characterizing the extent to which relationships between the various health outcomes discussed above and short-term exposures to ambient O_3 are likely causal, we note that several different factors have informed the judgments made in the CD and here. These factors include the nature of the evidence (i.e., controlled human exposure, epidemiological, and/or toxicological studies) and the weight of evidence, including such considerations as biological plausibility, coherence of evidence, strength of association, and consistency of evidence.

In assessing the health effects data base for O_3, it is clear that human studies provide the most directly applicable information because they are not limited by the uncertainties of dosimetry differences and species sensitivity differences, which would need to be addressed in extrapolating animal toxicology data to human health effects. Controlled human exposure studies provide dat

a with the highest level of confidence since they provide human effects data under closely monitored conditions and can provide clear exposure-response relationships. Epidemiological data provide evidence of associations between ambient O_3 levels and more serious acute and chronic health effects (e.g., hospital admissions and mortality) that cannot be assessed in controlled human exposure studies. For these studies the degree of uncertainty regarding potential confounding variables (e.g., other pollutants, temperature) and other factors affects the level of confidence that the health effects being investigated are attributable to O_3 exposures, alone and in combination with other copollutants.

In using a weight of evidence approach to inform judgments about the degree of confidence that various health outcomes are likely to be caused by exposure to O_3, confidence increases as the number of studies and other factors, such as strength, consistency, and coherence of evidence, consistently reporting a particular health endpoint grows. For example, there is a very high level of confidence that O_3 induces lung function decrements in healthy adults and children due in part to the dozens of studies consistently showing that these effects were observed. As noted above, the 2006 CD (p. 8-74) states that studies provide clear evidence of causality for associations between short-term O_3 exposures and statistically significant declines in lung function in children, asthmatics and adults who exercise outdoors. An increase in respiratory symptoms (e.g., cough, shortness of breath) has been observed in controlled human exposure studies of short-term O_3 exposures, and significant associations between ambient O_3 exposures and a wide variety of symptoms have been reported in epidemiology studies (2006 CD, p. 8-75). Aggregate population time-series studies showing robust associations with respiratory hospital admissions and ED visits are strongly supported by human clinical, animal toxicologic, and epidemiologic evidence for lung function decrements, respiratory symptoms,

airway inflammation, and airway hyperreactivity. Taken together, the 2006 CD (p. 8-77) concludes that the overall evidence supports the inference of a causal relationship between acute ambient O_3 exposures and increased respiratory morbidity outcomes resulting in increased asthma ED visits and respiratory hospitalizations during the warm season. Recent epidemiologic evidence has been characterized in the CD (p. 8-78) as highly suggestive that O_3 directly or indirectly contributes to non-accidental and cardiopulmonary-related mortality.

As discussed above in section 3.5 and in section 8.6 of the 2006 CD, conclusions regarding biological plausibility, consistency, and coherence of evidence of O_3-related health effects are drawn from the integration of epidemiological studies with mechanistic information from controlled human exposure studies and animal toxicological studies. This type of mechanistic linkage has been firmly established for several respiratory endpoints (e.g., lung function decrements, lung inflammation) but remains far more equivocal for cardiovascular endpoints (e.g., cardiovascular-related hospital admissions). Finally, for epidemiological studies, strength of association refers to the magnitude of the association and its statistical strength, which includes assessment of both effects estimate size and precision (section 3.4.1). In general, when associations yield large relative risk estimates, it is less likely that the association could be completely accounted for by a potential confounder or some other bias. Consistency refers to the persistent finding of an association between exposure and outcome in multiple studies of adequate power in different persons, places, circumstances and times (section 3.4.3). For example, the magnitude of effect estimates is relatively consistent across recent studies showing association between short-term, but not long-term, O_3 exposure and mortality.

Figure 3-5 summarizes our judgments for the various health outcomes discussed above concerning the extent to which relationships between various health outcomes and ambient O_3 exposures are likely causal. These judgments are informed by the conclusions and discussion in the CD and in earlier sections of this chapter, reflecting the nature of the evidence and overall weight of the evidence, and are taken into consideration in our quantitative risk assessment, presented below in Chapter 5.

3.7.6 Health Effects of Long-term Exposures to Ozone

In the 1996 CD, available data, primarily from animal toxicology studies, indicated that exposure to O_3 for periods of months to years causes structural changes in several regions of the respiratory tract (2006 CD, p. 8-79). Effects may be of greatest importance in the CAR, where the gas exchange region and conducting airways meet. This region of the lungs is typically affected in most human airway diseases. However, data from epidemiological are limited or inconclusive, and data from clinical studies are lacking. Most information on chronic O_3 effects in the distal lungs continues to come from animal toxicology studies.

Figure 3-5. Qualitative Characterization of Ozone-Related Health Effect Outcomes

Characterization	Overall Confidence in Causal Relationship With Ambient Ozone
Causal ↓ Suggestive	-Lung function decrements in healthy children -Lung function decrements in asthmatic children -Lung function decrements in healthy adults -Respiratory symptoms in asthmatic children -Respiratory symptoms in healthy adults -Increased lung inflammation -Aggravation of asthma (i.e., increased medication usage, increased asthma attacks) -Respiratory-related hospital admissions -Respiratory related emergency department visits -Respiratory-related doctors visits -Increased school absences -Respiratory-related mortality during the O_3 season -Cardiorespiratory-related mortality during the O_3 season -Total nonaccidental mortality during the O_3 season -Cardiovascular-related hospital admissions

What had been viewed previously as an apparent lack of reversibility of O_3-induced effects with a clean air recovery period has been investigated since 1996 with animal toxicology studies using exposure regimens simulating a seasonal exposure pattern. One long-term study exposed rhesus monkeys to a simulated seasonal O_3 pattern (0.5 ppm O_3 8hr/day for 5 days, every 14 days for 11 episodes) and reported: (1) remodeling in the distal airways; (2) abnormalities in tracheal basement membrane; (3) eosinophil accumulation in conducting airways; and (4) decrements in airway innervation. These findings support and advance the earlier information suggestive of injury and repair processes which are caused by seasonal O_3 exposures (2006 CD, p.8-79). Epidemiological studies investigating chronic effects in humans following long-term exposures to O_3 have provided only limited suggestive evidence. Further investigation will be necessary before we are able to draw firmer conclusions about chronic health effects of O_3 in human populations.

3.7.7 Health Effects of Pollutant Mixtures Containing Ozone

In the 1996 CD, it was recognized that coexposure to O_3 and other pollutants, such as NO_2, SO_2, H_2SO_4, HNO_3, or CO, showed additive response for lung spirometry or respiratory symptoms (2006 CD, p. 8-82). Since 1996, most animal toxicology studies investigating O_3 in a mixture with NO_2 and H_2SO_4 have shown that effects can be additive, synergistic, or even antagonistic, depending on the exposure regimen and the endpoint studied. Ozone has served for a long time as a surrogate or indicator for the overall photochemical oxidant mix. It is well recognized that the observed effects may be due to components of that mix alone or in combination with O_3 and other gases and PM in the ambient air. Although the issue of exposure to copollutants was previously described as poorly understood, especially with regard to chronic effects, newer information from human and animal studies of binary mixtures containing O_3 suggest potential interactions depending on the exposure regimen and pollutant mix (CD, p. 8-82). Examples of this newer information include: (1) continuous exposure to SO_2 and NO_2 increased inhaled O_3 bolus absorption, while continuous exposure to O_3 decreased O_3 bolus absorption; (2) asthmatics exhibited enhanced airway reactivity to house dust mite allergen following exposures to O_3, NO_2 and the combination of the two gases; however, spirometric response was impaired only by O_3 and O_3+ NO_2 at higher concentrations; and (3) animal toxicology studies with O_3 in mixture with NO_2, formaldehyde, and PM demonstrated additive, synergistic, or antagonistic effects depending on the exposure regimen and the endpoints evaluated.

One controlled-exposure study of children, designed to approximate conditions of an epidemiological study by matching population and exposure atmosphere (0.1 ppm O_3, 0.1 ppm SO_2, and 101 ug/m^2 H_2SO_4), failed to support the findings of the epidemiological study. This

demonstrates the difficulty of trying to link outcomes of epidemiological studies and controlled-exposure studies with pollutant mixtures.

3.7.8 Populations at Risk/Susceptibility Factors Associated with Ozone Exposure

The 1996 CD (2006 CD, p. 8-80) identified several factors that may increase sensitivity to O_3 of population groups, including: (1) biological variation in responsiveness to O_3; (2) preexisting lung disease (e.g., asthma); (3) activity patterns (e.g., exertion levels); (4) personal exposure history (e.g., time spent indoors v. outdoors); and (5) personal factors (e.g., age, nutritional status, gender, smoking history, ethnicity). Based on the information assessed in the 1996 CD (2006 CD, p. 8-80), population groups that demonstrated increased responsiveness to ambient concentrations of O_3 consisted of exercising, healthy and asthmatic individuals, including children, adolescents, and adults. Since 1996, evidence from controlled-exposure human and animal studies, as well as from epidemiological studies, has provided further support for these and other susceptibility factors and populations at risk. For example, controlled-exposure human studies continue to show differential biological response to O_3 based on physical activity (exertion) and age. These studies demonstrate a large variation in sensitivity and responsiveness to O_3, although specific factors that contribute to this intersubject variability are yet to be identified. Associations of increased summertime hospital admissions for asthma with ambient O_3 levels suggest that individuals with these respiratory diseases are populations at risk to O_3 exposure effects. Also, based on O_3-induced differential response in lung inflammation and airway responsiveness, asthmatic adults and children appear to have potentially increased susceptibility to O_3. There is limited evidence from epidemiologic studies and no evidence from controlled-exposure human studies which suggest that individuals with COPD are more sensitive to health effects of O_3.

There is some animal toxicology and limited epidemiologic evidence which has demonstrated the importance of genetic background in O_3 susceptibility. Genetic and molecular characterization studies of experimental animals have identified genetic loci responsible for both sensitivity and resistance.

Taking all of this information into account, the CD (p. 8-80 to 8-81) concludes that exercising (moderate to high physical exertion) healthy and asthmatic adults, adolescents, and children appear to exhibit increased responsiveness to ambient O_3 levels and continue to be considered at increased risk of O_3-induced health effects. Also, any individual with respiratory or cardiovascular disease or any healthy individual who is engaged in vigorous physical activity outdoors during periods when O_3 levels are high (e.g., active outdoor children) is potentially at increased risk to O_3-induced health effects. In addition, healthy individuals and those with

cardiorespiratory impairment (e.g., those with asthma or cardiovascular disease) who are "hyperresponsive" to O_3 exposure (i.e., exhibit much higher than normal lung function decrements and/or respiratory symptoms) would be considered at greater risk to O_3 exposure. Finally, individuals who are more likely to be exposed to air pollution while engaged in physical activity (e.g., outdoor workers) and those with genetic polymorphisms for antioxidant enzymes and inflammatory genes may be at heightened risk of effects of O_3 (2006 CD, p. 8-81).

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4. CHARACTERIZATION OF HUMAN EXPOSURE TO OZONE

4.1 INTRODUCTION

As part of the last O_3 NAAQS review, EPA conducted exposure analyses for the general population, children who spent more time outdoors, and outdoor workers. Exposure estimates were generated for nine urban areas for "as is" (i.e., a recent year) air quality and for just meeting the existing 1-hr standard and several alternative 8-hr standards. EPA also conducted a health risk assessment that produced risk estimates for the number of children and percent of children experiencing impaired lung function and other respiratory symptoms associated with the exposures estimated for these same nine urban areas.

The exposure analysis conducted for the current review builds upon the methodology and lessons learned from the exposure analyses conducted for the last review (U.S. EPA, 1996a). The methodology used to conduct the exposure analysis as well as summary results from the exposure analysis are described in this chapter. The exposure analysis technical support document, *Ozone Population Exposure Analysis for Selected Urban Areas* (U.S. EPA, 2007) (hereafter cited as "Exposure Analysis TSD") presents a detailed description of the exposure analysis methodology.

Population exposures to ambient O_3 levels are modeled for 12 urban areas located across the U.S. using the Air Pollutants Exposure (APEX) model, also referred to as the Total Risk Integrated Methodology Inhalation Exposure (TRIM.Expo) model (U.S. EPA, 2006a,b). Exposure estimates are developed for O_3 levels in recent years, based on 2002, 2003, and 2004 ambient air quality measurements. Exposures are also estimated for O_3 levels associated with just meeting the current 8-hr O_3 NAAQS and several potential alternative standards, based on adjusting data derived from the ambient monitoring network as described in section 4.5.8. Exposures to background levels of O_3 are also estimated, based on O_3 concentrations predicted by the GEOS-CHEM atmospheric photochemical model.

Exposures are modeled for 1) the general population, 2) all school-age children (ages 5-18), 3) active school-age children, and 4) asthmatic school-age children.[1] The strong emphasis on children reflects the finding of the last O_3 NAAQS review that children are an important at-risk group. Two groups at potentially increased risk where exposure modeling

[1] Subsequent to completion of the modeling conducted in 2006, EPA analysis of uncertainty of the exposure modeling results uncovered an error in how children are characterized as active. This error resulted in an overestimate of the number of active children in the population. Thus, exposure estimates for active children are not included in this chapter.

results are not presented are older adults and outdoor workers. Although older adults are in the general population exposure estimates, we did not separately tabulate exposures for older adults since we do not have specific exposure-response data for this group. Exposure to outdoor workers has not been modeled due to insufficient data to properly characterize this population.

This chapter provides a brief overview of the types of studies that provide data on which this analysis is based, followed by a description of the exposure model used for this analysis, the model input data, and the results of the analysis. The final sections of this chapter discuss the exposure estimates in comparison to those from the prior review and summarize the sensitivity analyses and model evaluation that have been conducted for the O_3 exposure model described in this chapter. The uncertainty assessment and a technical description of the modeling effort are provided in separate documents (Langstaff, 2007; U.S. EPA, 2007).

4.2 OZONE EXPOSURE STUDIES

Many studies have produced information and data supporting the development of methods for estimating human exposure to ambient O_3 over the past several decades. These studies have been reviewed in the current and previous EPA Ozone Air Quality Criteria Documents (U.S. EPA, 1986, 1996b, 2006c).

The types of studies which provide the basis for modeling human exposure to O_3 include studies of people's activities, work and exercise patterns, physiology, physics and O_3-related chemistry in microenvironments, atmospheric modeling of O_3, chamber studies of atmospheric chemistry, and modeling of meteorology. Measurements that have proven to be useful for understanding and estimating exposure obtained from personal exposure assessment studies include fixed-site ambient concentrations, concentrations in specific indoor and outdoor microenvironments, personal exposure levels, personal activity patterns, air exchange rates, infiltration rates, deposition and decay rates, and meteorology.

4.2.1 Exposure Concepts and Definitions

Human exposure to a contaminant is defined as "contact at a boundary between a human and the environment at a specific contaminant concentration for a specific interval of time," and has units of concentration times time (National Research Council, 1991). For airborne pollutants the contact boundary is nasal and oral openings in the body, and *personal exposure* of an individual to a chemical in the air for a discrete time period is quantified as (Lioy, 1990; National Research Council, 1991):

$$E_{[t_1, t_2]} = \int_{t_1}^{t_2} C(t)dt \qquad (4\text{-}1)$$

where $E_{[t_1,t_2]}$ is the personal exposure during the time period from t_1 to t_2, and $C(t)$ is the concentration at time t in the breathing zone. We refer to the *exposure concentration* to mean the concentration to which one is exposed. The breathing rate (ventilation rate) at the time of exposure is an important determinant of the dose received by the individual. Although we do not estimate dose, we refer to *intake* as the total amount of O_3 inhaled (product of exposure concentration, duration, and minute ventilation rate).

Personal exposure to O_3 can be estimated directly by monitoring the concentration of O_3 in the person's breathing zone (close to the nose/mouth) using a personal exposure monitor. Exposure can also be estimated indirectly, by estimating or monitoring the concentrations over time in locations in which the individual spends time and estimating the time and duration the individual spends in each location. In both of these methods, Equation 4-1 is used to calculate an estimate of personal exposure. A key concept in modeling exposure is the *microenvironment*, a term that refers to the immediate surroundings of an individual. A microenvironment is a location in which pollutant concentrations are relatively homogeneous for short periods of time. Microenvironments can be outdoors or indoors; some examples are outdoors near the home, outdoors near the place of work, bedrooms, kitchens, vehicles, stores, restaurants, street-corner bus stops, schools, and places of work. A bedroom may be treated as a different microenvironment than a kitchen if the concentrations are significantly different in the two rooms. The concentrations in a microenvironment typically change over time; for example, O_3 concentrations in a kitchen while cooking with a gas stove may be lower than when these activities are not being performed, due to scavenging of O_3 by nitric oxide (NO) emissions from the gas burned.

An important factor affecting the concentrations of O_3 indoors is the degree to which the ambient outdoor air is transported indoors. This can be modeled using physical factors such as air exchange rates (AERs), deposition and decay rates, and penetration factors. The *volumetric exchange rate* (m^3/hour) is the rate of air exchange between the indoor and outdoor air. The *AER* between indoors and outdoors is the number of complete air exchanges per hour and is equal to the volumetric exchange rate divided by the volume of the well-mixed indoor air. Indoor concentrations of O_3 can be decreased by uptake of O_3 by surfaces and by chemical reactions. The *deposition* and *chemical decay rates* are the rates (per hour) at which O_3 is removed from the air by surface uptake and chemical reactions. Some exposure models employ an infiltration factor, which is conceptually useful if distinguishing between the air exchange processes of air blowing through open doors and windows and the infiltration of air through smaller openings. Since measurements of AERs account for both of these processes (including infiltration), this distinction is not useful in applied modeling of O_3 exposures and will not be discussed further here. Simpler exposure models use a "factor model" approach to estimate indoor O_3

concentrations by multiplying the ambient outdoor concentrations by an indoor/outdoor concentration ratio, referred to as a *penetration factor*.

4.2.2 Monitoring Equipment Considerations

Exposure assessment studies involve monitoring airborne O_3 and/or other pollutants, and monitor design and placement play a critical role in interpreting the results of these studies. For exposure assessment purposes there are two general classes of monitors, personal exposure monitors (PEMs) and fixed site monitors.

PEMs are designed to be worn or carried easily by individuals and to measure the concentrations experienced by individuals over a period of hours, days, or weeks. The placement of PEMs is important; the desired placement is usually in the breathing zone near the mouth and nose, but where the monitor will not be excessively impacted by exhaled air. This placement is intended to represent the concentrations the individual breathes in. PEMs typically report continuously measured O_3 concentrations with averaging times ranging from 1 to 24 hours.

The CD reviews O_3 PEMS (CD, Appendix AX3, p. 163-5) and notes that humidity, wind velocity, badge placement, and interference with other pollutants may result in measurement error. The CD reports PEM detection limits ranging from 5 to 23 ppb for averaging times from 24-hr to 1-hr, respectively.

Fixed-site monitors measure concentrations over time at a given location. There are numerous fixed-site O_3 monitors which are part of national, state, and local air monitoring networks. In addition to their role of being used to determine which areas are in compliance with existing O_3 NAAQS, these are also useful for alerting the public to high O_3 days, providing air quality data in support of photochemical modeling and exposure assessments for a study area, for tracking O_3 levels and trends, and for studying the representativeness of measurements at these monitors for the study area. Existing fixed-site monitors usually report hourly averaged concentrations, and are in operation over a period of years. Federal reference and equivalent O_3 monitoring methods are required to have a lower detectable limit of 0.01 ppm and precision of 0.01 ppm for 1-hr average concentrations (40 CFR Ch. 1, §53.21). A discussion of monitoring equipment and networks can be found in Chapter 2 of this draft Staff Paper and in section 2.6 in the CD.

There are also stationary monitors expressly set up for particular exposure field studies. These are used to measure concentrations over time in microenvironments, such as rooms in a home, just outside a home, roadsides, and so forth. The stationary monitors which are outdoors can provide information about community-scale representativeness of routinely operated fixed-site monitors in or near the community.

4.2.3 Personal Ozone Exposure Assessment Studies

Useful PEM studies would have data collected repeatedly from each individual in the study over a period of time, yielding a longitudinal time series of hourly (or shorter) average concentrations each individual is exposed to. These studies would permit analysis of both the temporal and spatial variability of each person's personal exposure to O_3.

Some studies could be designed so that the data are sampled randomly from the population, which reduces bias and allows one to make inferences about exposure in the broader population. Most studies addressing O_3 exposure have not been of random design and the measurement averaging times are longer than hourly. They might have specific goals for which randomness is not required, or be subject to constraints which do not allow for random sampling. These non-random studies have been helpful in the development of models of exposure; however, we recognize that they may not be representative of the broader population.

4.2.4 Microenvironmental Studies

The focus of microenvironmental studies is on measuring concentrations in different locations that people spend time in, as well as on measuring the movement of pollutants from one microenvironment to another and on measuring other parameters that contribute to variability in exposure. Typically, microenvironmental measurements include indoor and outdoor concentrations of O_3 and other pollutants, AERs, infiltration factors, deposition rates, decay rates, emissions of O_3, NO_x, VOCs, and other pollutants, operating characteristics of air conditioning systems, and meteorological data such as wind velocity, temperature, and humidity. The CD discusses several studies of microenvironments that contribute to our understanding of the factors and processes that affect exposure to O_3 (CD Appendix AX3, p. 191-216).

There is a great deal of variability among individuals in the amount of time spent indoors, but the majority of people spend most of their time indoors (Graham & McCurdy, 2004), and therefore the concentrations of O_3 indoors can be an important determinant of people's exposure to O_3. There are several factors affecting O_3 concentrations indoors. The ambient outdoor concentration of O_3 and the AER are the primary determinants of the indoor concentrations. Removal processes are also significant, the most important of which is deposition onto indoor surfaces such as carpets, furnishings, and ventilation ductwork. Chemical reactions of O_3 with other compounds, such as solvents from consumer products or nitric oxide emissions from gas stoves, also deplete O_3 indoors (Weschler, 2000; Monn, 2001).

The primary sources of O_3 indoors are O_3-generating air cleaners and some photocopiers and laser printers. Ozone generators can increase indoor concentrations by more than 0.05 ppm. Some older photocopiers, if run continuously in an enclosed area, can increase O_3 concentrations

by as much as 0.15 ppm. Older laser printers can produce concentrations of up to 0.18 ppm indoors (U.S. EPA, 1995; CARB, 2005).

4.3 EXPOSURE MODELING

Models of human exposure to airborne pollutants are typically driven by estimates of ambient outdoor concentrations of the pollutants, which vary by time of day as well as by location. These outdoor concentration estimates may be provided by measurements, by air quality models, or by a combination of these. It is only possible to address hypothetical future scenarios using some form of modeling. The main purpose of this exposure analysis is to allow comparisons of population exposures to O_3 within each urban area, associated with current air quality levels and with several potential alternative air quality standards or scenarios. Human exposure, regardless of the pollutant, depends on where an individual is located and what they are doing. Inhalation exposure models are useful in realistically estimating personal exposures and intake based on activity-specific ventilation rates, particularly when recognizing that these measurements cannot be performed for a given population. This section provides a brief overview of the model used by EPA to estimate O_3 population exposure. Details about the application of the model to estimate O_3 population exposure are provided in the following sections and in the Exposure Analysis TSD (U.S. EPA, 2007).

4.3.1 The APEX Model

The EPA has developed the APEX model for estimating human population exposure to criteria and air toxic pollutants. APEX also serves as the human inhalation exposure model within the Total Risk Integrated Methodology (TRIM) framework (Richmond et al., 2002; EPA 2006c). APEX is conceptually based on the probabilistic NAAQS Exposure Model (pNEM) that was used in the last O_3 NAAQS review (Johnson et al., 1996a; 1996b: 1996c). Since that time the model has been restructured, improved, and expanded to reflect conceptual advances in the science of exposure modeling and newer input data available for the model. Key improvements to algorithms include replacement of the cohort approach with a probabilistic sampling approach focused on individuals, accounting for fatigue and oxygen debt after exercise in the calculation of ventilation rates, and a new approach for construction of longitudinal activity patterns for simulated persons. Major improvements to data input to the model include updated AERs, census and commuting data, and the daily time-activities database. These improvements are described later in this chapter.

APEX is a probabilistic model designed to account for the numerous sources of variability that affect people's exposures. APEX simulates the movement of individuals through time and space and estimates their exposure to a given pollutant in indoor, outdoor, and in-

vehicle microenvironments. Figure 4-1 provides a schematic overview of the APEX model. The model stochastically generates simulated individuals using census-derived probability distributions for demographic characteristics (Figure 4-1, steps 1-3). The population demographics are drawn from the year 2000 Census at the tract level, and a national commuting database based on 2000 census data provides home-to-work commuting flows between tracts.[2] Any number of simulated individuals can be modeled, and collectively they approximate a random sampling of people residing in a particular study area.

Daily activity patterns for individuals in a study area, an input to APEX, are obtained from detailed diaries that are compiled in the Consolidated Human Activity Database (CHAD) (McCurdy et al., 2000; EPA, 2002). The diaries are used to construct a sequence of activity events for simulated individuals consistent with their demographic characteristics, day type, and season of the year, as defined by ambient temperature regimes (Graham & McCurdy, 2004) (Figure 4-1, step 4). The time-location-activity diaries input to APEX contain information regarding an individuals' age, gender, race, employment status, occupation, day-of-week, daily maximum hourly average temperature, the location, start time, duration, and type of each activity performed. Much of this information is used to best match the activity diary with the generated personal profile, using age, gender, employment status, day of week, and temperature as first-order characteristics. The approach is designed to capture the important attributes contributing to an individuals' behavior, and of particular relevance here, time spent outdoors (Graham and McCurdy, 2004). Furthermore, these diary selection criteria give credence to the use of the variable data that comprise CHAD (e.g., data collected were from different seasons, different states of origin, etc.). APEX calculates the concentration in the microenvironment associated with each event in an individual's activity pattern and sums the event-specific exposures within each hour to obtain a continuous series of hourly exposures spanning the time period of interest (Figure 4-1, steps 5, 6).

APEX has a flexible approach for modeling microenvironmental concentrations, where the user can define the microenvironments to be modeled and their characteristics. Typical indoor microenvironments include residences, schools, and offices. Outdoor microenvironments include near roadways, at bus stops, and playgrounds. Inside cars, trucks, and mass transit vehicles are microenvironments which are classified separately from indoors and outdoors.

[2] There are approximately 65,400 census tracts in the ~3,200 counties in the U.S.

Figure 4-1. Overview of the APEX Model

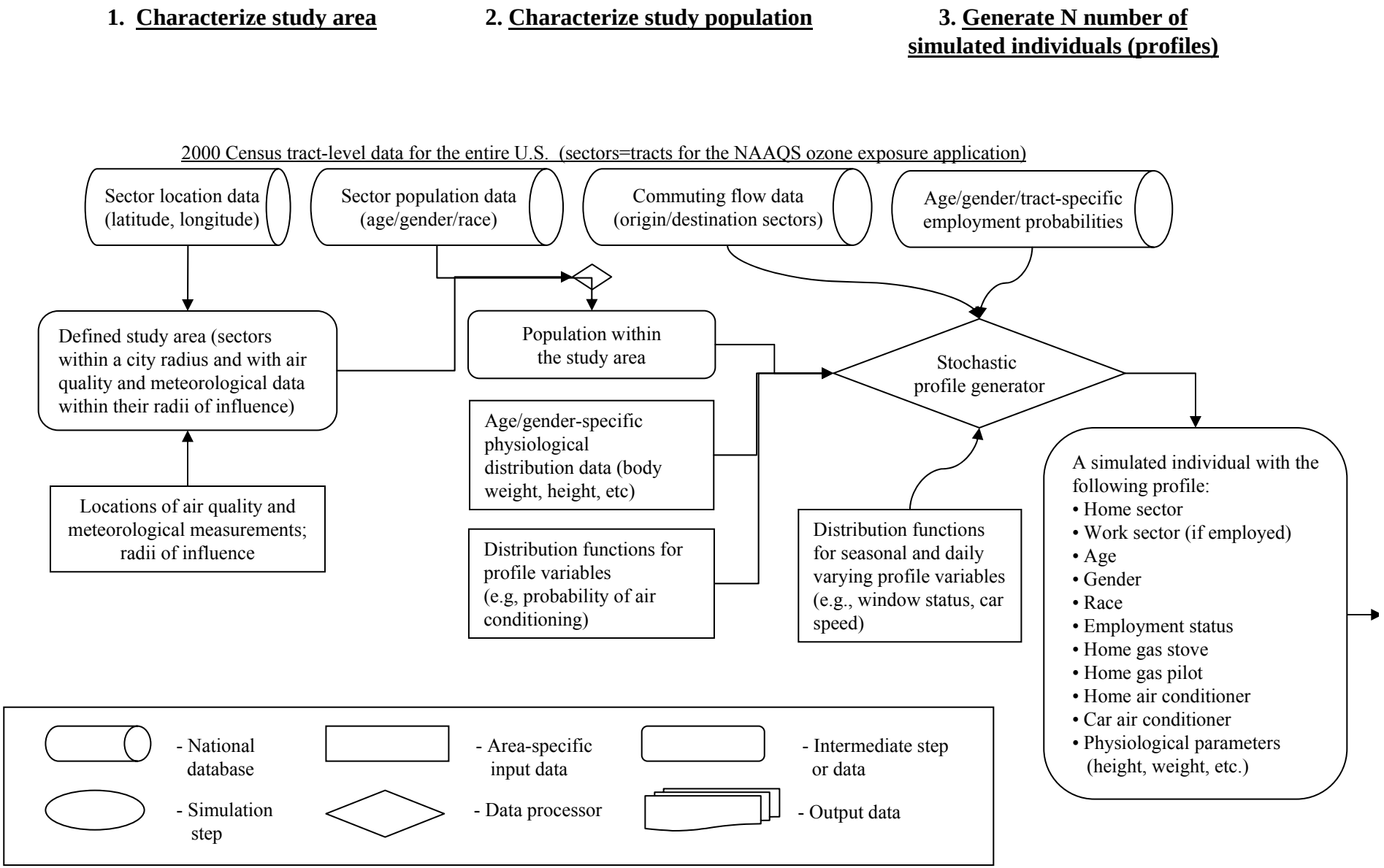


Figure 4-1. Overview of the APEX Model, continued

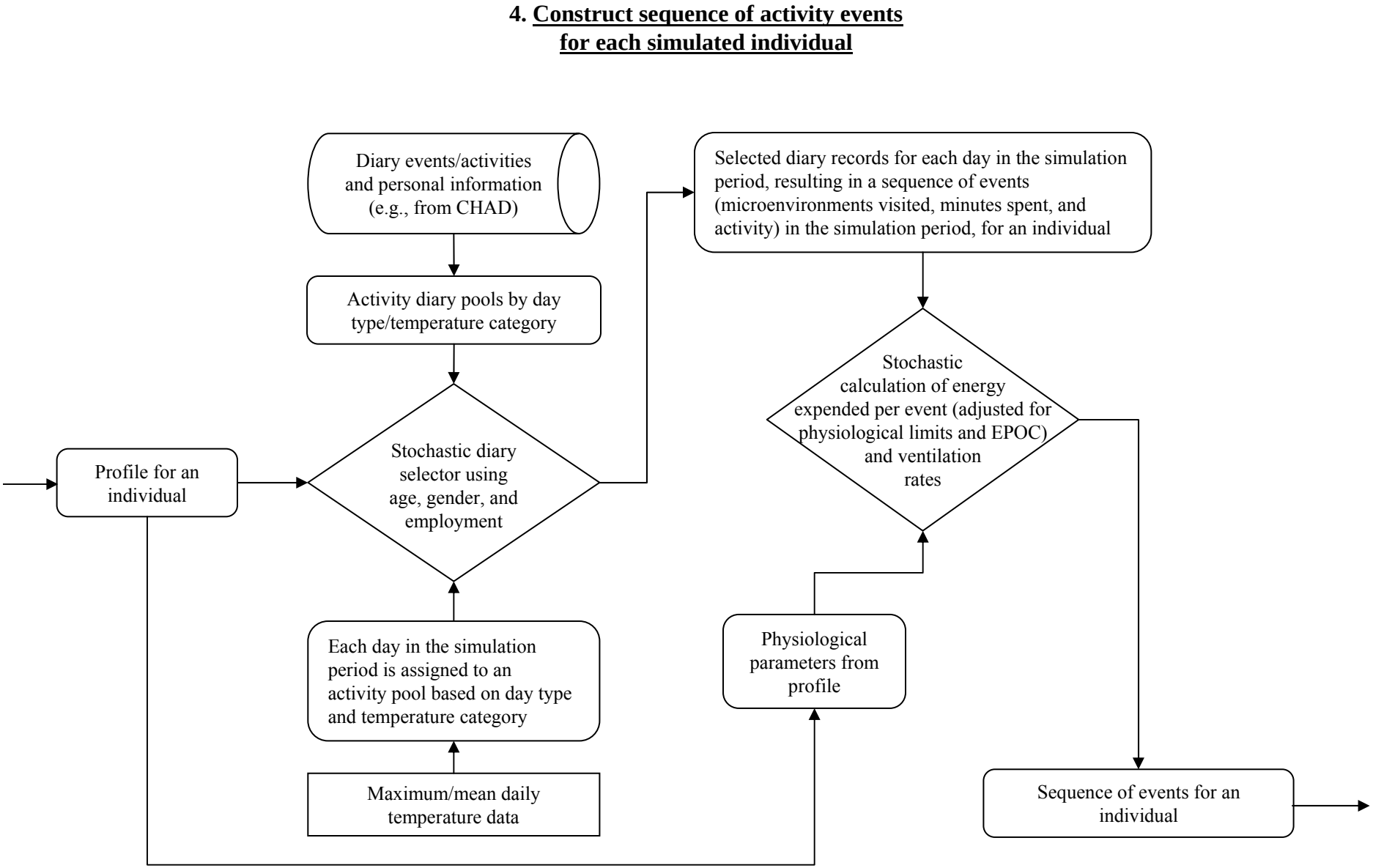
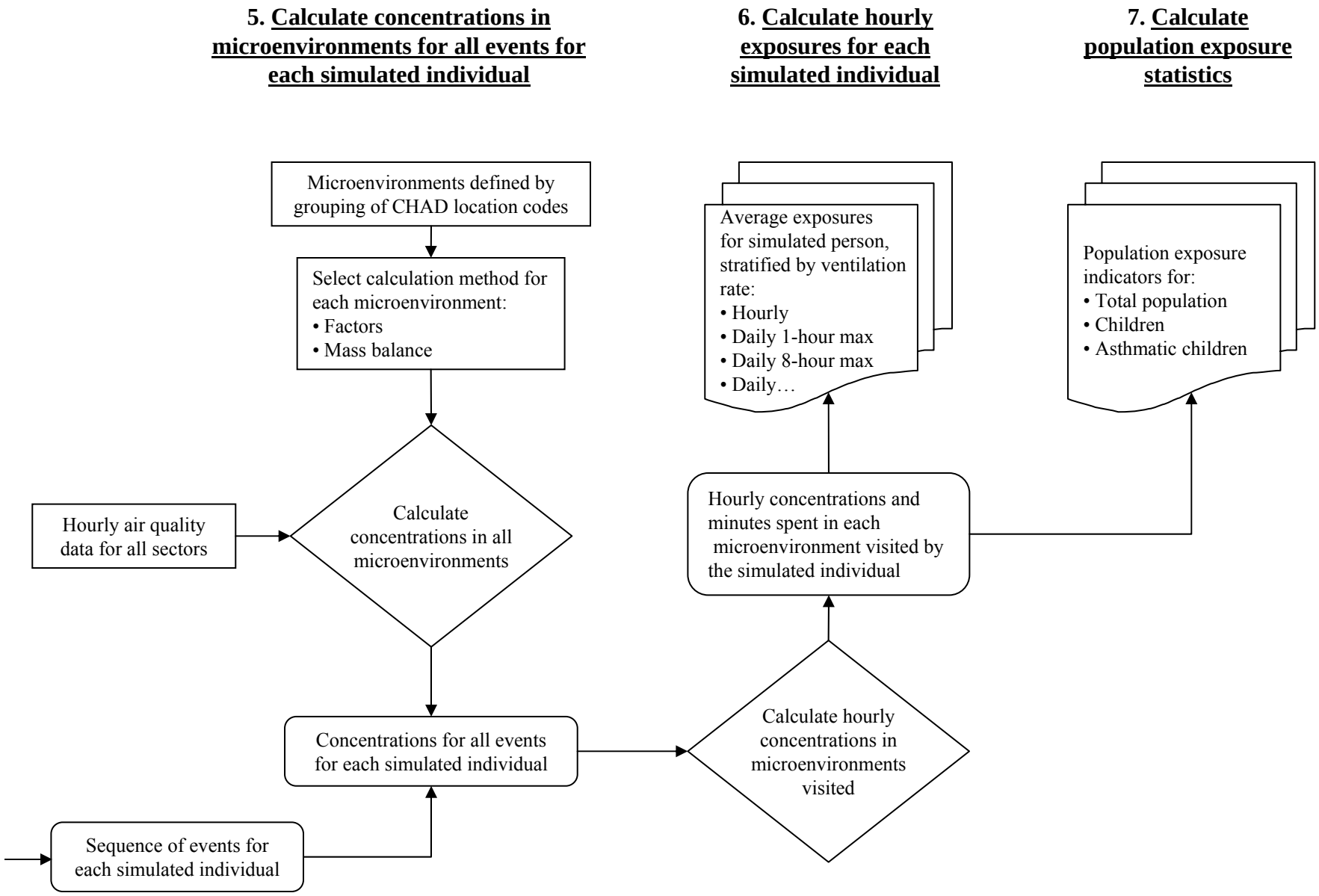


Figure 4-1. Overview of the APEX Model, concluded



Activity-specific simulated breathing rates of individuals are used in APEX to characterize intake received from an exposure. These breathing, or ventilation, rates are derived from energy expenditure estimates for each activity included in CHAD and are adjusted for age- and gender-specific physiological parameters associated with each simulated individual. Energy expenditure estimates themselves are derived from METS (metabolic equivalents of work) distributions associated with every activity in CHAD (McCurdy et al., 2000), largely based upon the Ainsworth et al. (1993) "Compendium of Physical Activities." METS are a dimensionless ratio of the activity-specific energy expenditure rate to the basal or resting energy expenditure rate, and the metric is used by exercise physiologists and clinical nutritionists to estimate work undertaken by individuals as they go through their daily life (Montoye et al., 1996). This approach is discussed more thoroughly in McCurdy (2000).

4.3.2 Key Algorithms

Ozone concentrations in each microenvironment are estimated using either a mass-balance or transfer factors approach, and the user specifies probability distributions for the parameters that are used in the microenvironment model that reflect the observed variabilities in the parameters. These distributions can depend on the values of other variables calculated in the model or input to APEX. For example, the distribution of AERs in a home, office, or car can depend on the type of heating and air conditioning present, which are also stochastic inputs to the model, as well as the ambient temperature. The user can choose to keep the value of a stochastic parameter constant for the entire simulation (which would be appropriate for the volume of a house), or can specify that a new value shall be drawn hourly, daily, or seasonally from specified distributions. APEX also allows the user to specify diurnal, weekly, or seasonal patterns for various microenvironmental parameters. The distributions of parameters input to APEX characterize the variability of parameter values, and are not intended to reflect uncertainties in the parameter estimates.

The mass balance method used within APEX assumes that the air in an enclosed microenvironment is well-mixed and that the air concentration is fairly spatially uniform at a given time within the microenvironment. The following four processes are modeled to predict the concentration of an air pollutant in such a microenvironment:

- Inflow of air into the microenvironment;
- Outflow of air from the microenvironment;
- Removal of a pollutant from the microenvironment due to deposition, filtration, and chemical degradation; and
- Emissions from sources of a pollutant inside the microenvironment.

The transfer factors model is simpler than the mass balance model, however, still most parameters are derived from distributions rather than single values, to account for observed variability. It does not calculate concentration in a microenvironment from the concentration in the previous hour and it has only two parameters, a proximity factor, used to account for proximity of the microenvironment to sources or sinks of pollution, or other systematic differences between concentrations just outside the microenvironment and the ambient concentrations (at the measurements site), and a penetration factor, which quantifies the degree to which the outdoor air penetrates into the microenvironment. When there are no indoor sources, the penetration factor is essentially the ratio of the concentration in the microenvironment to the outdoor concentration.

Regardless of the method used to estimate the microenvironmental concentrations, APEX calculates a time series of exposure concentrations that a simulated individual experiences during the modeled time period. APEX estimates the exposure using the concentrations calculated for each microenvironment and the time spent in each of a sequence of microenvironments visited according to the "activity diary" of each individual. The hourly average exposures of each simulated individual are time-weighted averages of the within-hour exposures. From hourly exposures, APEX calculates the time series of 8-hr and daily average exposures that simulated individuals experience during the simulation period. APEX then statistically summarizes and tabulates the hourly, 8-hr, and daily exposures.

4.3.3 Model Output

There are several useful indicators of exposure and intake of people to O_3 air pollution. Factors that are important include the magnitude and duration of exposure, frequency of repeated high exposures, and the breathing rate of individuals at the time of exposure. In this analysis, exposure indicators include daily maximum 1-hr and 8-hr average O_3 exposures, stratified by a measure of the level of exertion at the time of exposure. The level of exertion of individuals engaged in particular activities is measured by an equivalent ventilation rate (EVR), ventilation normalized by body surface area (BSA, in m^2), which is calculated as V_E/BSA, where V_E is the ventilation rate (liters/minute). Table 4-1 lists the ranges of EVR corresponding to moderate and heavy levels of exertion.

Table 4-1. Exertion levels in terms of equivalent ventilation rates (liters/min-m^2)

Averaging time	Moderate exertion	Heavy exertion
1 hour	16-30 EVR	$\geq$ 30 EVR
8 hour	13-27 EVR	$\geq$ 27 EVR

from Whitfield et al., 1996, page 15.

APEX calculates two general types of exposure estimates: counts of the estimated number of people exposed to a specified O_3 concentration level and the number of times per O_3 season that they are so exposed; the latter metric is in terms of person-occurrences or person-days. The former highlights the number of individuals exposed *one or more* times per O_3 season to the exposure indicator of interest. In the case where the exposure indicator is a benchmark concentration level, the model estimates the number of people who are expected to experience exposures to that level of air pollution, or higher, at least once during the modeled period. APEX also reports counts of individuals with multiple exposures. The person-occurrences measure estimates the number of times per season that individuals are exposed to the exposure indicator of interest and then accumulates these estimates for the entire population residing in an area. This metric conflates people and occurrences: one occurrence for each of 10 people is counted the same as 10 occurrences for one person.

APEX tabulates and displays the two measures for exposures above levels ranging from 0.0 to 0.16 ppm by 0.01 ppm increments, where the exposures are:

- Daily maximum 1-hour average exposures
- Daily maximum 8-hour average exposures
- Daily average exposures.

These results are tabulated for the following population groups:

- All ages and activity levels
- Children at all activity levels
- Asthmatic children.

Separate output tables are produced for different levels of exertion concomitant with the exposures:

- All exertion levels
- Moderate exertion levels
- Heavy exertion levels.

APEX also produces tables of the time spent in different microenvironments, stratified by exposure levels.

4.3.4 Strengths and Limitations of the Model

APEX has a strong scientific foundation and incorporates several significant algorithmic improvements and updates to input data since it's predecessor, pNEM, was used in the last review. In this section we discuss qualitatively some of the general strengths and limitations of the application of APEX to model population exposures to O_3 pollution. The discussion is divided into four fundamental areas: estimation of ambient air quality, estimation of concentrations in microenvironments, characterization of population demographics and activity patterns, and modeling physiological processes as applicable to this exposure assessment.

Additional details for advancements made to the model of specific relevance for this ozone exposure assessment are given in section 4.5.

In general, limitations and uncertainties result from variability not modeled or modeled incorrectly, erroneous or uncertain inputs, errors in coding, simplifications of physical, chemical, and biological processes to form the conceptual model, and flaws in the conceptual model. The implications of these limitations for the uncertainty of the APEX results is discussed in Langstaff (2007).

4.3.4.1 Estimation of Ambient Air Quality

For estimating ambient O_3 concentrations to use in the exposure model, the urban areas modeled here have several monitors measuring hourly O_3 concentrations. Having multiple monitors in the simulated areas collecting time-resolved data allows for the utilization of APEX spatial and temporal capabilities in estimating exposure. Since APEX uses actual records of where individuals are located at specific times of the day, more realistic exposure estimates are obtained in simulating the contact of individuals with these spatially and temporally diverse concentrations. Primary uncertainties in the air quality data input to the model result from estimating concentrations at locations which may not be in close proximity to monitoring sites (as estimated by spatial interpolation of actual data points) and from the method used to estimate missing data. In addition, concentrations of O_3 near roadways are particularly difficult to estimate due to the rapid reaction of O_3 with nitric oxide emitted from motor vehicles.

We have modeled the O_3 seasons for 2002, 2003, and 2004, to better account for year-to-year variability of air quality and meteorology. For most of the 12 areas modeled, O_3 concentrations were lower in 2004 than previous years, due to a combination of reduced emissions of precursors and weather patterns less conducive to the formation of O_3. Having this wide range of air quality data across multiple years available for use in the exposure simulation has a direct impact on more realistically estimating the range of exposures, rather than using a single year of air quality data.

Modeling exposures for an unspecified future year simulated to just meet alternative air quality standards has, in addition to the uncertainties involved with modeling historical scenarios, the uncertainties of the complex process of projecting to future years air quality, population demographics, activity patterns, and other changing parameters. For the purpose of estimating population exposure as an input to decisions about the appropriate level of a NAAQS, EPA has historically not incorporated any projections in population demographics, activity patterns, or other factors (e.g., air conditioning use, changes in housing types, etc). This allows policy makers to focus on the impact of changing the allowed air quality distribution on

population exposure and public health while avoiding the additional uncertainties that inclusion of these other factors would introduce.

4.3.4.2 Estimation of Concentrations in Indoor Microenvironments

The importance of estimation of concentrations in indoor microenvironments (e.g., homes, offices, schools, restaurants, vehicles) is underscored by the finding that personal exposure measurements of O_3 are often not well-correlated with ambient measurements (CD, pages 3-59 to 3-61).

APEX has been designed to better estimate human exposure through use of algorithms that attempt to capture the full range of O_3 concentrations expected within several important microenvironments. Parameters used to estimate the concentrations in microenvironments can be highly variable, both between microenvironments (e.g., different houses have varying characteristics) and within microenvironments (e.g., the characteristics of a given house can vary over time). Since APEX is a probabilistic model, if data accurately characterizing this variability are provided to the model, then such variabilities would not result in uncertainties in the estimation of the microenvironmental concentrations. Thus, it is the input data used in development of the parameters that are the limiting factor, and to date, APEX uses the most current available data to develop required input parameters for estimation of microenvironmental concentrations.

Air Exchange Processes

The AER is the single most important factor in determining the relationship between outdoor and indoor concentrations of O_3. AERs are highly variable, both within a microenvironment over time and between microenvironments of the same type. AERs depend on the physical characteristics of a microenvironment and also on the behavior of the occupants of the microenvironment. There is a strong dependence on temperature, and some dependence on other atmospheric conditions. APEX uses probabilistic distributions of AERs which were derived from several measurement studies in a number of locations, and are stratified by both temperature and the presence or absence of air conditioning. These two variables are the most influential variables influencing AER distributions (see Appendix A of the Exposure Analysis TSD).

Removal Processes

Concentrations within indoor microenvironments can be reduced due to removal processes such as deposition to surfaces and by reaction with other chemicals in the air. The rate of deposition of O_3 to a surface depends on the surface composition, the humidity, and the concentration of O_3. The rate of removal of O_3 from a microenvironment depends on its dimensions, the ratio of surface area to volume, and of course the presence, amount, and type of

surface coverings and furnishings in the microenvironment. Deposition is modeled probabilistically in APEX by a using a distribution of decay rates derived from a study that measured decay rates in 26 homes in Southern California (Lee et al., 1999). Although it is not expected that inter-city differences in decay rates would be more important than differences between homes within cities, there is uncertainty given the small sample size the distribution was derived from. The lack of capturing inter-city variability in decay rates is not anticipated to be a large contributor to the uncertainty in the modeling results. There can be additional O_3 loss, which is not currently modeled, due to the use of HVAC systems, which significantly increase the effective surface area as air recirculates through ductwork and filters.

Ozone reacts with a number of indoor pollutants, such as nitric oxide from gas stoves and VOCs from consumer products. With the exception of nitric oxide, O_3 reacts slowly with most indoor pollutants, and this is typically a less influential removal process than air exchange and surface removal (Weschler, 2000). The lack of a better treatment of indoor air chemistry is not considered to be a significant limitation of APEX for modeling O_3.

4.3.4.3 Characterization of Population Demographics and Activity Patterns

The approach to reasonably estimating exposure considering a variety of alternative scenarios is best done using models that better represent the contact of a human with the contaminant of concern. By using actual time-location-activity diaries that capture the duration and frequency of occurrence of visitations/activities performed, APEX can simulate expected variability in human behavior, both within and between individuals. Fundamentals of energy expenditure are then used to estimate relative intensity of activities performed. This, combined with microenvironmental concentrations, allows for the reasonable estimation of the magnitude, frequency, pattern, and duration of exposures an individual experiences.

CHAD is the best source of human activity data for use in exposure modeling. The database contains time-location-activity patterns for individuals of both genders across a wide range of ages (0-99). The database is geographically diverse, containing diaries from individuals residing in major cities, suburban and rural areas across the U.S. Time spent performing activities within particular locations can be on a minute-by minute basis, thus avoiding the smoothing of potential peak exposures longer time periods would give.

There are some limitations to the database, however, many of which are founded in the individual studies from which activity patterns were derived (Graham and McCurdy, 2004). A few questions remain regarding the representativeness of CHAD diaries to the simulated population, such as the numbers of diaries available for use in a simulation (i.e., 20,000 used to represent several million people over long periods of time), the age of diary data (i.e., some data were generated in the 1980s), and diary structure differences (i.e., real-time versus recall method

of data collection). Many of the assumptions about use of these activity patterns in exposure modeling are strengthened by the manner in which they are used by APEX, through focusing on the most important individual attributes that contribute to variability in human behavior (e.g., age, gender, time spent outdoors, day of week, ambient temperature, occupation).

The extent to which the human activity database provides a balanced representation of the population being modeled is likely to vary across areas. Although the algorithm that constructs activity sequences accounts to some extent for the effects of population demographics and local climate on activity, this adjustment procedure may not account for all inter-city differences in people's activities. A new methodology has been developed to more appropriately assign individual diaries to reflect time-location-activity patterns in simulated individuals (discussed further in section 4.5.3). Input distributions used in the new procedure for constructing multi-day activity patterns are based on longitudinal activity data from children of a specific age range (appropriate for this application where similar aged children are simulated), however the data used were limited to one study and may not be appropriate for other simulated individuals. Thus, there are limitations in approximating within-person variance and between-person variance for certain variables (e.g., time spent outdoors). Personal activity patterns are also likely to be affected by many local factors, including topography, land use, traffic patterns, mass transit systems, and recreational opportunities, which are not incorporated in the current exposure analysis approach due to the complexity of scale and lack of data to support the development of a reasonable approach.

4.3.4.4 Modeling Physiological Processes

The modeling of physiological processes that are relevant to the exposure and intake of O_3 is a complicated endeavor, particularly when attempting to capture inter- and intra-personal variability in these rates. APEX has a physiological module capable of estimating ventilation rates (V_E) for every activity performed by an individual, which primarily drives O_3 intake dose rate estimates. See section 2.5 of the draft Exposure Assessment TSD for a discussion of this module. Briefly, the module is based on the relationship between energy expenditure and oxygen consumption rate, thus both within- and between-person variability in ventilation can be addressed through utilization of the unique sequence of events individuals go through each simulated day. These activity-specific V_E estimates, when normalized by BSA, are then used to characterize an individual's exertion level in compiling the summary exposure tables (Table 4-1). One of the key determinants of estimated V_E is the exertion level of an individual's activity, where exertion levels have units of metabolic equivalents of work (MET), which is the ratio of energy expenditure for an activity to the person's basal, or resting, metabolic rate.

There are some limitations in using MET values for this purpose, due mostly to the manner in which the time-location-activity diaries were generated and subsequent estimates of exertion level. An individual (or their caregiver if younger than eight years old) would record the activity performed with a start and end time, with no information on the associated exertion level of the activity. Exertion level (MET) was then inferred by developers of the CHAD database (McCurdy et al., 2000) using standard values and distributions of those values reported by a expert panel of exercise physiologists (Ainsworth et al., 1993). Although this approach allows for an appropriate range of exertion levels to be assigned to the individuals' activities (and to the simulated population), children's activity levels fluctuate widely within a single activity category; their pattern is often characterized as having bursts of high energy expenditure within a longer time frame of less energy expenditure (Freedson, 1989). These fluctuations in energy expenditure that occur within an activity (and thus a simulated event) are not well captured by the MET assignment procedure.

4.4 SCOPE OF EXPOSURE ASSESSMENT

4.4.1 Selection of Urban Areas to be Modeled

The selection of urban areas to include in the exposure analysis takes into consideration the location of O_3 epidemiological studies, the availability of ambient O_3 data, and the desire to represent a range of geographic areas, population demographics, and O_3 climatology. These selection criteria are discussed further in Chapter 5. Based on these criteria, we chose the 12 urban areas listed in Table 4-2 to develop population exposure estimates.[3] The geographic extent of each modeled area consists of the census tracts in the combined statistical area (CSA) as defined by OMB (OMB, 2005).

4.4.2 Time Periods Modeled

The exposure periods modeled are the O_3 seasons for which routine hourly O_3 monitoring data are available. These periods include most of the high-ozone events in each area. The seasons modeled for each area are listed in Table 4-2.

4.4.3 Populations Modeled

Exposure modeling was conducted for the general population residing in each area modeled, as well as for school-age children (ages 5 to 18) and asthmatic school-age children. Due to the increased amount of time spent outdoors engaged in relatively high levels of physical activity (which increases intake), school-age children as a group are particularly at risk for experiencing O_3-related health effects. We report results for school-age children down to age

[3] In the remainder of this chapter the city name in bold in Table 4-2 is used to represent the entire CSA.

Table 4-2. Urban areas and time periods modeled

Urban Area (CSA)	Period modeled
Atlanta-Sandy Springs-Gainesville, GA-AL	March 1 to Oct. 31
Boston-Worcester-Manchester, MA-NH	April 1 to Sept. 30
Chicago-Naperville-Michigan City, IL-IN-WI	April 1 to Sept. 30
Cleveland-Akron-Elyria, OH	April 1 to Oct. 31
Detroit-Warren-Flint, MI	April 1 to Sept. 30
Houston-Baytown-Huntsville, TX	Jan. 1 to Dec. 30
Los Angeles-Long Beach-Riverside, CA	Jan. 1 to Dec. 30
New York-Newark-Bridgeport, NY-NJ-CT-PA	April 1 to Sept. 30
Philadelphia-Camden-Vineland, PA-NJ-DE-MD	April 1 to Oct. 31
Sacramento--Arden-Arcade--Truckee, CA-NV	Jan. 1 to Dec. 30
St. Louis-St. Charles-Farmington, MO-IL	April 1 to Oct. 31
Washington-Baltimore-N. Virginia, DC-MD-VA-WV	April 1 to Oct. 31

five, however, there is a trend for younger children to attend school. Some states allow 4-year-olds to attend kindergarten, and more than 40 states have preschool programs for children younger than five (Blank and Mitchell, 2001). In 2000, six percent of U.S. children ages 3 to 19 who attend school were younger than five years old (2000 Census Summary File 3, Table QT-P19: School Enrollment). We are not taking these younger children into account in our analysis due to a lack of information which would let us characterize this group of children.

The population of asthmatic children is estimated for each city using asthma prevalence data from the National Health Interview Survey (NHIS) for 2003 (Dey and Bloom, 2005). Asthma prevalence rates for children aged 0 to 17 years were calculated for each age, gender, and geographic region. The regions defined by NHIS are "Midwest," "Northeast," "South," and "West." For this analysis, asthma prevalence was defined as the probability of a "Yes" response to the question: "Ever been told that … had asthma?" among those that responded "Yes" or "No" to this question. The responses were weighted to take into account the complex survey design of the NHIS survey. Standard errors and confidence intervals for the prevalence were calculated using a logistic model, taking into account the survey design. A scatter plot smoothing technique using the LOESS smoother was applied to smooth the prevalence curves and compute the standard errors and confidence intervals for the smoothed prevalence estimates. Logistic analysis of the prevalence curves shows statistically significant differences in prevalence by

gender and by region. Therefore we did not combine the prevalence rates for different genders or regions. A detailed description of this analysis is presented in the Exposure Analysis TSD.

The "modeled population" column in Table 4-3 lists the year 2000 populations of the modeled CSAs. The 12 modeled areas combined represent 40 percent of the total U.S. urban population (approximately 222 million in 2000). Table 4-3 also gives the modeled populations of children ages 5-18 and children ages 5-18 characterized as asthmatic.

Table 4-3. Population coverage of modeled areas (2002 analysis)

Urban Area (CSA)	Modeled population (thousands)	Modeled children[1] (thousands)	Asthmatic children (thousands)
Atlanta	4,548	943	117
Boston	5,714	1,096	182
Chicago	9,311	1,951	279
Cleveland	2,945	594	89
Detroit	5,357	1,110	162
Houston	4,815	1,089	136
Los Angeles	16,371	3,667	457
New York	21,357	4,147	643
Philadelphia	5,832	1,186	193
Sacramento	1,930	412	51
St. Louis	2,754	582	83
Washington, DC	7,572	1,485	187
Total of all 12 areas	88,506	18,262	2579

[1] ages 5-18.

4.5 INPUTS TO THE EXPOSURE MODEL

The data inputs to the APEX model are briefly described in this section. A more detailed description of the development of these data and the derivation of input distributions can be found in the Exposure Analysis TSD.

4.5.1 Population Demographics

APEX takes population characteristics into account to develop accurate representations of study area demographics. Population counts and employment probabilities by age and gender are used to develop representative profiles of hypothetical individuals for the simulation. Tract-level population counts by age in one-year increments, from birth to 99 years, come from the

2000 Census of Population and Housing Summary File 1. The Summary File 1 contains the 100-percent data, which is the information compiled from the questions asked of all people and about every housing unit.

Employment data from the 2000 Census provide employment probabilities for each gender and specific age groups for every Census tract. The employment age groupings are: 16-19, 20-21, 22-24, 25-29, 30-34, 35-44, 45-54, 55-59, 60-61, 62-64, 65-69, 70-74, and >75 years of age. Children under the age of 16 are assigned employment probabilities of zero.

4.5.2 Population Commuting Patterns

To ensure that individuals' daily activities are accurately represented within APEX, it is important to integrate working patterns into the assessment. The APEX commuting data are derived from the 2000 Census and collected as part of the Census Transportation Planning Package (CTPP). CTPP contains tabulations by place of residence, place of work, and the flows between the residence and work. These data are available from the U.S. Department of Transportation, Bureau of Transportation Statistics (U.S. Department of Transportation and U.S. Census Bureau, 2000).

For school age children we have not included commuting to and from school. This results in the implicit assumption that children attend a school with ambient O_3 concentrations similar to concentrations near their residence. To the extent that the highest ozone levels are generally in the period June through August when most students are not in school, the absence of school commuting is less likely to have a significant impact on the exposure estimates. As more communities go to year-round schools, school commuting patterns may become important to model.

It is assumed that all persons with home-to-work distances up to 120 km are daily commuters, and that persons who travel further than 120 km do not commute daily. Therefore the list of commuting destinations for each home tract is restricted to only those work tracts that are within 120 km of the home tract.

APEX allows the user to specify how to handle individuals who commute to destinations outside the study area. One option is to drop them from the simulation. If they are included, the user specifies values for two additional parameters, called L_M and L_A (Multiplicative and Additive factors for commuters who Leave the area). While a commuter is at work, if the workplace is outside the study area, then the ambient concentration cannot be determined from any air district (since districts are inside the study area). Instead, it is assumed to be related to the average concentration $C_{AVE}(t)$ over all air districts at the time in question. The ambient concentration outside the study area at time t, $C_{OUT}(t)$, is estimated as:

$$C_{OUT}(t) = L_M * C_{AVE}(t) + L_A \qquad (4\text{-}2)$$

The microenvironmental concentration (for example, in an office outside the study area) is determined from this ambient concentration by the same model (mass balance or factor) as applies inside the study area. The parameters L_M and L_A were both set to zero for this modeling analysis; thus, exposures to individuals are set to zero when they are outside of the study area. Although this tends to underestimate exposures, it is a small effect and this was done since we have not estimated ambient concentrations of O_3 in counties outside of the modeled areas.

4.5.3 Human Activity Data

The human activity data are drawn from the most recent version (December 2000) of the Consolidated Human Activity Database (CHAD) (McCurdy et al., 2000; EPA, 2002), developed and maintained by the Office of Research and Development's (ORD) National Exposure Research Laboratory (NERL). The CHAD includes data from several surveys covering specific time periods at city, state, and national levels, with varying degrees of representativeness. Table 4-4 summarizes the studies in CHAD used in this modeling analysis, providing nearly 16,000 diary-days of activity data (3,075 diary-days for ages 5-18) collected between 1982 and 1998.

A key issue in this assessment is the development of an approach for creating O_3-season or year-long activity sequences for individuals based on a cross-sectional activity data base of 24-hour records. The typical subject in the time/activity studies in CHAD provided less than two days of diary data. For this reason, the construction of a season-long activity sequence for each individual requires some combination of repeating the same data from one subject and using data from multiple subjects. An appropriate approach should adequately account for the day-to-day and week-to-week repetition of activities common to individuals while maintaining realistic variability between individuals. The method in APEX for creating longitudinal diaries was designed to capture the tendency of individuals to repeat activities, based on reproducing realistic variation in a key diary variable, which is a user-selected function of diary variables. For this analysis the key variable is set to the amount of time an individual spends outdoors each day, which is one of the most important determinants of exposure to high levels of O_3.

The actual diary construction method targets two statistics, a population diversity statistic (***D***) and a within-person autocorrelation statistic (***A***). The ***D*** statistic reflects the relative importance of within-person variance and between-person variance in the key variable. The ***A*** statistic quantifies the lag-one (day-to-day) key variable autocorrelation. Desired ***D*** and ***A*** values for the key variable are selected by the user and set in the APEX parameters file, and the method algorithm constructs longitudinal diaries that preserve these parameters. Longitudinal diary data from a field study of school-age children (Geyh et al., 2000) and subsequent analyses (Xue et al., 2004) suggest that ***D*** and ***A*** are stable over time (and perhaps over cohorts as well). Based on these studies of children ages 7-12, appropriate target values for the two statistics for outdoor

Table 4-4. Studies in CHAD used in this analysis

Study name	Geographic coverage	Study time period	Subject ages	Diary-days	Diary-days (ages 5-18)	Diary type and study design	Reference
Baltimore	One building in Baltimore	01/1997-02/1997, 07/1998-08/1998	72 - 93	292	0	Diary	Williams et al. (2000)
California Adolescents (CARB)	California	10/1987-09/1988	12 - 17	181	181	Recall; Random	Robinson et al. (1989), Wiley et al. (1991a)
California Adults (CARB)	California	10/1987-09/1988	18 - 94	1,552	36	Recall; Random	Robinson et al. (1989), Wiley et al. (1991a)
California Children (CARB)	California	04/1989- 02/1990	<1 - 11	1,200	683	Recall; Random	Wiley et al. (1991b)
Cincinnati (EPRI)	Cincinnati metro. area	03/1985-04/1985, 08/1985	<1 - 86	2,587	740	Diary; Random	Johnson (1989)
Denver (EPA)	Denver metro. area	11/1982- 02/1983	18 - 70	791	7	Diary; Random	Johnson (1984), Akland et al. (1985)
Los Angeles: Elementary School	Los Angeles	10/1989	10 - 12	51	51	Diary	Spier et al. (1992)
Los Angeles: High School	Los Angeles	09/1990-10/1990	13 - 17	42	42	Diary	Spier et al. (1992)
National: NHAPS-Air	National	09/1992-10/1994	<1 - 93	4,326	634	Recall; Random	Klepeis et al. (1996), Tsang and Klepeis (1996)
National: NHAPS-Water	National	09/1992-10/1994	<1 - 93	4,332	691	Recall; Random	Klepeis et al. (1996), Tsang and Klepeis (1996)
Washington, D.C. (EPA)	Wash., D.C. metro. area	11/1982-02/1983	18 - 98	639	10	Diary; Random	Hartwell et al. (1984), Akland et al. (1985)
Total diary days				**15,993**	**3,075**		

time are determined to be 0.2 for ***D*** and 0.2 for ***A***. In the absence of data for estimating these statistics for younger children and others outside the study age range, these values are used for all ages. This new method for constructing longitudinal diaries from the CHAD data is described in detail in the Exposure Analysis TSD.

4.5.4 Physiological Data

APEX requires values for various physiological parameters for subjects in order to accurately model their metabolic processes that affect pollutant intake. This is because physiological differences may cause people with the same exposure and activity scenarios to have different pollutant intake levels. The physiological parameters file distributed with APEX contains physiological data or distributions by age and gender for maximum ventilatory capacity (in terms of age- and gender-specific maximum oxygen consumption potential), body mass, resting metabolic rate, and oxygen consumption-to-ventilation rate relationships.

4.5.5 Microenvironments Modeled

In APEX, microenvironments provide the exposure locations for modeled individuals. For exposures to be accurately estimated, it is important to have realistic microenvironments that are matched closely to where people are physically located on a daily and hourly basis. As discussed in section 4.3.2 above, the two methods available in APEX for calculating pollutant concentrations within microenvironments are a mass balance model and a transfer factor approach. Table 4-5 lists the 12 microenvironments selected for this analysis and the exposure calculation method for each. The parameters used in this analysis for modeling these microenvironments are described in this section.

4.5.5.1 Air Exchange Rates for Indoor Residential Environments

Distributions of AERs for the indoor microenvironments were developed using data from several studies. The analysis of these data and the development of the distributions used in the modeling are described in detail in the Exposure Analysis TSD. This analysis showed that the AER distributions for the residential microenvironments depend on the type of air conditioning (A/C) and on the outdoor temperature, as well as other variables for which we do not have sufficient data to estimate. This analysis clearly demonstrates that the AER distributions vary greatly across cities and A/C types and temperatures, so that the selected AER distributions for the modeled cities should also depend upon the city, A/C type, and temperature. For example, the mean AER for residences with A/C ranges from 0.39 for Los Angeles between 30 and 40 ºC to 1.73 for New York between 20 and 25 ºC. The mean AER for residences without A/C ranges from 0.46 for San Francisco on days with temperature between 10 and 20 ºC to 2.29 for

Table 4-5. Microenvironments modeled by APEX

Microenvironment	Calculation Method	Parameters[1]
Indoors – Residence	Mass balance	AER and DE
Indoors – Bars and restaurants	Mass balance	AER and DE
Indoors – Schools	Mass balance	AER and DE
Indoors – Day-care centers	Mass balance	AER and DE
Indoors – Office	Mass balance	AER and DE
Indoors – Shopping	Mass balance	AER and DE
Indoors – Other	Mass balance	AER and DE
Outdoors – Near road	Factors	PR
Outdoors – Public garage/parking lot	Factors	PR
Outdoors – Other	Factors	None
In-vehicle – Cars and Trucks	Factors	PE and PR
In-vehicle – Mass Transit	Factors	PE and PR

[1] AER=air exchange rate, DE=decay-deposition rate, PR=proximity factor, PE=penetration factor

New York on days with temperature between 20 and 25 ºC. The need to account for the city as well as the A/C type and temperature is illustrated by the result that for residences with A/C on days with temperature between 20 and 25 ºC, the mean AER ranges from 0.52 for Research Triangle Park to 1.73 for New York. For each combination of A/C type, city, and temperature with a minimum of 11 AER values, exponential, lognormal, normal, and Weibull distributions were fit to the AER values and compared. Generally, the lognormal distribution was the best-fitting of the four distributions, and so, for consistency, the fitted lognormal distributions are used for all the cases.

One limitation of this analysis was that distributions were available only for selected cities, and yet the summary statistics and comparisons demonstrate that the AER distributions depend upon the city as well as the temperature range and A/C type. Another important limitation of the analysis was that distributions were not able to be fitted to all of the temperature ranges due to limited data in these ranges. A description of how these limitations were addressed can be found in the Exposure Analysis TSD.

City-specific AER distributions were used where possible; otherwise data for a similar city were used. We obtained estimates of A/C prevalence from the American Housing Survey (AHS, 2003) for each metropolitan area. The final AER distributions used for the exposure modeling are given the Exposure Analysis TSD.

Some residences, particularly in the Southwest, use evaporative coolers, also known as "swamp coolers," for cooling. We performed an analysis of AER distributions of residences without A/C, with and without evaporative coolers, using data from three AER measurement studies. This comparison is described in Appendix F in the Exposure Analysis TSD. This analysis showed no improvement in the statistical air exchange model when the data were also stratified by evaporative cooler presence or absence, given that they are already stratified by CSA, air conditioner presence or absence, and outdoor temperature range.

4.5.5.2 AER Distributions for Other Indoor Environments

To estimate AER distributions for non-residential, indoor environments (e.g., offices and schools), we obtained and analyzed two AER data sets: "Turk" (Turk et al., 1989); and "Persily" (Persily and Gorfain, 2004; Persily et al., 2005). The earlier Turk data set includes 40 AER measurements from offices (25 values), schools (7 values), libraries (3 values), and multi-purpose buildings (5 values), each measured using an SF_6 tracer over two or four hours in different seasons of the year. The more recent Persily data were derived from the U.S. EPA Building Assessment Survey and Evaluation (BASE) study, which was conducted to assess indoor air quality, including ventilation, in a large number of randomly selected office buildings throughout the U.S. This data base consists of a total of 390 AER measurements in 96 large, mechanically ventilated offices. AERs were measured both by a volumetric method and by a CO_2 ratio method, and included their uncertainty estimates. For these analyses, we used the recommended "Best Estimates" defined by the values with the lower estimated uncertainty; in the vast majority of cases the best estimate was from the volumetric method.

Due to the small sample size of the Turk data, the data were analyzed without stratification by building type and/or season. For the Persily data, the AER values for each office space were averaged, rather using the individual measurements, to account for the strong dependence of the AER measurements for the same office space over a relatively short period. The mean values are similar for the two studies, but the standard deviations are about twice as high for the Persily data. The proposed AER distributions were derived from the more recent Persily data only.

We fitted exponential, lognormal, normal, and Weibull distributions to the 96 office space average AER values, and the best fitting of these was the lognormal. The fitted parameters for this distribution, used for AER distributions for the indoor non-residential microenvironments, can be found in the Exposure Analysis TSD.

4.5.5.3 Proximity and Penetration Factors for Outdoors and In-vehicle Microenvironments

For the outdoors near-road, public garage/parking lot, and in-vehicle proximity factors, and for the in-vehicle penetration factors, we use distributions developed from the Cincinnati Ozone Study (American Petroleum Institute, 1997, Appendix B; Johnson et al., 1995). This field study was conducted in the greater Cincinnati metropolitan area in August and September, 1994. Vehicle tests were conducted according to an experimental design specifying the vehicle type, road type, vehicle speed, and ventilation mode. Vehicle types were defined by the three study vehicles: a minivan, a full-size car, and a compact car. Road types were interstate highways (interstate), principal urban arterial roads (urban), and local roads (local). Nominal vehicle speeds (typically met over one minute intervals within 5 mph) were at 35 mph, 45 mph, or 55 mph. Ventilation modes were as follows:

- Vent Open: Air conditioner off. Ventilation fan at medium. Driver's window half open. Other windows closed.
- Normal A/C: Air conditioner at normal. All windows closed.
- Max A/C: Air conditioner at maximum. All windows closed.

Ozone concentrations were measured inside the vehicle, outside the vehicle, and at six fixed-site monitors in the Cincinnati area.

The Exposure Analysis TSD documents the distributions and the rationale for the selection of distributions of penetration and proximity factors for outdoors and in-vehicle microenvironments used in this modeling analysis.

4.5.5.4 Ozone Decay and Deposition Rates

A distribution for combined O_3 decay and deposition rates was obtained from the analysis of measurements from a study by Lee et al. (1999). This study measured decay rates in the living rooms of 43 residences in Southern California. Measurements of decay rates in a second room were made in 24 of these residences. The 67 decay rates range from 0.95 to 8.05 $hour^{-1}$. A lognormal distribution was fit to the measurements from this study, yielding a geometric mean of 2.5 and a geometric standard deviation of 1.5. This distribution is used for all indoor microenvironments.

4.5.6 Meteorological Data

Daily average and maximum 1-hour temperatures were obtained from hourly surface temperature measurements obtained from the National Weather Service. APEX uses the data from the closest weather station to each Census tract. Temperatures are used in APEX both in selecting human activity data and in estimating AERs for indoor microenvironments.

4.5.7 Ambient Ozone Concentrations

APEX requires hourly ambient O_3 concentrations at a set of locations in the study area. Data from EPA's AIRS Air Quality Subsystem were used to prepare the ambient air quality input files for 2002, 2003, and 2004. The hourly O_3 concentrations at the AIRS sites in each CSA were used as input to APEX to represent the ambient concentrations within each urban area. For near-road and parking garage microenvironments the ambient concentrations are adjusted by proximity factors, as described in the Exposure Analysis TSD.

4.5.8 Modeling Alternative Standards

In addition to modeling exposures based on historical air quality, an analysis was conducted using air quality representative of just meeting the current 8-hr O_3 standard of 0.08 ppm, and considering the previous rounding convention that allowed for concentrations up to 0.084 ppm. Several alternative standards, reflecting different combinations of standard levels and form were also considered. Alternatives examined were intended to reflect improved precision in the measurement of the ambient concentrations, where the precision would extend to three instead of two decimal places (in ppm) (e.g., 0.080 ppm rather than 0.08 ppm) for several different levels. Differing forms of the standard were also explored outside of the average 4th daily maximum 8-hr average scenario currently used. For example, a 3rd-highest form (the average of the annual 3rd-highest daily maximum 8-hour concentrations averaged over the three year period) was considered for 0.084 and 0.074 ppm levels, and a 5th-highest form for the 0.074 ppm level (the average of the annual 5th-highest daily maximum 8-hour concentrations averaged over the three year period). These alternative scenarios are modeled using a quadratic rollback approach to adjust the hourly O_3 concentrations observed in 2002-2004 to yield a design value corresponding to the standard being modeled. Table 4-6 shows the attainment thresholds (to which the design values are rolled back), the form of the standard used for each scenario, and the notation used in the remainder of this chapter. Design values for the current 8-hr O_3 standard are calculated as the 3-year averages of the annual 4th-highest daily maximum 8-hr average concentration based on the maximum monitor within an urban area. These are given in Table 4-7 for the 2002-2004 period.

The quadratic rollback technique combines both linear and quadratic elements to reduce higher concentrations more than lower concentrations near ambient background levels. The quadratic rollback adjustment procedure was considered in a sensitivity analysis during the last review of the O_3 NAAQS and has been shown to be more realistic than the linear proportional rollback method, where all of the ambient measurements are reduced by a constant multiplicative factor regardless of their individual magnitudes. The quadratic rollback approach and evaluation of this approach are described by Johnson (1997), Duff et al. (1998), and Rizzo (2005, 2006).

Table 4-6. Alternative 8-hr ozone standard scenarios

Attain. Threshold	Form of Standard	Notation
0.084 ppm	3rd-highest form 4th-highest form	84/3 84/4
0.080 ppm	4th-highest form	80/4
0.074 ppm	3rd-highest form 4th-highest form 5th-highest form	74/3 74/4 74/5
0.070 ppm	4th-highest form	70/4
0.064 ppm	4th-highest form	64/4

Table 4-7. 2002-2004 8-hr ozone design values for the modeled areas

Urban Area (CSA)	2002-2004 design value[1] (ppm)	Ratio of 0.084 to the design value
Atlanta	0.093	0.90
Boston	0.091	0.92
Chicago	0.094	0.89
Cleveland	0.095	0.88
Detroit	0.092	0.91
Houston	0.101	0.83
Los Angeles	0.127	0.66
New York	0.094	0.89
Philadelphia	0.094	0.89
Sacramento	0.102	0.82
St. Louis	0.089	0.94
Washington, DC	0.089	0.94

[1] These design values are calculated based on the entire CSA, which in some cases differ slightly from current non-attainment area definitions.

4.6 MODEL EVALUATION, SENSITIVITY, AND UNCERTAINTY ANALYSES

The methods used to evaluate the APEX model and characterize the uncertainty of the model predictions are described in this section. First, we discuss the results of the exposure modeling in comparison to the modeling performed as part of the previous review of the O_3 NAAQS, completed in 1997.

Second, we report the results of a limited APEX model evaluation, which involves the comparison of model exposure estimates to personal exposure measurement data. Although the available exposure measurements are 6-day averages, this comparison can serve to gain insight on whether the model is reasonably estimating exposures.

Another approach for evaluating the model involves model sensitivity analyses, evaluating how the model responds to variation in the input data and parameters used in several of the key algorithms. An analysis such as this is important for several reasons, such as indicating those data that have the greatest impact on estimated exposures and the relative confidence that one has in the model estimates as measured by the degree of certainty in the model inputs and their appropriate use.

A comprehensive analysis of the uncertainties of the exposure modeling was performed, and the results of that analysis are described here. At the end of this section, the conclusions drawn from the evaluation, sensitivity, and uncertainty analyses are summarized.

4.6.1 Comparison with Exposure Estimates from the Prior Review

There have been significant improvements to the exposure model and the model inputs since the review in 1997, as discussed in Section 4.3.1. In the previous review, six urban areas were modeled using the pNEM model, Houston, Los Angeles, New York, Philadelphia, St. Louis, and Washington (U.S. EPA, 1996a,c). These six cities (as well as six others) are also modeled in the current review, although the geographic areas modeled are larger than in the previous review, with over twice the population coverage. When modeling a larger area, extending well beyond the urban core, there will be more people exposed, but a smaller percentage of the modeled population will be exposed at high levels, if O_3 concentrations are lower in the extended areas. Typical years, in terms of O_3 air quality, were modeled in the 1997 review (1990 for some cities and 1991 for others). The only alternative standard for which we have results for both reviews is the "84/3" standard. Exposures to children who tend to spend more time outdoors were estimated in the previous review but not in the current review, and there is no population group for which we can make a direct comparison of the exposure estimates for the two reviews. Therefore, a quantitative comparison of the exposure results is not appropriate.

4.6.2 Comparison of Model Estimates with Measured Personal Exposures

APEX simulation results were compared to personal O_3 concentration measurements obtained from the Harvard Southern California Chronic Ozone Exposure Study (Xue et al. 2004, Geyh et al. 2000). Although this study was limited in scope, and the measurements of ozone are averaged over 6 or 7 days, it is the only study found that measured enough personal exposures to O_3 to be useful for this evaluation and for which the data are available. In this study, children 7 to 12 years old were monitored from June 1995 to May 1996. There were 160 subjects on which longitudinal O_3 concentrations were collected in at least 6 of the 12 months of the study period. Passive O_3 samplers were used to measure 6-day or 7-day average personal O_3 concentrations, as well as indoor and outdoor concentrations at participants' homes, for six days each month. The subjects resided in two separate areas of San Bernardino County: urban Upland CA, and the small mountain towns of Lake Arrowhead, Crestline, and Running Springs, CA. There was a total of 91 6-day or 7-day periods with measurements used in this evaluation.

For the APEX simulations we used the same model inputs as for the Los Angeles simulations, described in Section 4.5 above, except for the air exchange rates. The AERs used were those developed for Sacramento from measurements taken in Sacramento and the inland portions of the Los Angeles area: Riverside and San Bernardino Counties. The hourly outdoor O_3 concentrations were from fixed site monitors located in Upland and Crestline.

For each period for which personal measurements were available we simulated 10,000 subjects in the 7 – 12 age range in each of the two study areas. For each case the distribution of simulated 6-day or 7-day average exposure concentrations was compared to the corresponding distribution of measured values, which ranged from 8 to 31 subjects. Comparisons were also made between the continuous measurements made inside the subjects' homes and the APEX indoor residential concentration estimates during the times of exposure, and between the O_3 concentrations measured outside the homes of the study subjects and those measured at the nearby fixed site monitors.

Figures 4-2 and 4-3 show the ranges and means of the population distributions of modeled exposures from APEX and the ranges and means of the 8 to 31 measured personal exposures for each 6-day or 7-day period, for Upland and for Lake Arrowhead, Crestline, and Running Springs.

Variability in exposure is reasonably estimated by APEX across the population, although the APEX predictions tend to have a lower variance than the measured values. The mean

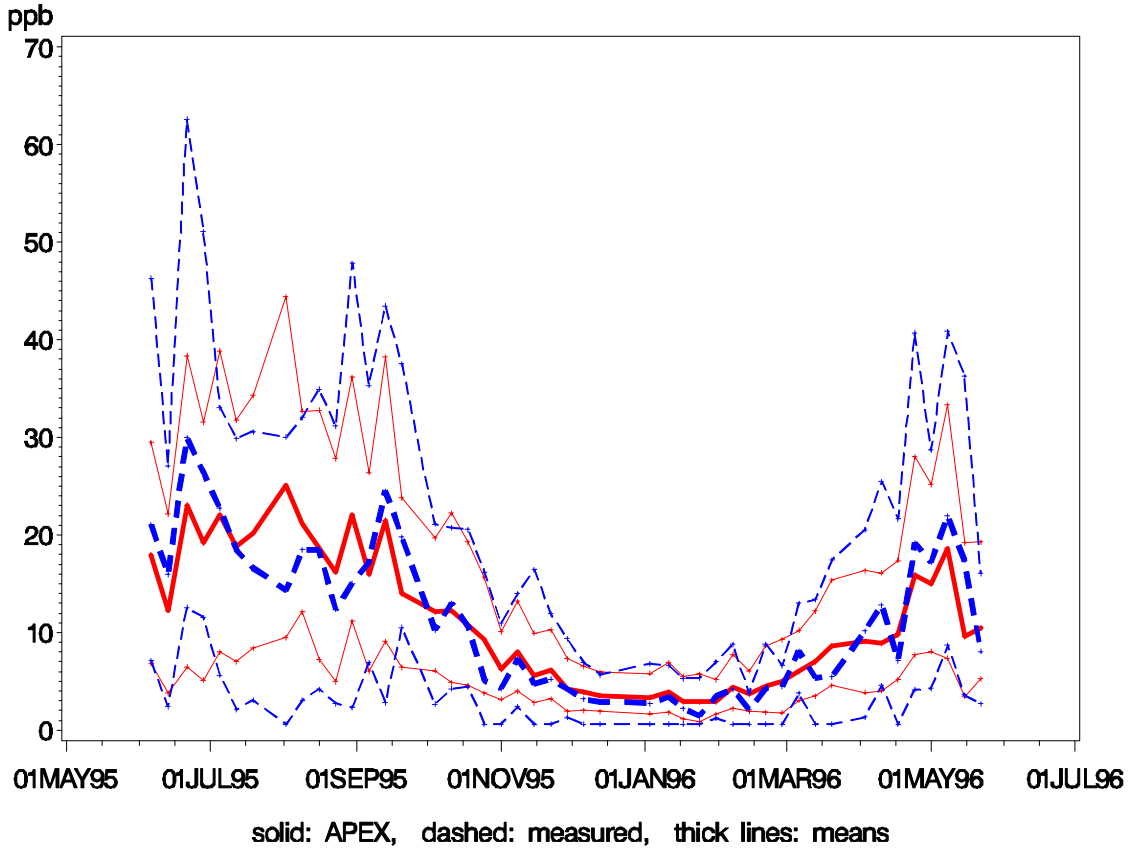


Figure 4-2. Modeled and measured weekly average personal exposures: means, minima, maxima for each week. Upland, CA.

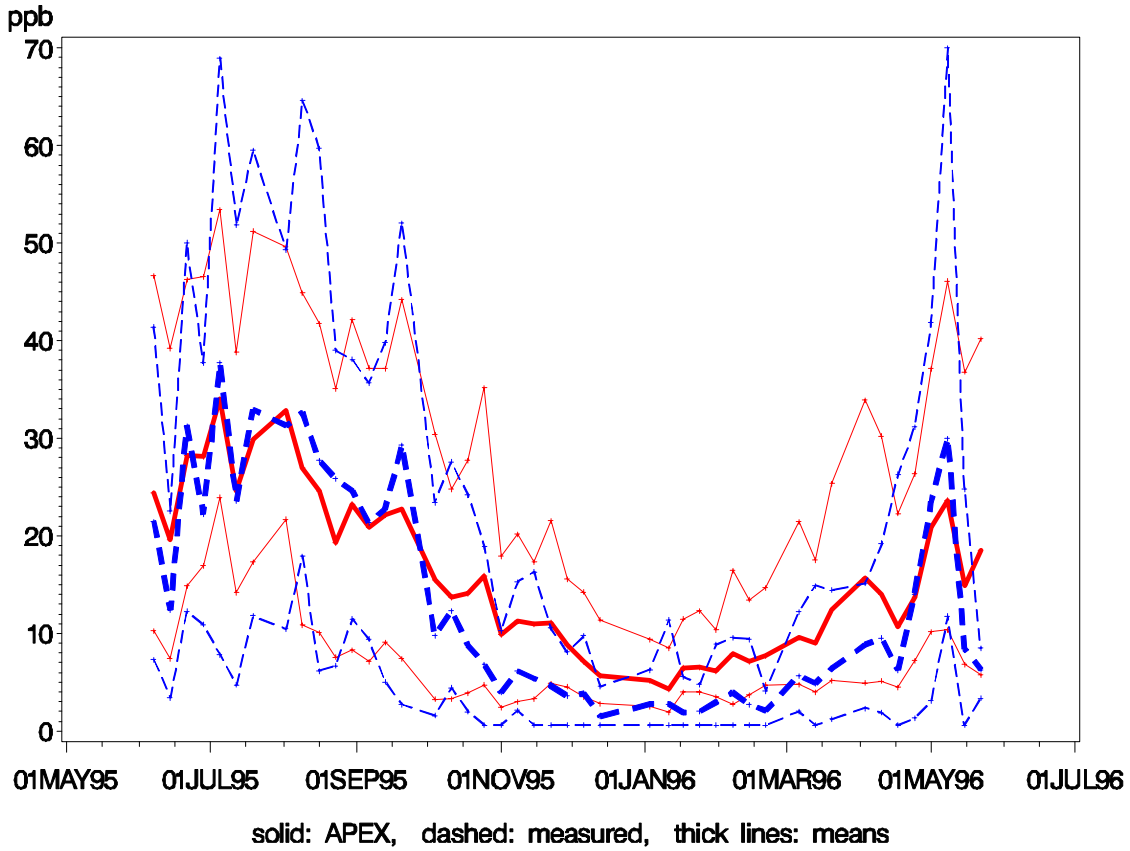


Figure 4-3. Modeled and measured weekly average personal exposures: means, minima, maxima for each week. Lake Arrowhead, Crestline, and Running Springs, CA.

exposures predicted by APEX are generally in good agreement with the measured means, except for the Winter and Spring seasons for the Lake Arrowhead, Crestline, and Running Springs communities, when the exposures are very low. It is not clear how these results might translate to model performance of 8-hour average exposures, although the underprediction of variance indicates that APEX may be underpredicting the upper quantiles of the 8-hour exposures. Additional results and analyses of the evaluation are discussed in the Exposure Assessment TSD. Since this evaluation is based on 6-day and 7-day average exposures, it is only of limited relevance for evaluating daily maximum 8-hr average exposure simulation results.

4.6.3 Sensitivity Analyses

The sensitivity analyses most relevant to evaluating APEX are summarized here. Additional sensitivity analyses are described in the Exposure Assessment TSD.

4.6.3.1 Near-Road Residential Exposures

APEX does not take into account the effects of mobile source emissions of NO on the concentrations of O_3 in residences near roadways; therefore, we conducted a sensitivity analysis to assess the potential effect on exposures in residences of the titration of O_3 by mobile source NO. We performed APEX simulations for the Boston and Houston 2002 base cases for each of these three subsets of the population, defined by the distance of their residence from a major roadway, based on the fractions of the population in each Census tract that live in three bands:

a) 0-75 m from a major roadway,
b) 75-200 m from a major roadway, and
c) >200 m from a major roadway.

We used proximity factors to decrease the ambient concentrations outside their residences in accordance with the distance from roadways. The combined results of these three simulations account for the decreased exposures in residences near roadways. A comparison of these model results with the standard simulations show decreases of one to three percent in the estimated counts of exposures above 0.07 ppm-8hr. This demonstrates that the uncertainty engendered by not accounting for the titration of O_3 by NO in residences near roadways is quite small for high exposures. Since high exposures are generally associated with outdoor and not indoor activities, this is not surprising.

4.6.3.2 Air Exchange Rates and the Prevalence of Residential Air Conditioning

Air exchange rates and the prevalence of residential air conditioning are two of the influential determinants of exposures while indoors, and while the uncertainty of these two model input parameters are accounted for in the uncertainty analysis described below, it is possible that these inputs might be biased, and this sensitivity analysis assessed the effect of

potential biases in the values used in the APEX exposure modeling. APEX simulations for the Boston 2002 base case and current standard scenarios were performed, varying the air conditioning prevalence rate from 85% to 70% and to 50%, and also varying the geometric means of all of the AER distributions simultaneously by +0.1 and by −0.1. Biases of these magnitudes in these parameters turn out to have little effect on the high 8-hour average exposures predicted by APEX. For example, changes in the percents of children experiencing exposures over 0.06, 0.07, and 0.08 ppm-8hr change by less than one percentage point.

4.6.3.3 Activity Patterns: Representativeness of CHAD

Many of the studies included in the CHAD data base are not national in scope, nor do they necessarily correspond to the modeled urban areas, and we conducted an analysis to assess how similar the exposure results are when using individual component studies. Strong similarity would suggest that extrapolation of activity data gathered from one sample population to another population is appropriate. The largest and most comprehensive individual study in CHAD is the National Human Activity Pattern Study (NHAPS), and we compared the APEX exposure results using all of CHAD with corresponding results using only the NHAPS data. NHAPS is national in scope, with a random design, and comprises more than half of the CHAD diaries for all ages, and 43 percent of the diary days in CHAD for children ages 5 to 18. Sensitivity analyses conducted using other subsets of CHAD are described in the Exposure Assessment TSD.

APEX simulations were performed using only NHAPS diaries for all 12 urban areas, for the 2002 base case and the scenario of meeting the current NAAQS. The results of this comparison for the 2002 base case simulations are presented in Table 4-8 for the percent of children at moderate exertion with 8-hour exposures above exposure levels of 0.06, 0.07, and 0.08 ppm-8hr. The comparison of estimated reductions in exposures to children at moderate exertion in going from the base case to the current standard is presented in Table 4-9.

There is good agreement between the APEX predictions of the reductions in exposures from the current base case to the current standard, whether all of CHAD or only the NHAPS component of CHAD is used, indicating that these results are not being unduly influenced by any single study in CHAD. The differences are larger for the absolute percentages, reflecting the effect of the uncertainties of the CHAD database.

4.6.3.4 Activity Patterns: Underestimation of Repeated Exposures

Not only is the actual exposure level important for the development of adverse health outcomes but also the frequency of exposures at given levels of concern. In the absence of specific data to directly evaluate the repeated exposure results generated by APEX,

Table 4-8. Comparison of APEX 2002 base case simulations: All CHAD vs. the NHAPS part of CHAD. Percent of children at moderate exertion with 8-hour exposures above levels of 0.06, 0.07, 0.08 ppm-8hr.

	Above 0.06 ppm-8hr			Above 0.07 ppm-8hr			Above 0.08 ppm-8hr		
CSA	All CHAD	NHAPS only	difference	All CHAD	NHAPS only	difference	All CHAD	NHAPS only	difference
Atlanta	68%	62%	(6%)	42%	38%	(4%)	15%	14%	(1%)
Boston	66%	60%	(6%)	46%	41%	(5%)	24%	21%	(2%)
Chicago	69%	61%	(8%)	44%	38%	(6%)	17%	16%	(1%)
Cleveland	77%	71%	(6%)	63%	56%	(7%)	38%	33%	(4%)
Detroit	72%	65%	(7%)	51%	45%	(7%)	20%	20%	0%
Houston	58%	51%	(7%)	31%	28%	(3%)	13%	12%	(0%)
Los Angeles	61%	58%	(3%)	36%	33%	(2%)	16%	16%	0%
New York	74%	68%	(6%)	54%	48%	(6%)	28%	25%	(2%)
Philadelphia	77%	69%	(8%)	60%	53%	(7%)	38%	33%	(5%)
Sacramento	66%	59%	(7%)	39%	35%	(3%)	15%	15%	0%
St. Louis	70%	61%	(9%)	51%	43%	(8%)	21%	20%	(1%)
Washington	73%	66%	(7%)	53%	48%	(6%)	29%	25%	(3%)

Table 4-9. Comparison of APEX simulations: All CHAD vs. the NHAPS part of CHAD. Percent reduction[1] from the 2002 base case to the current standard of the number of children at moderate exertion with 8-hour exposures above levels of 0.06, 0.07, 0.08 ppm-8hr.

	Above 0.06 ppm-8hr			Above 0.07 ppm-8hr			Above 0.08 ppm-8hr		
CSA	All CHAD	NHAPS only	difference	All CHAD	NHAPS only	difference	All CHAD	NHAPS only	difference
Atlanta	22%	22%	(0%)	51%	52%	1%	71%	67%	(4%)
Boston	18%	19%	1%	38%	37%	(1%)	57%	56%	(1%)
Chicago	24%	25%	1%	55%	50%	(5%)	85%	81%	(4%)
Cleveland	14%	17%	3%	39%	41%	1%	81%	78%	(2%)
Detroit	15%	17%	2%	44%	38%	(5%)	85%	81%	(5%)
Houston	53%	51%	(2%)	76%	73%	(3%)	91%	91%	0%
Los Angeles	88%	85%	(3%)	98%	96%	(1%)	100%	99%	(0%)
New York	32%	34%	1%	70%	67%	(3%)	91%	89%	(2%)
Philadelphia	17%	18%	1%	39%	39%	0%	71%	71%	(0%)
Sacramento	49%	47%	(2%)	81%	78%	(3%)	93%	93%	1%
St. Louis	9%	11%	2%	28%	24%	(4%)	53%	48%	(5%)
Washington	19%	20%	1%	45%	46%	0%	74%	75%	1%

[1] The percent reductions are calculated as 100(base case results – current standard results)/(base case results).

the population of outdoor workers was targeted as a group that can be used indirectly for this type of evaluation. To this end, a comparison of estimated exposures of outdoor workers in two urban areas (Atlanta and Sacramento) with exposures estimated by APEX for those urban areas was performed.

Due to data limitations, APEX has not been set up to specifically model outdoor workers, but can estimate exposures for all employed adults in an urban area. Comparison of estimates of repeated exposures to outdoor workers with the corresponding APEX estimates for all workers reveals that APEX significantly underestimates the number of multiple exposures for working adults. For example, in Atlanta, APEX estimates 490,000 workers to be exposed at least once to levels above 0.07 ppm-8hr in 2002; however, only 220 were estimated to experience repeated exposures above 0.07 ppm-8hr six or more times. Because there is no information on the exertion levels of outdoor workers, these estimates include all exertion levels and are not restricted to moderate or greater exertion. In contrast to the APEX estimates, a separate estimate of exposures to outdoor workers, based on outdoor worker frequency estimates for occupation categories, gives a range of from 62,000 to 140,000 outdoor workers experiencing six or more repeated exposures to levels above 0.07 ppm-8hr in Atlanta for 2002 air quality. Table 4-10 summarizes the comparisons performed for Atlanta and Sacramento for repeated exposures to levels above 0.06, 0.07, and 0.08 ppm-8hr. A description of the method used for estimating these exposures of outdoor workers is given in Langstaff (2007). Since outdoor workers are a subset of all employed adults, it is clear from Table 4-10 that APEX is underestimating repeated exposures to adult workers.

Table 4-10. Comparison of estimated outdoor workers' repeated exposures with APEX results for all workers, in Atlanta and Sacramento, 2002. Numbers of people with at least six repeated 8-hour exposures above 0.06, 0.07, and 0.08 ppm-8hr.[1]

	# above 0.06 ppm-8hr		# above 0.07 ppm-8hr		# above 0.08 ppm-8hr	
	Est. outdoor workers	APEX all workers	Est. outdoor workers	APEX all workers	Est. outdoor workers	APEX all workers
Atlanta	63,000 – 150,000	74,000	62,000 – 140,000	220	41,000 – 94,000	0
Sacramento	30,000 – 61,000	30,000	27,000 – 55,000	95	21,000 – 42,000	0

[1] The numbers in this table have been rounded to two significant digits.

This underestimation results primarily from the way that people's activities are modeled using CHAD, which does not properly account for repeated behavior of individuals. The new longitudinal methodology does increase the similarity of daily activities for a given simulated individual in terms of the time spent outdoors, and some simulated individuals tend to spend

more time outdoors than others, compared to a more random assignment of diaries from CHAD to modeled individuals. However, repeated routine behavior from one weekday to the next is not simulated. For example, there are no simulated individuals representing children in summer camps who spend a large portion of their time outdoors, or adults with well-correlated weekday schedules. These limitations apply to both children and adults, and therefore multiple exposures to children are also expected to be underestimated by APEX.

4.6.4 Uncertainty Analysis

An understanding of the uncertainty of the APEX model predictions has been developed through sets of complementary analyses addressing different aspects of the overall uncertainty. A Monte Carlo analysis was performed which accounts for most of the uncertainties of the APEX model inputs. Sensitivity analyses have been conducted to address the potential influence of other sources of uncertainty, described in the previous section. This section provides a summary of the results of the exposure modeling uncertainty analysis; the details of the uncertainty analysis are described in Langstaff (2007).

A Monte Carlo approach was selected for a detailed uncertainty analyses. Monte Carlo methods for analysis of model uncertainty use statistical sampling techniques to estimate statistics which characterize uncertainty. Essentially, a Monte Carlo approach entails performing many model runs with model inputs randomly sampled from distributions reflecting the uncertainty of the inputs. This propagates the uncertainty of the model inputs through to the model results, taking into account input parameter dependencies and the interaction of uncertainties within the model. These simulations provide uncertainties of model results in terms of uncertainty distributions of the model outputs. From these we calculate 95 percent uncertainty intervals (UI) for a particular model result as the interval from the 2.5^{th} to the 97.5^{th} percentile of the uncertainty distribution for that result.

The Monte Carlo uncertainty analysis performed accounts for the following sources of uncertainty:

- Ambient air concentrations measurement error
- Spatial interpolation of ambient concentrations
- Air exchange rates
- Air conditioning prevalence rates
- Ozone deposition and decay rates
- Vehicle penetration factors
- Longitudinal diary assembly parameters
- Metabolic equivalents (MET)
- Model convergence

The Monte Carlo uncertainty analysis was performed for Boston 2002, for the recent year base case and the current standard scenarios, as well as for the estimated reductions in exposures in going from the base case to the current standard. Uncertainties of model results for other areas and years are expected to be similar.

Figure 4-4 illustrates the uncertainty distributions for one model result, the percent of children with exposures above 0.06 ppm-8hr while at moderate exertion. This distribution results from approximately 2000 Monte Carlo APEX simulations of the Boston 2002 base case with model inputs varied randomly according to their uncertainty. The "point estimate" of 62 percent is the result from the APEX simulation using our best estimates of the model inputs, as described in Section 4.5. The corresponding result from the Monte Carlo simulations ranges from 56 to 67 percent, with a 95 percent UI of 58 to 65 percent. Figure 4-5 and Figure 4-6 illustrate the uncertainty distributions for two other model results, the percents of children with exposures above 0.07 and 0.08 ppm-8hr while at moderate exertion.

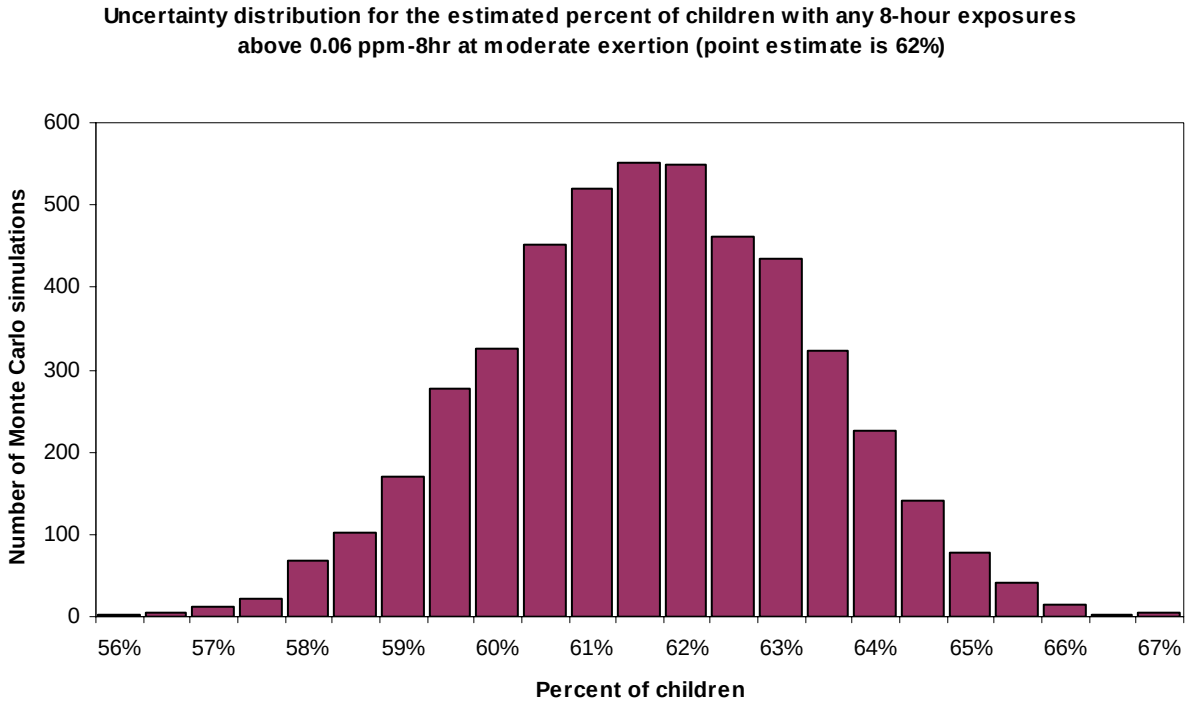


Figure 4-4. Uncertainty of percent of children with exposures above 0.06 ppm-8hr (Boston 2002 base case)

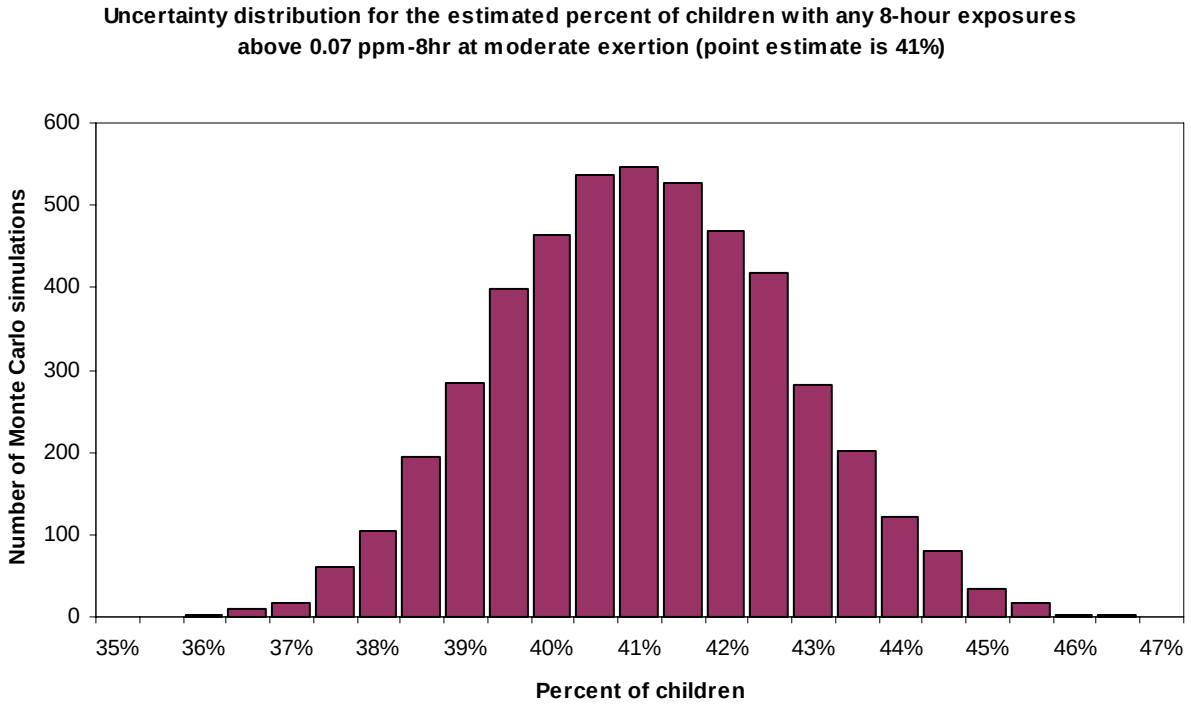


Figure 4-5. Uncertainty of percent of children with exposures above 0.07 ppm-8hr (Boston 2002 base case)

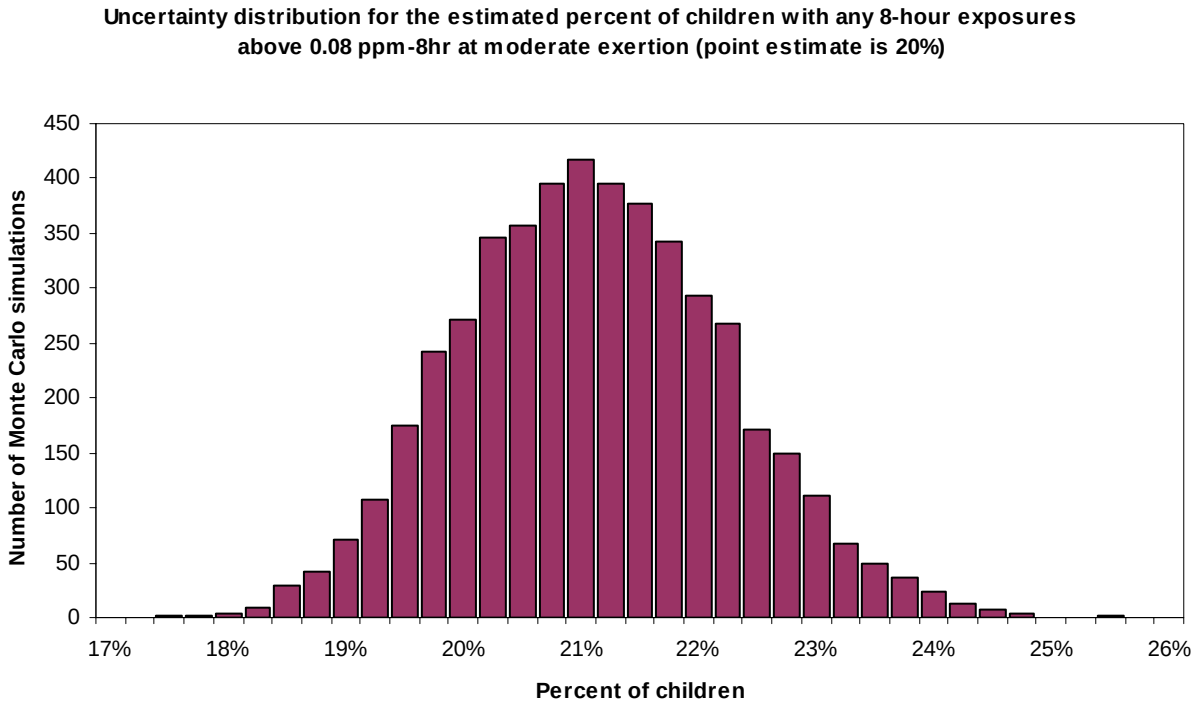


Figure 4-6. Uncertainty of percent of children with exposures above 0.08 ppm-8hr (Boston 2002 base case)

Uncertainty intervals are presented in Table 4-11 and Table 4-12 for the estimated percentages of all children and asthmatic children with exposures above different 8-hour exposure levels under moderate exertion. The UIs for the estimated reductions in exposures, going from the 2002 base case to the current standard, for these two groups are given in Table 4-13. Across these three tables, the spans of the 95 percent UIs range from 2 to 10 percentage points, and the point estimates are generally within 5 percentage points of the UI endpoints. The uncertainties of the exposures to asthmatic children are slightly higher than for all children. These results are very positive, and the modeling uncertainty is small enough to lend confidence to the use of the model results.

Table 4-11. Uncertainty of the estimated percent of children exposed at moderate exertion, Boston, 2002

Exposure level (ppm-8hr)	Air quality scenario	Point estimate	95% UI
0.06	base case	62%	58-65%
0.07	base case	41%	38-44%
0.08	base case	20%	19-24%
0.06	current standard	49%	46-52%
0.07	current standard	24%	23-27%
0.08	current standard	8.5%	8-10%

Table 4-12. Uncertainty of the estimated percent of asthmatic children exposed at moderate exertion, Boston, 2002

Exposure level (ppm-8hr)	Air quality scenario	Point estimate	95% UI
0.06	base case	65%	60-67%
0.07	base case	43%	39-46%
0.08	base case	21%	19-25%
0.06	current standard	52%	48-56%
0.07	current standard	24%	23-30%
0.08	current standard	9%	8-11%

Table 4-13. Uncertainty of the estimated percent reduction, from the base case to the current standard, of all children and asthmatic children exposed at moderate exertion, Boston, 2002

	All children		Asthmatic children	
Exposure level (ppm-8hr)	Point estimate	95% UI	Point estimate	95% UI
0.06	21%	18-22%	19%	16-22%
0.07	41%	38-42%	43%	37-45%
0.08	58%	55-59%	58%	53-63%

4.6.5 Key Findings

Uncertainty of the APEX model predictions results from uncertainties in the spatial interpolation of measured concentrations, the microenvironment models and parameters, people's activity patterns, and, to a lesser extent, model structure. The predominant sources of uncertainty appear to be the activity pattern information and the spatial interpolation of ambient concentrations from monitoring sites to other locations. The primary findings of these analysis are the following:

- The Monte Carlo analysis of the uncertainties of the APEX model estimates of exposure distributions indicates that the uncertainty is relatively small. The APEX estimates of the percent of children or asthmatic children with exposures above 0.06, 0.07, or 0.08 ppm-8hr under moderate exertion have 95% uncertainty intervals of at most ±6 percentage points.
- An investigation into the representativeness of the CHAD activity diaries with respect to the specific urban areas and time periods modeled indicates uncertainties of only a few percent in the APEX estimates of the numbers of children with exposures above 0.06, 0.07, or 0.08 ppm-8hr under moderate exertion.
- Although the effect on exposures in residences of the titration of O_3 by mobile source NO is not explicitly modeled by APEX, the resulting uncertainty is small, on the order of 1 to 3 percent.
- APEX significantly underestimates the frequency of occurrence of individuals experiencing repeated 8-hour average exposures greater than 0.06, 0.07, and 0.08 ppm-8hr. The reasons for this are understood, and further research will be required to address this.

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4.7 EXPOSURE ASSESSMENT RESULTS

4.7.1 APEX Modeling Results

The results of the exposure analysis[4] are presented as a series of exhibits and graphs focusing on a range of benchmark levels, described in Chapters 3 and 6, as being of particular health concern. In addition, a wide range of concentrations in the air quality data collected over the three year period (2002-2004) were used in the exposure model, providing a broad range of estimated exposures output by the model. Exposure results are presented for the range of alternative standard scenarios given in Table 4-6. Estimates of exposures for the year 2003 were developed since the second draft of this Staff Paper for only two alternative standard levels (74/4 and 64/4) due to time constraints. This section is organized into two main subsections, the first addressing the exposures estimated for each of the particular benchmarks and the second reporting on the estimates of repeated exposures. Tables of exposures for children and asthmatic children under moderate exertion are presented in Appendix 4-A.

4.7.2 Estimated Exposures above Selected Benchmark Levels

A series of exhibits are presented for each of the benchmark levels of persons who experience daily maximum 8-hour average exposures above 0.080, 0.070, and 0.060 ppm-8hr. A few notes regarding the exhibits are necessary to mention. Exposure estimates are presented for those individuals experiencing moderate levels of exertion during the same 8-hr period that the exposure occurred. The exertion level is characterized by breathing rates, as described in section 4.3.3. Results for children exposed to O_3 while engaged in moderate exertion are presented in each of the subsequent exhibits, however results for any other population group could have been presented with similar exposure outcomes and patterns across the 12 cities modeled. For example, the comparison of three population groups, children, asthmatic children, and all persons, indicates a very similar pattern of exposure estimates, regardless of the subgroup considered (Figure 4-7). In addition, use of the multiple years of ambient air quality data generated a range of exposure concentrations, bracketed by the year 2002 (highest exposure estimates) and the year 2004 (lowest exposure estimates). Exposure estimates for year 2003 generally fell in between the exposures estimated for the other two years (Figure 4-8).

[4] An error was found in the exposure model in January 2007. This error has been corrected and the model runs have been redone, generally resulting in small increases in the exposure estimates. The corrected results are presented in this Staff Paper and in the Exposure Analysis TSD.

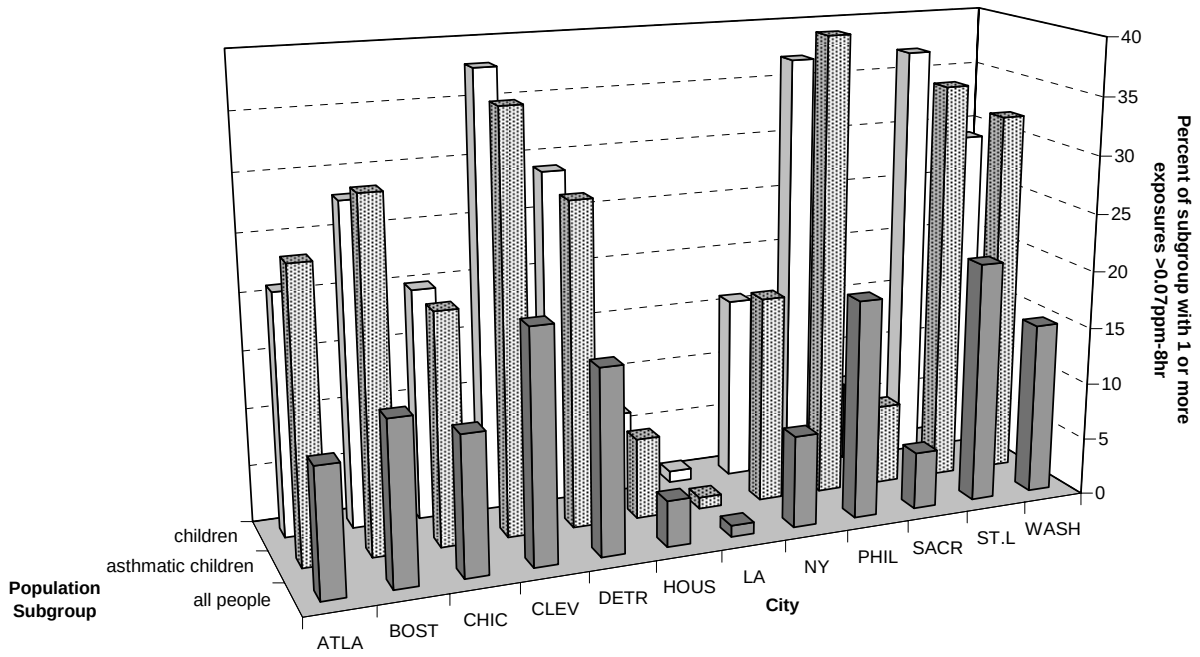


Figure 4-7. Comparison of population groups. Percent of population subgroups with one or more 8-hr ozone exposures above 0.070 ppm-8hr, concomitant with moderate or greater exertion, with just reaching the current standard, based on 2002 ambient air quality.

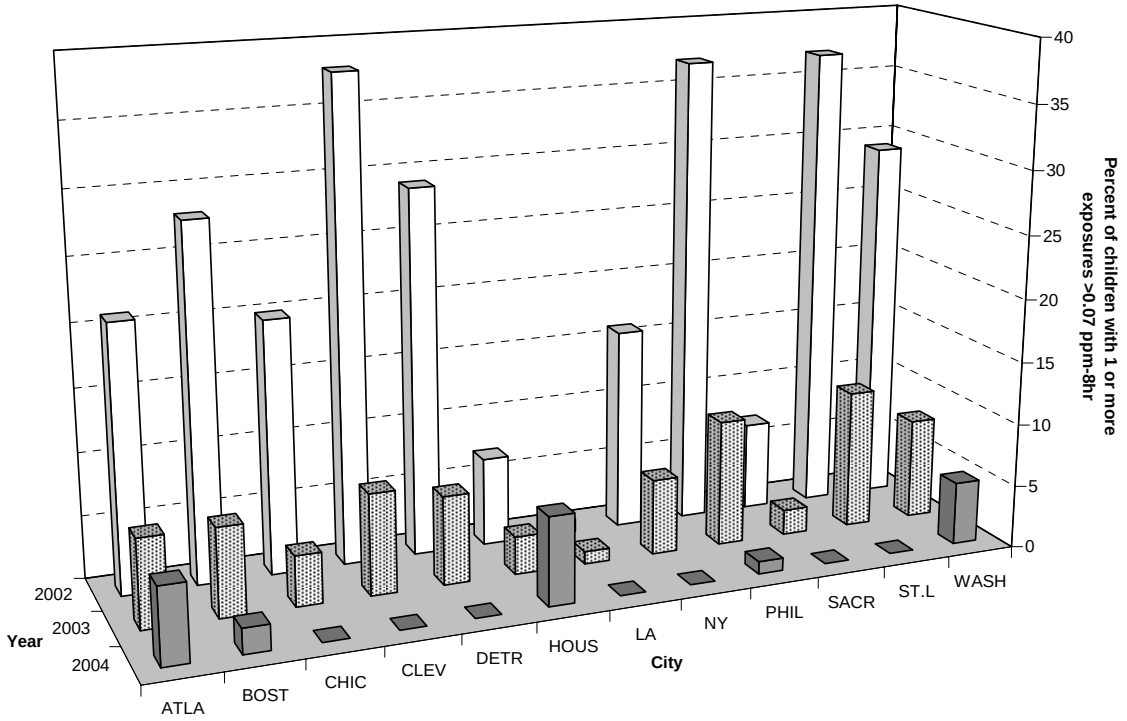


Figure 4-8. Comparison of years. Percent of children with one or more 8-hr ozone exposures above 0.07 ppm-8hr, concomitant with moderate or greater exertion, with just reaching the current standard.

Nine exhibits follow, representing the exposures estimated for years 2002, 2003, and 2004, each at the three benchmark levels (0.080, 0.070, and 0.060 ppm-8hr). Each exhibit contains three exposure metrics: the estimated percent of children exposed, the estimated number of children exposed, and the number of person-days exposure occurred above the particular benchmark level. The notation used for the alternative standards is defined in Table 4-6. Exposures were also estimated using the respective year air quality data without application of the rollback procedure, and are presented as the "base" scenario in each exhibit. The numbers for the base scenarios are omitted from some of the figures within the exhibits (i.e., the numbers of children and person days) since they increased the vertical scale, obscuring the details of interest.

The patterns of estimated exposures are variable from city to city, primarily due to differences in air quality (local emissions and meteorology affect these), the rollback procedure as applied to each separate area, and people's time-location-activity patterns. The year-to-year variability in exposures is also evident. All of the urban areas modeled except Houston and Los Angeles have fewer exposures in 2003 and 2004 than in 2002, due in varying degrees to changes in weather and emissions of precursors to O_3.

Inspection of these exhibits shows marked differences between urban areas in the levels of exposures under alternative standards. For example, under the same 0.074 ppm, 4^{th} daily maximum 8-hr average alternative standard, it is estimated that 10 percent of the Boston children but very few of the Los Angeles children experience 8-hr O_3 exposures above 0.070 ppm-8hr while engaged in moderate exertion using 2002 air quality. This is primarily due to the larger range of 2002-2004 4^{th}-highest concentrations for Boston compared to the rest of its air quality distribution, which in general, allowed for retention of higher concentrations within the air quality distribution (and therefore exposures) following the rollback of the air quality. In Los Angeles, much more of the upper range of the air quality distribution needed to be rolled back to allow for the meeting of the alternative standards, thus significantly reducing the frequency of occurrence of high ambient concentrations (and therefore exposures).

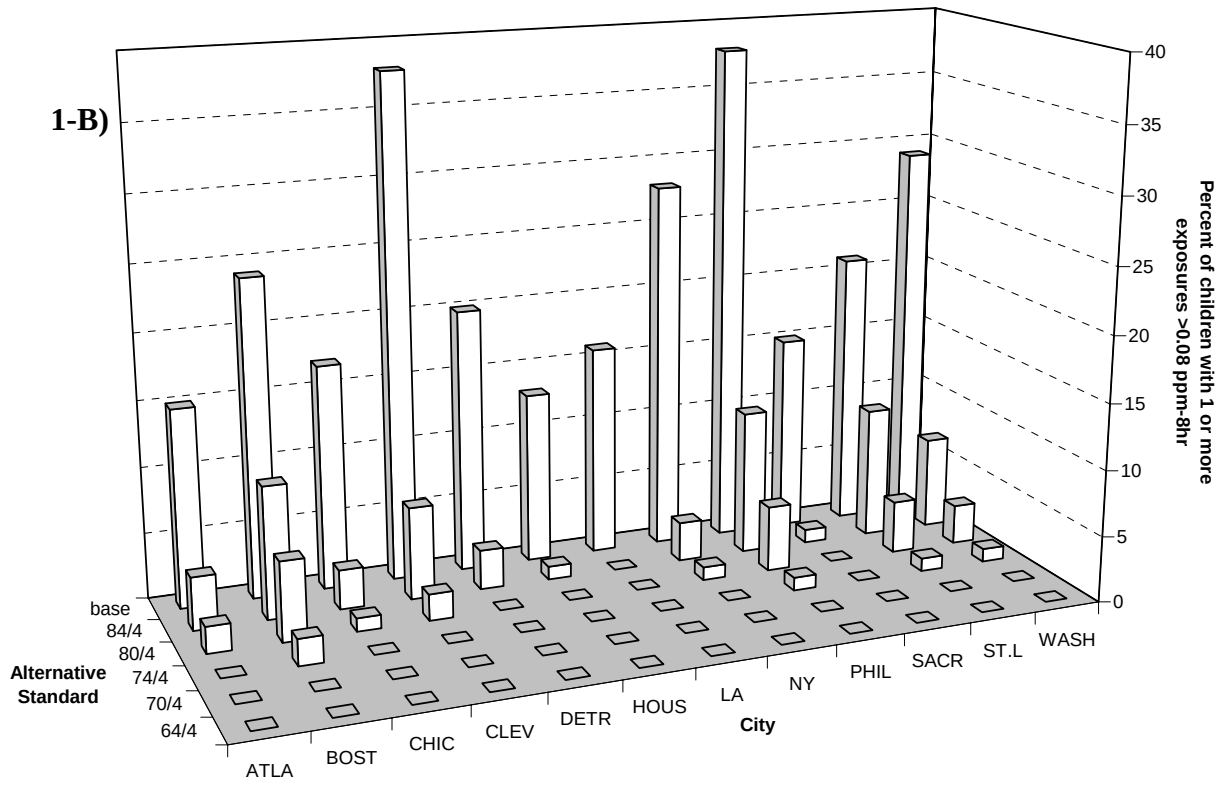
The form of the standard had an impact on the number of estimated exposures, the general magnitude of which is dependent on the benchmark level selected and year of air quality data used. For example in considering the series of 74/x scenarios (74/5, 74/4, 74/3), the impact could be a low as a few percent (percent of children with estimated exposures above 0.060 ppm using 2004 air quality go from, on average, 3% to 1%) or much greater (percent of children with estimated exposures above 0.060 ppm using 2002 air quality go from, on average, 28% to 21%)

Exhibit 1. Summary of exposure metrics regarding estimated exceedances of 0.080 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2002 air quality. A) Table of percent of children with at least one exposure above 0.080 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.080 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.080 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.080 ppm 8-hr.

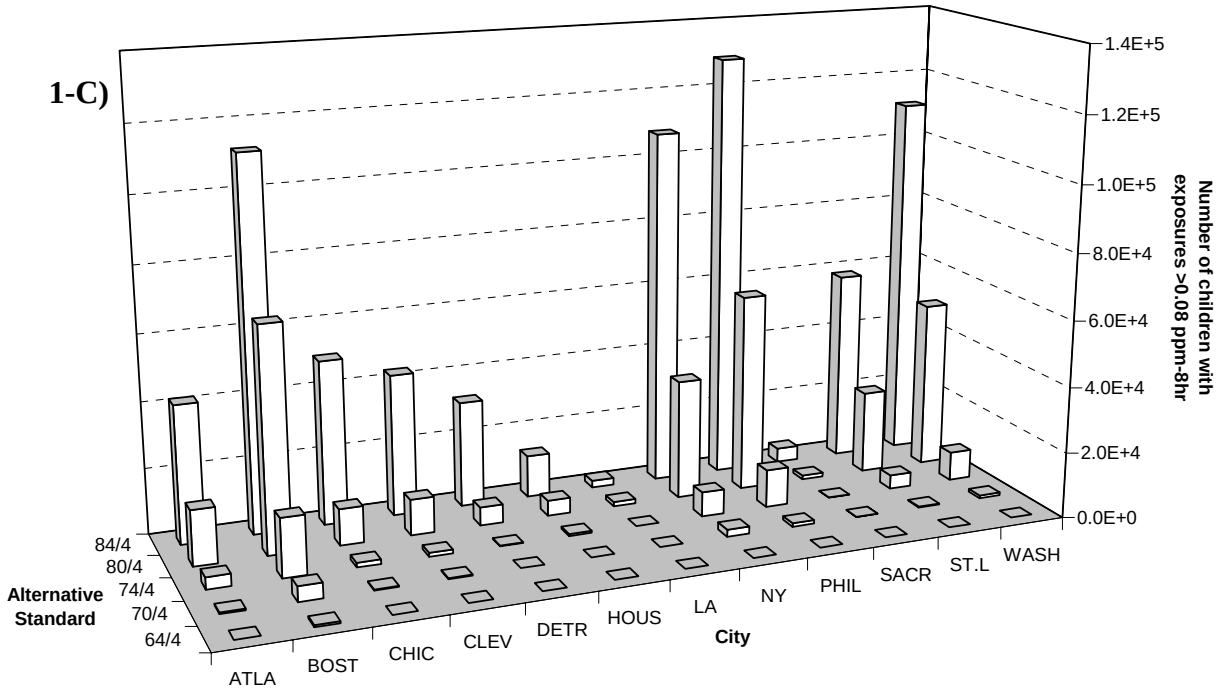
1-A) Percent of children with exposures > 0.080 ppm, moderate exertion, 2002 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	15%	4%	4%	2%	1%	0%	0%	0%	0%
Boston	24%	10%	6%	6%	5%	2%	1%	0%	0%
Chicago	17%	3%	2%	1%	0%	0%	0%	0%	0%
Cleveland	38%	7%	2%	2%	0%	0%	0%	0%	0%
Detroit	20%	3%	1%	0%	0%	0%	0%	0%	0%
Houston	13%	1%	1%	0%	0%	0%	0%	0%	0%
Los Angeles	16%	0%	0%	0%	0%	0%	0%	0%	0%
New York	28%	3%	2%	1%	0%	0%	0%	0%	0%
Philadelphia	38%	11%	6%	5%	2%	1%	0%	0%	0%
Sacramento	15%	1%	0%	0%	0%	0%	0%	0%	0%
St. Louis	21%	10%	6%	4%	1%	1%	0%	0%	0%
Washington	29%	7%	4%	3%	1%	1%	0%	0%	0%

1-B)



1-C)


1.4E+5
1.2E+5
1.0E+5
8.0E+4
6.0E+4
4.0E+4
2.0E+4
0.0E+0
Number of children with exposures >0.08 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

1-D)

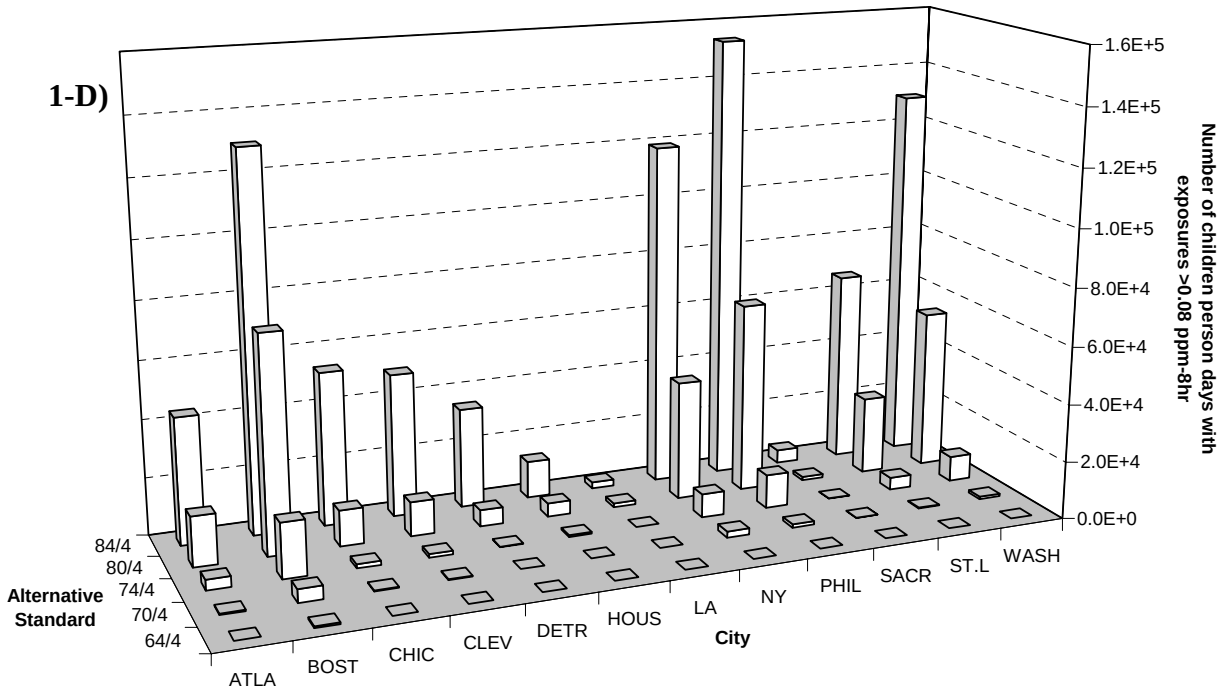
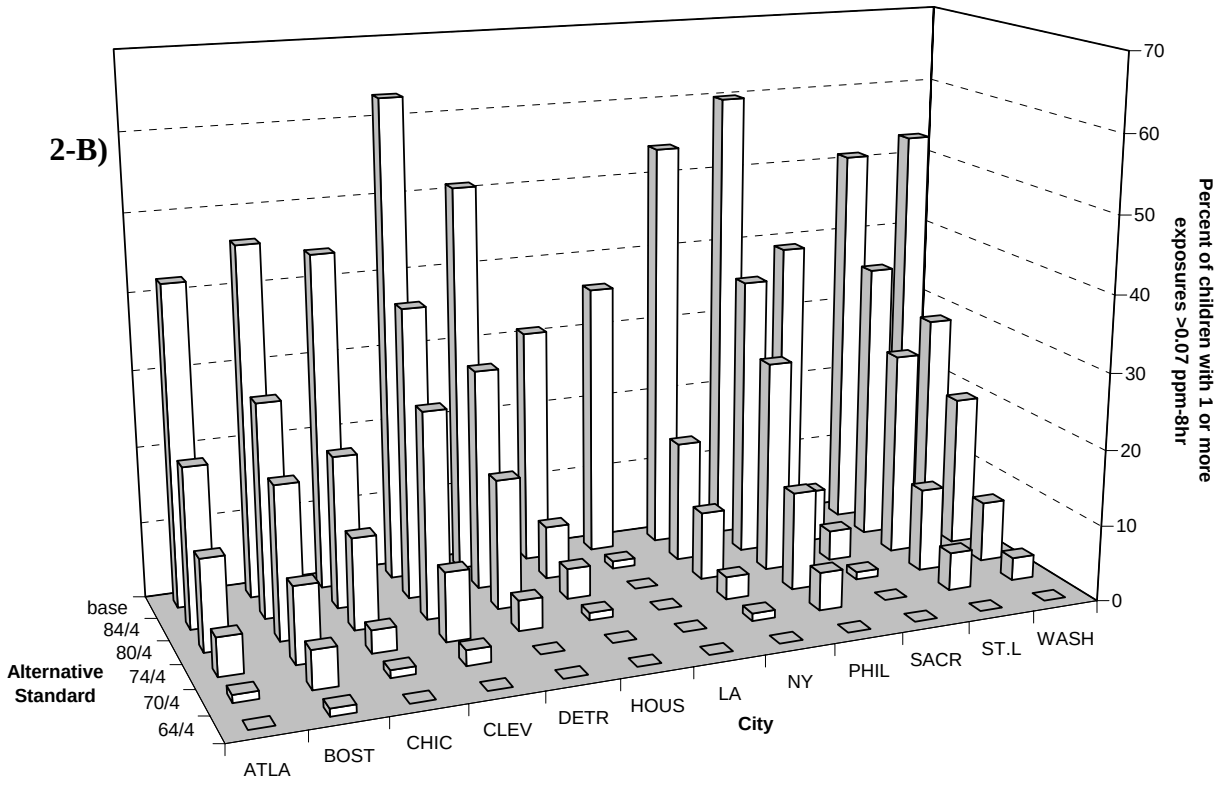

1.6E+5
1.4E+5
1.2E+5
1.0E+5
8.0E+4
6.0E+4
4.0E+4
2.0E+4
0.0E+0
Number of children person days with exposures >0.08 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 2. Summary of exposure metrics regarding estimated exceedances of 0.070 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2002 air quality. A) Table of percent of children with at least one exposure above 0.070 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.070 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.070 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.070 ppm 8-hr.

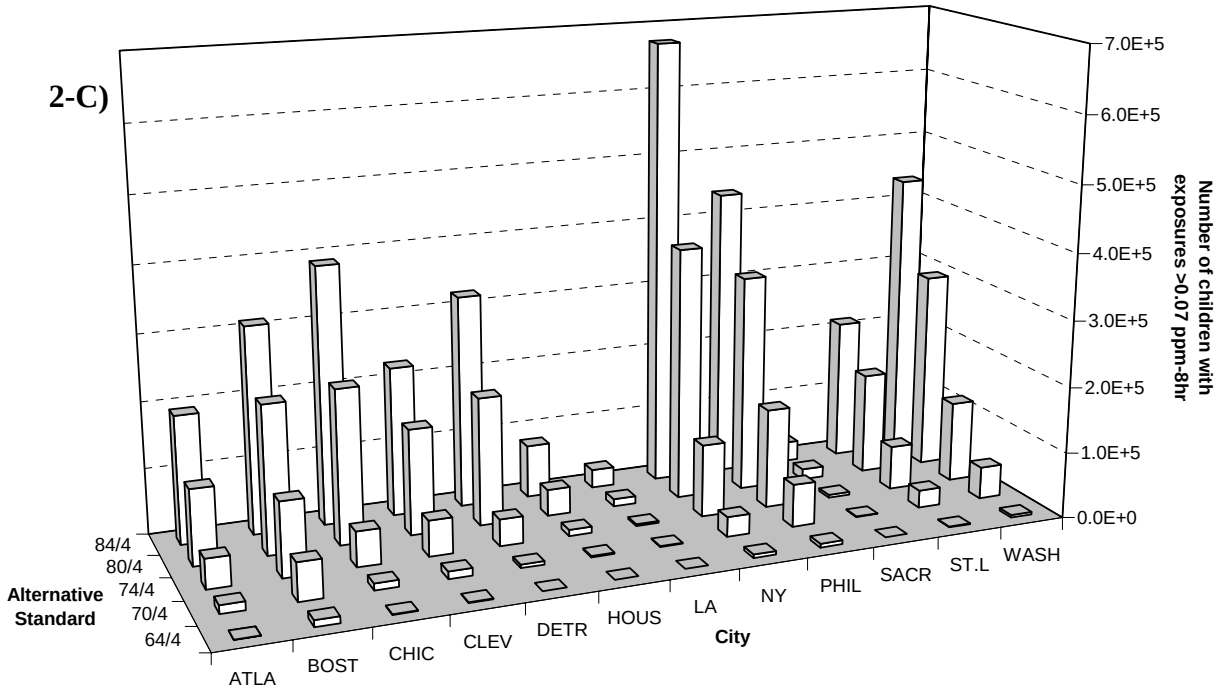
2-A) Percent of children with exposures > 0.070 ppm, moderate exertion, 2002 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	42%	21%	19%	12%	7%	5%	5%	1%	0%
Boston	46%	28%	21%	20%	18%	10%	7%	5%	1%
Chicago	44%	20%	15%	12%	6%	3%	2%	1%	0%
Cleveland	63%	38%	29%	27%	12%	9%	3%	2%	0%
Detroit	51%	29%	20%	17%	15%	4%	1%	0%	0%
Houston	31%	7%	4%	4%	1%	1%	0%	0%	0%
Los Angeles	36%	1%	1%	0%	0%	0%	0%	0%	0%
New York	54%	16%	13%	9%	2%	3%	2%	1%	0%
Philadelphia	60%	37%	30%	28%	16%	13%	8%	5%	0%
Sacramento	39%	7%	5%	4%	2%	1%	1%	0%	0%
St. Louis	51%	37%	31%	27%	15%	11%	7%	5%	0%
Washington	53%	29%	21%	20%	12%	8%	4%	3%	0%

2-B)



2-C)


7.0E+5
6.0E+5
5.0E+5
4.0E+5
3.0E+5
2.0E+5
1.0E+5
0.0E+0
Number of children with exposures >0.07 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

2-D)

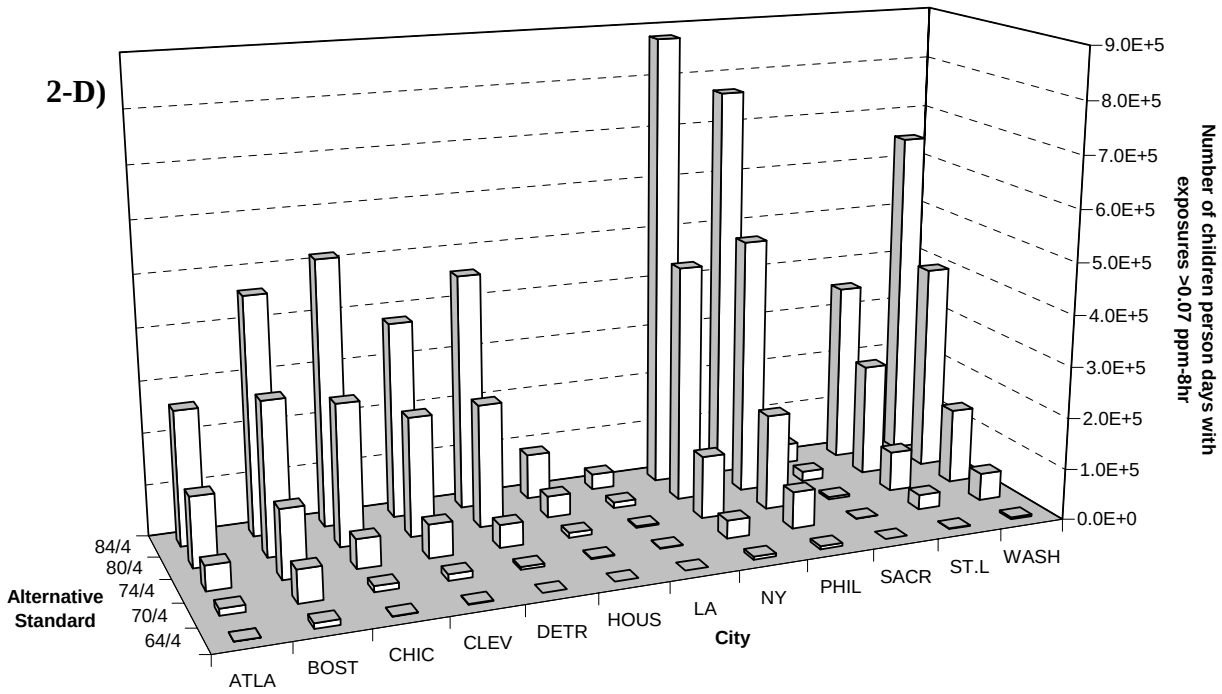
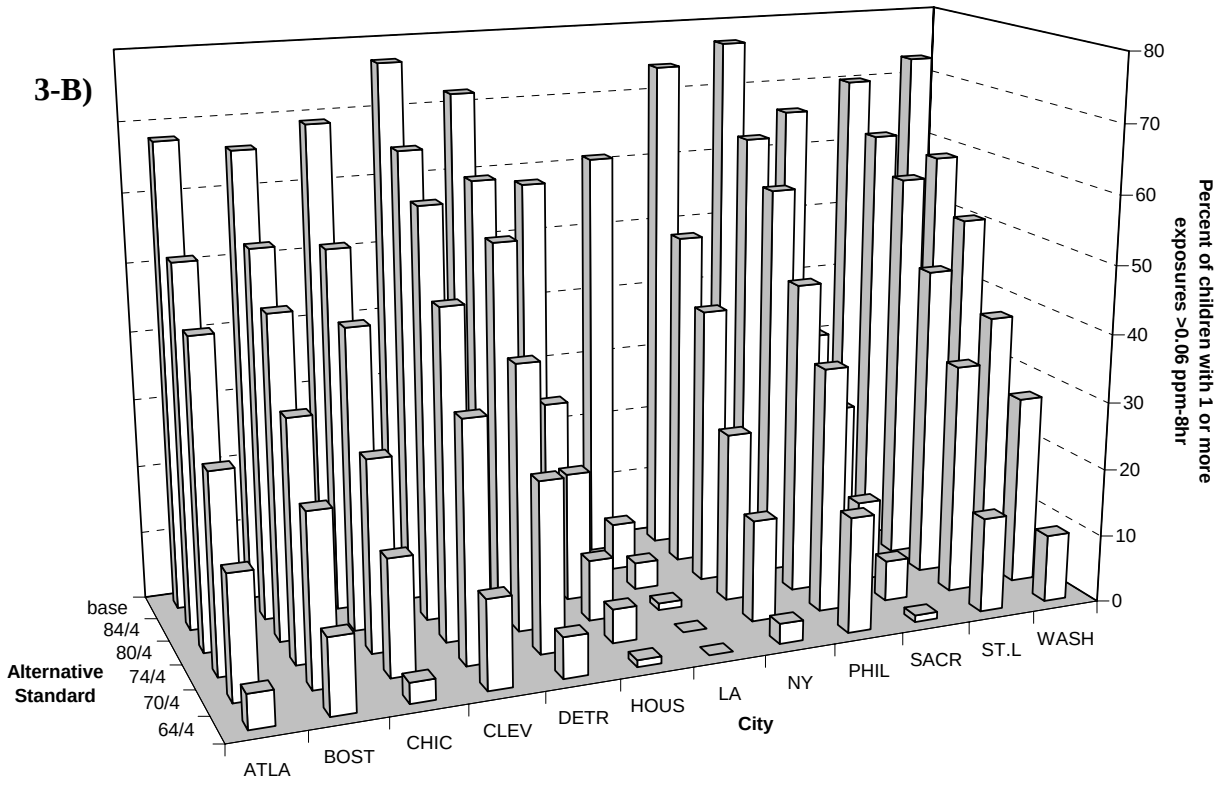

9.0E+5
8.0E+5
7.0E+5
6.0E+5
5.0E+5
4.0E+5
3.0E+5
2.0E+5
1.0E+5
0.0E+0
Number of children person days with exposures >0.07 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 3. Summary of exposure metrics regarding estimated exceedances of 0.060 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2002 air quality. A) Table of percent of children with at least one exposure above 0.060 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.060 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.060 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.060 ppm 8-hr.

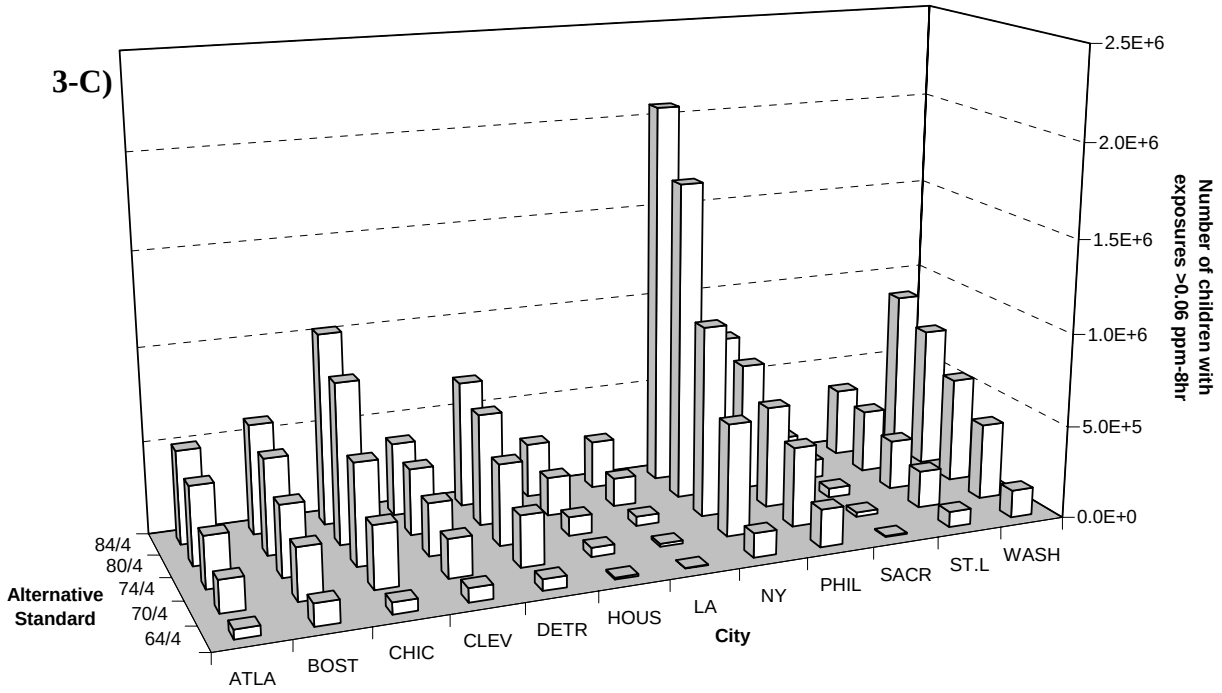
3-A) Percent of children with exposures > 0.060 ppm, moderate exertion, 2002 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	68%	53%	53%	45%	35%	29%	29%	18%	5%
Boston	66%	54%	48%	47%	44%	35%	29%	25%	11%
Chicago	69%	53%	48%	44%	37%	28%	23%	17%	3%
Cleveland	77%	66%	62%	60%	51%	48%	40%	35%	13%
Detroit	72%	61%	55%	54%	52%	39%	30%	25%	6%
Houston	58%	27%	21%	19%	11%	9%	7%	5%	1%
Los Angeles	61%	7%	6%	4%	1%	1%	1%	0%	0%
New York	74%	50%	46%	41%	23%	25%	21%	15%	3%
Philadelphia	77%	64%	60%	58%	49%	46%	41%	36%	17%
Sacramento	66%	33%	29%	24%	15%	12%	10%	6%	1%
St. Louis	70%	63%	60%	58%	50%	46%	40%	34%	14%
Washington	73%	59%	52%	51%	43%	38%	30%	28%	10%

3-B)



3-C)


2.5E+6
2.0E+6
1.5E+6
1.0E+6
5.0E+5
0.0E+0
Number of children with exposures >0.06 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

3-D)

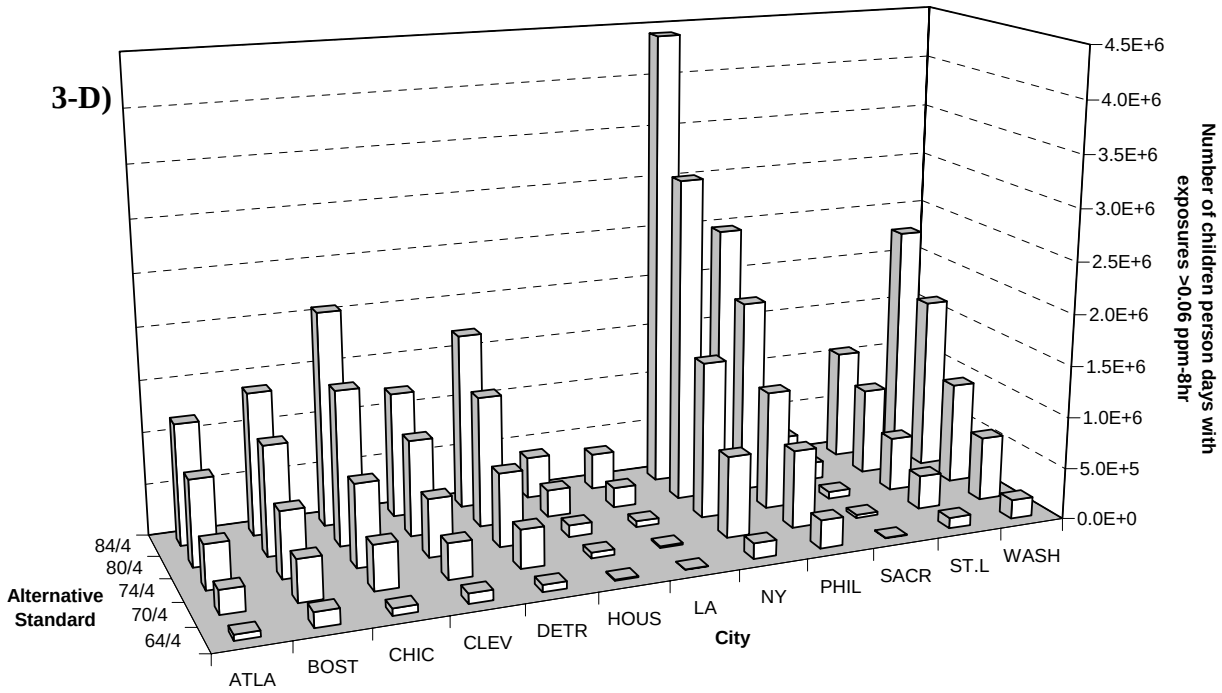
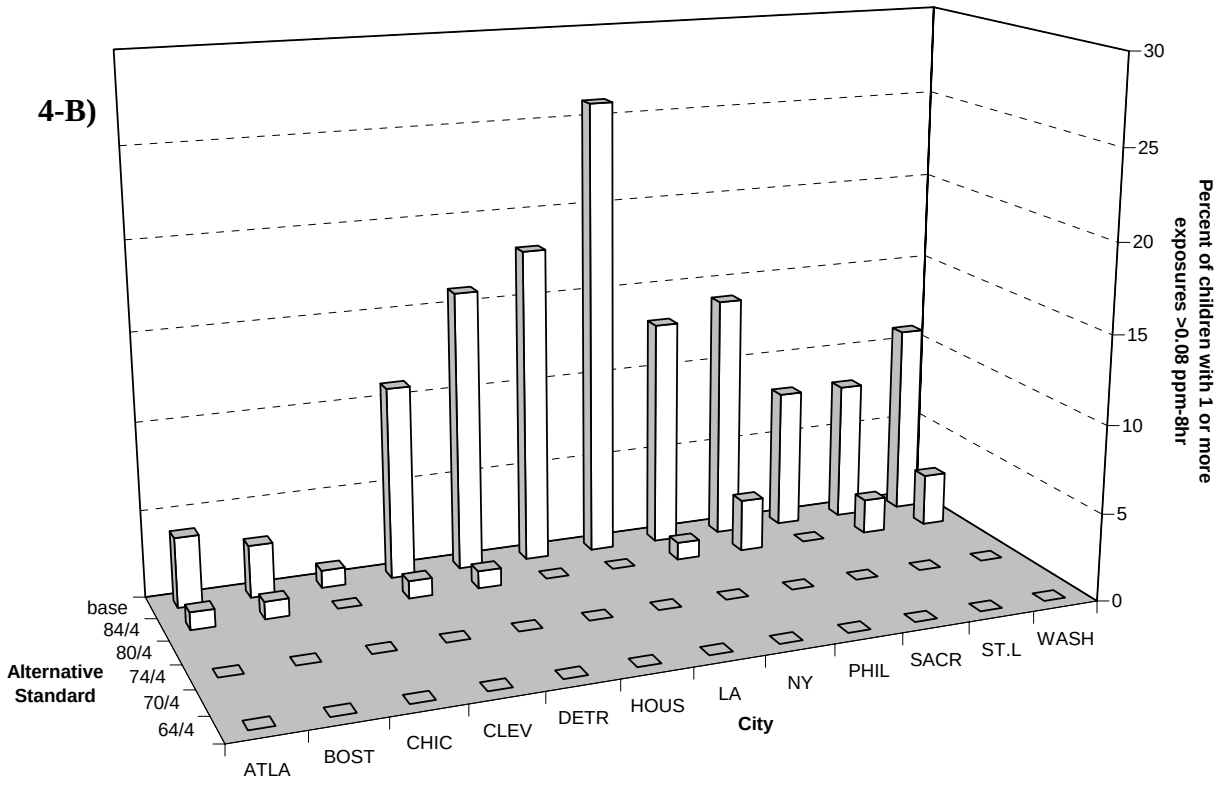

4.5E+6
4.0E+6
3.5E+6
3.0E+6
2.5E+6
2.0E+6
1.5E+6
1.0E+6
5.0E+5
0.0E+0
Number of children person days with exposures >0.06 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 4. Summary of exposure metrics regarding estimated exceedances of 0.080 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2003 air quality. A) Table of percent of children with at least one exposure above 0.080 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.080 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.080 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.080 ppm 8-hr.

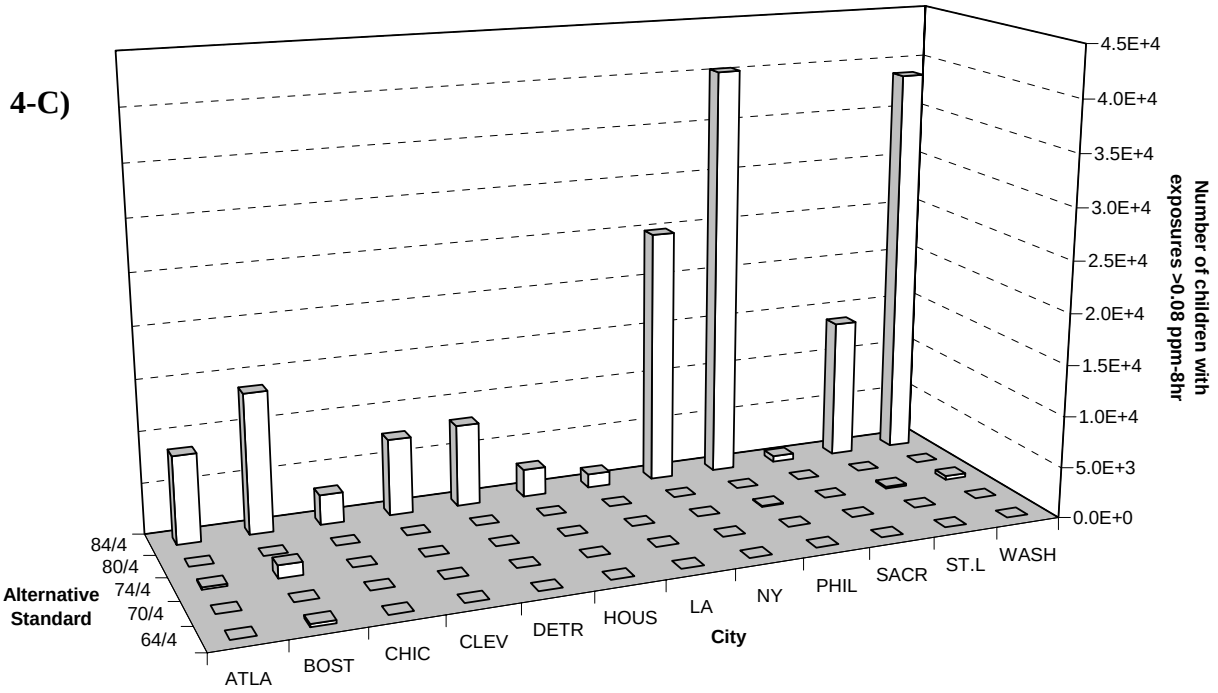
4-A) Percent of children with exposures > 0.080 ppm, moderate exertion, 2003 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	4%	1%	.	.	.	0%	.	.	0%
Boston	3%	1%	.	.	.	0%	.	.	0%
Chicago	1%	0%	.	.	.	0%	.	.	0%
Cleveland	11%	1%	.	.	.	0%	.	.	0%
Detroit	16%	1%	.	.	.	0%	.	.	0%
Houston	18%	0%	.	.	.	0%	.	.	0%
Los Angeles	26%	0%	.	.	.	0%	.	.	0%
New York	13%	1%	.	.	.	0%	.	.	0%
Philadelphia	14%	3%	.	.	.	0%	.	.	0%
Sacramento	8%	0%	.	.	.	0%	.	.	0%
St. Louis	8%	2%	.	.	.	0%	.	.	0%
Washington	11%	3%	.	.	.	0%	.	.	0%

4-B)



4-C)


4.5E+4
4.0E+4
3.5E+4
3.0E+4
2.5E+4
2.0E+4
1.5E+4
1.0E+4
5.0E+3
0.0E+0
Number of children with exposures >0.08 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

4-D)

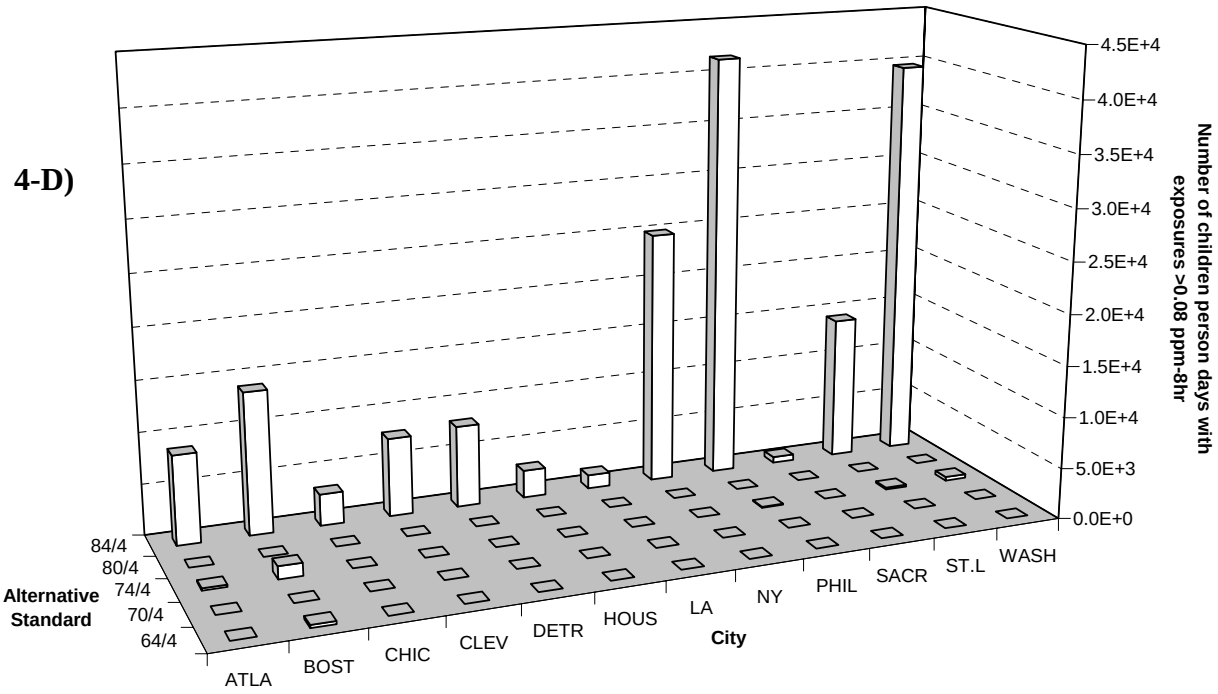
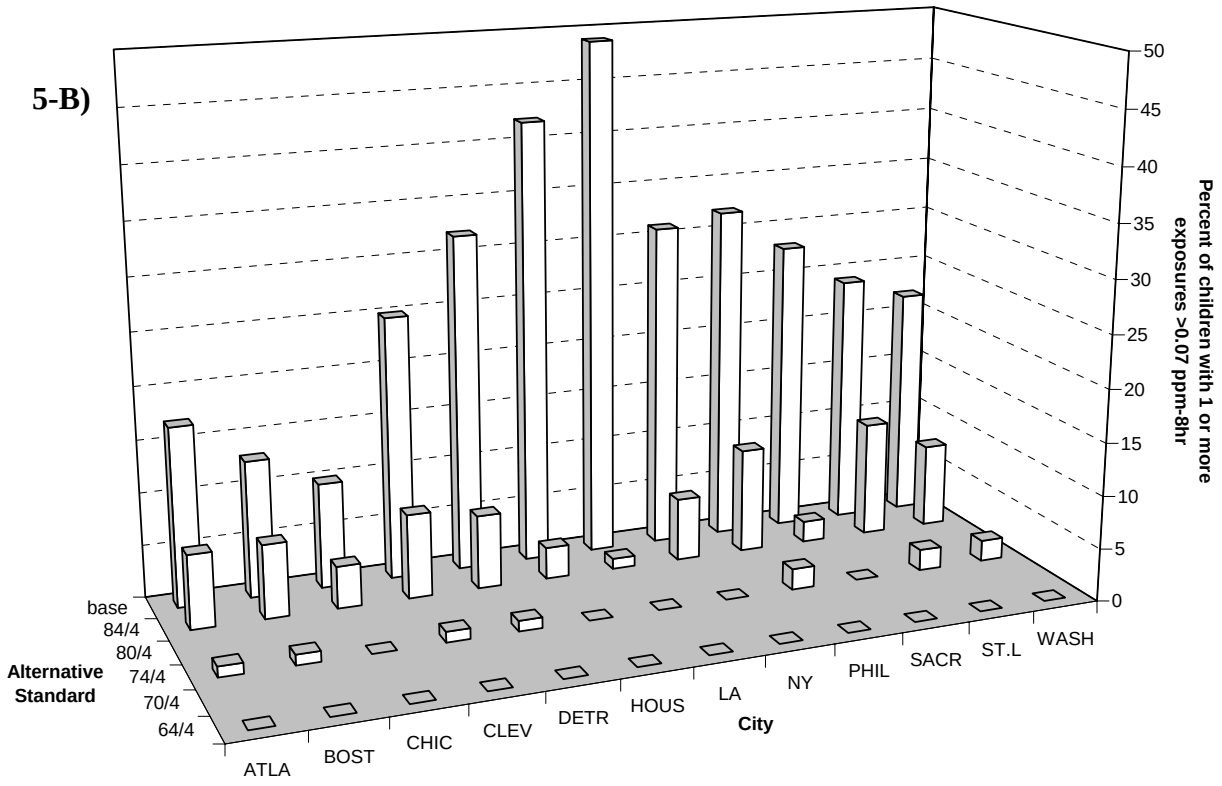

4.5E+4
4.0E+4
3.5E+4
3.0E+4
2.5E+4
2.0E+4
1.5E+4
1.0E+4
5.0E+3
0.0E+0
Number of children person days with exposures >0.08 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 5. Summary of exposure metrics regarding estimated exceedances of 0.070 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2003 air quality. A) Table of percent of children with at least one exposure above 0.070 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.070 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.070 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.070 ppm 8-hr.

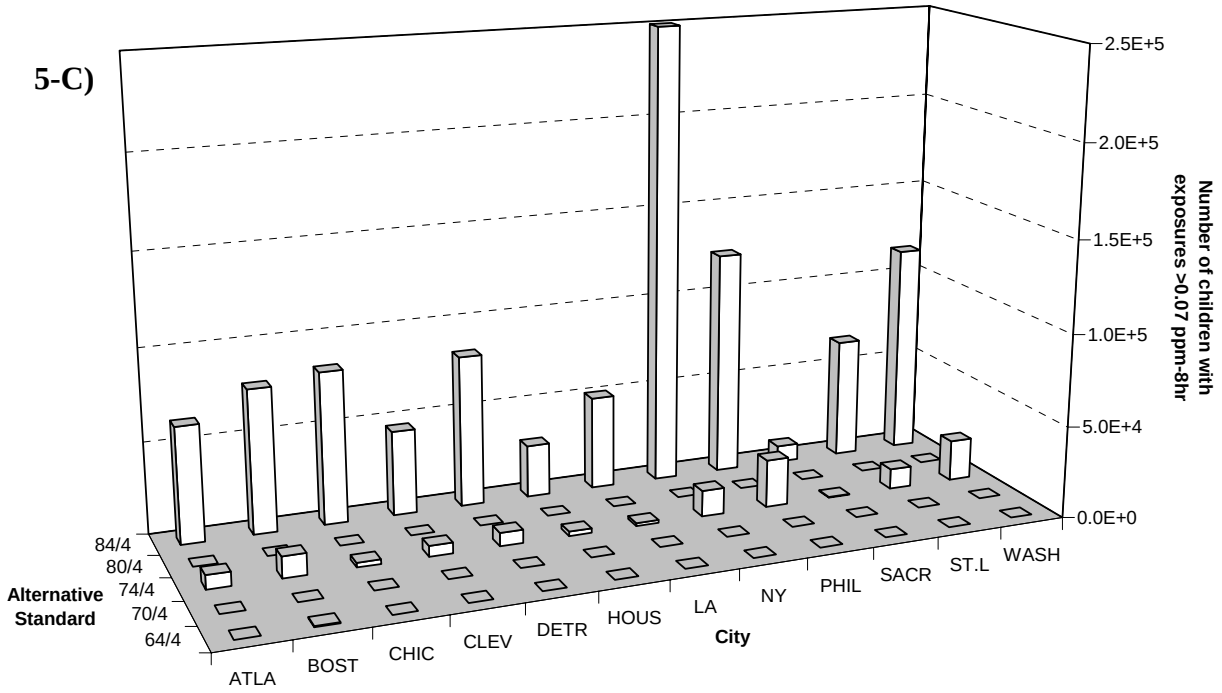
5-A) Percent of children with exposures > 0.070 ppm, moderate exertion, 2003 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	17%	7%	.	.	.	1%	.	.	0%
Boston	13%	7%	.	.	.	1%	.	.	0%
Chicago	10%	4%	.	.	.	0%	.	.	0%
Cleveland	25%	8%	.	.	.	1%	.	.	0%
Detroit	32%	7%	.	.	.	1%	.	.	0%
Houston	42%	3%	.	.	.	0%	.	.	0%
Los Angeles	49%	1%	.	.	.	0%	.	.	0%
New York	31%	6%	.	.	.	0%	.	.	0%
Philadelphia	32%	10%	.	.	.	2%	.	.	0%
Sacramento	28%	2%	.	.	.	0%	.	.	0%
St. Louis	24%	11%	.	.	.	2%	.	.	0%
Washington	22%	8%	.	.	.	2%	.	.	0%

5-B)



5-C)


2.5E+5
2.0E+5
1.5E+5
1.0E+5
5.0E+4
0.0E+0
Number of children with exposures >0.07 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

5-D)

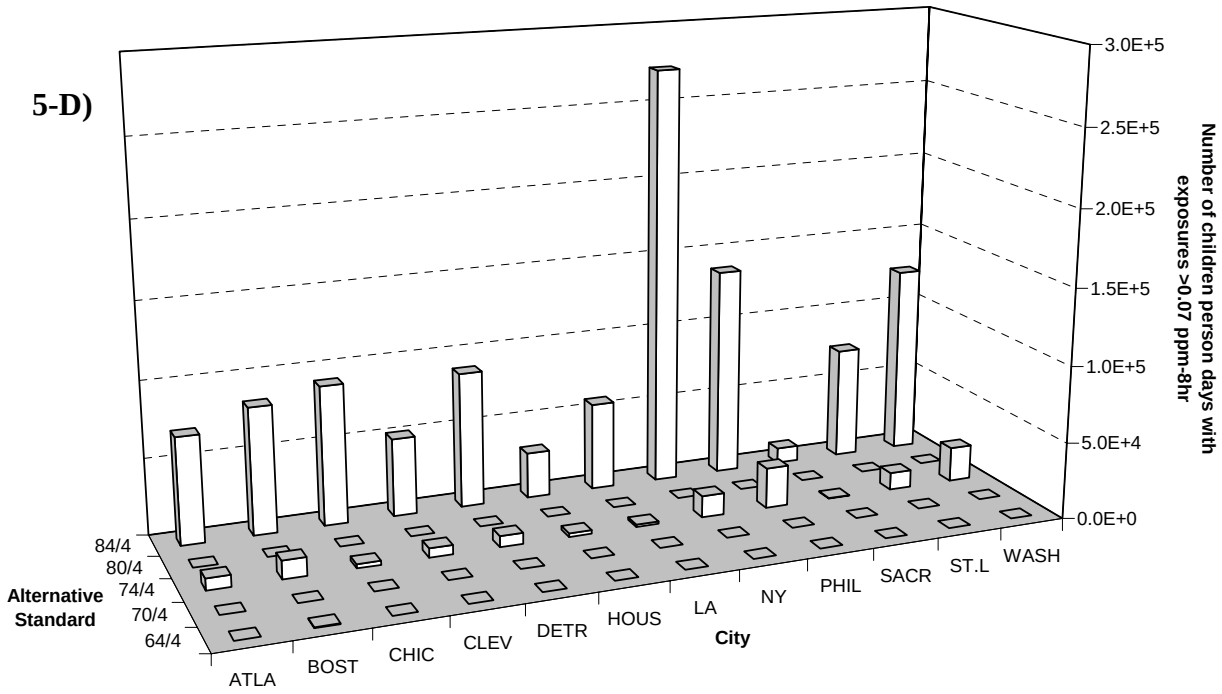
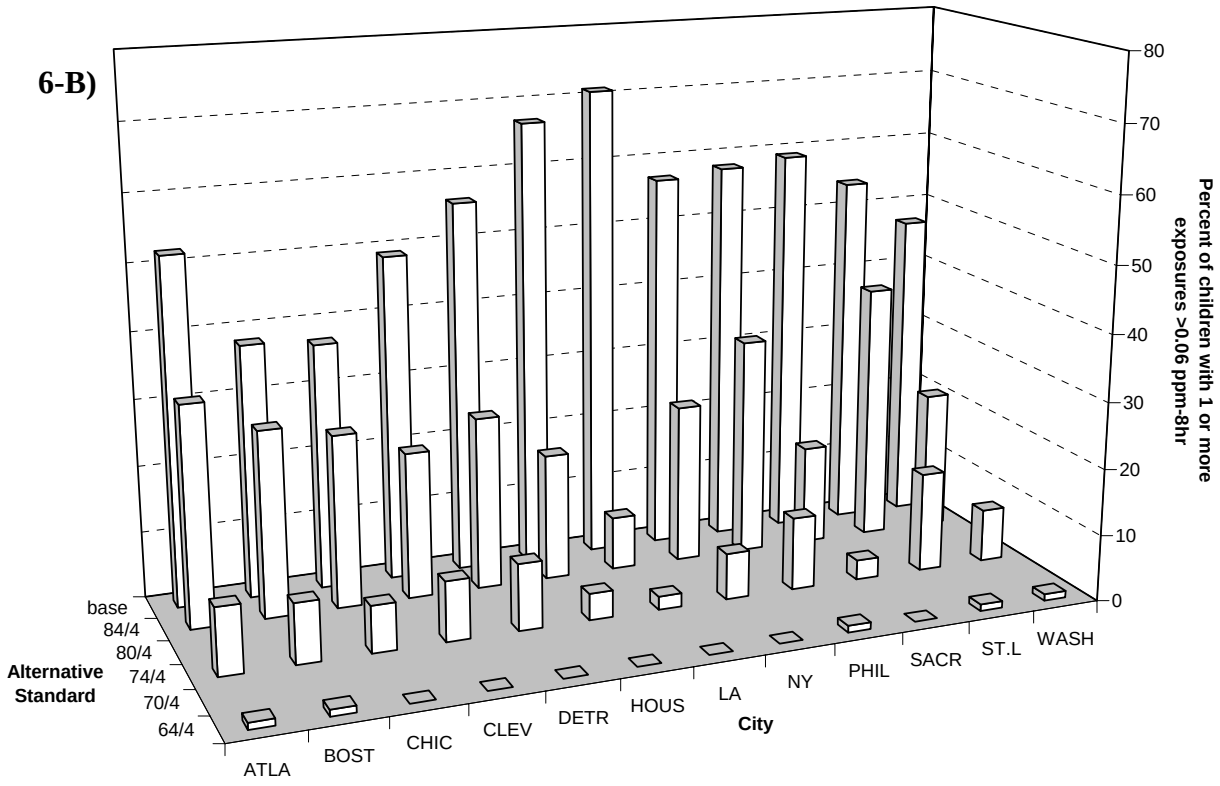

3.0E+5
2.5E+5
2.0E+5
1.5E+5
1.0E+5
5.0E+4
0.0E+0
Number of children person days with exposures >0.07 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 6. Summary of exposure metrics regarding estimated exceedances of 0.060 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2003 air quality. A) Table of percent of children with at least one exposure above 0.060 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.060 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.060 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.060 ppm 8-hr.

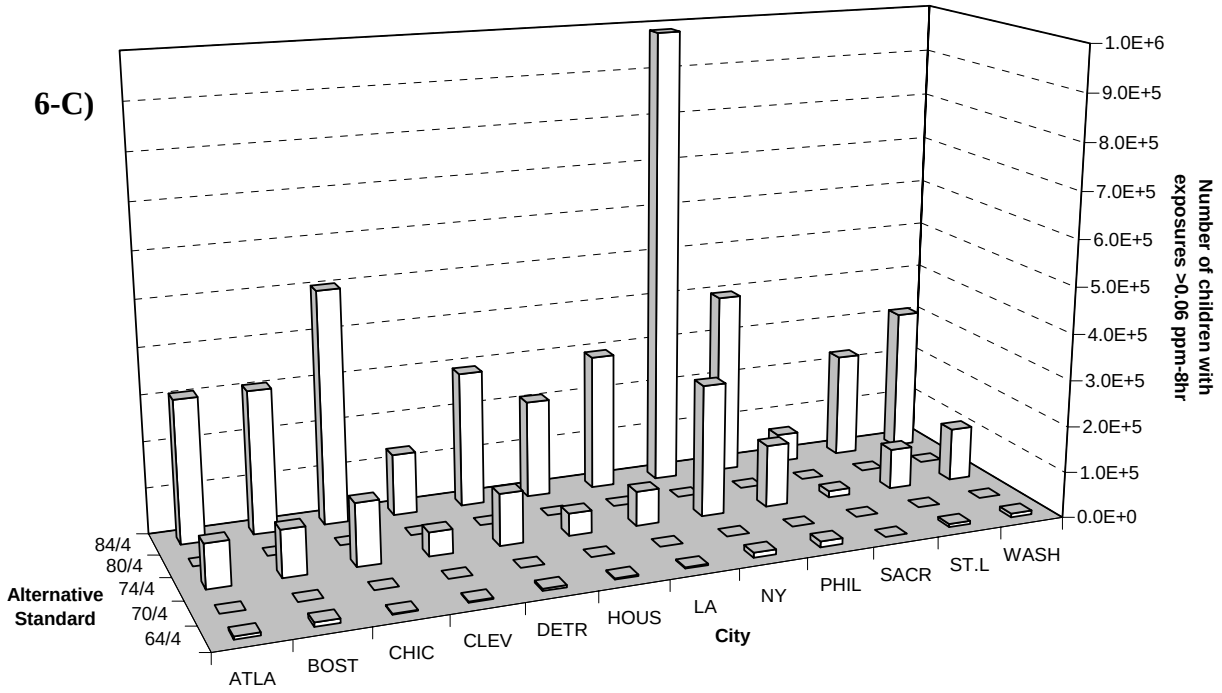
6-A) Percent of children with exposures > 0.060 ppm, moderate exertion, 2003 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	52%	33%	.	.	.	10%	.	.	1%
Boston	38%	28%	.	.	.	9%	.	.	1%
Chicago	37%	26%	.	.	.	7%	.	.	0%
Cleveland	49%	22%	.	.	.	9%	.	.	0%
Detroit	56%	26%	.	.	.	10%	.	.	0%
Houston	67%	19%	.	.	.	4%	.	.	0%
Los Angeles	71%	8%	.	.	.	2%	.	.	0%
New York	57%	24%	.	.	.	7%	.	.	0%
Philadelphia	58%	33%	.	.	.	11%	.	.	1%
Sacramento	59%	15%	.	.	.	3%	.	.	0%
St. Louis	54%	39%	.	.	.	15%	.	.	1%
Washington	47%	21%	.	.	.	8%	.	.	1%

6-B)



6-C)


1.0E+6
9.0E+5
8.0E+5
7.0E+5
6.0E+5
5.0E+5
4.0E+5
3.0E+5
2.0E+5
1.0E+5
0.0E+0
Number of children with exposures >0.06 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

6-D)

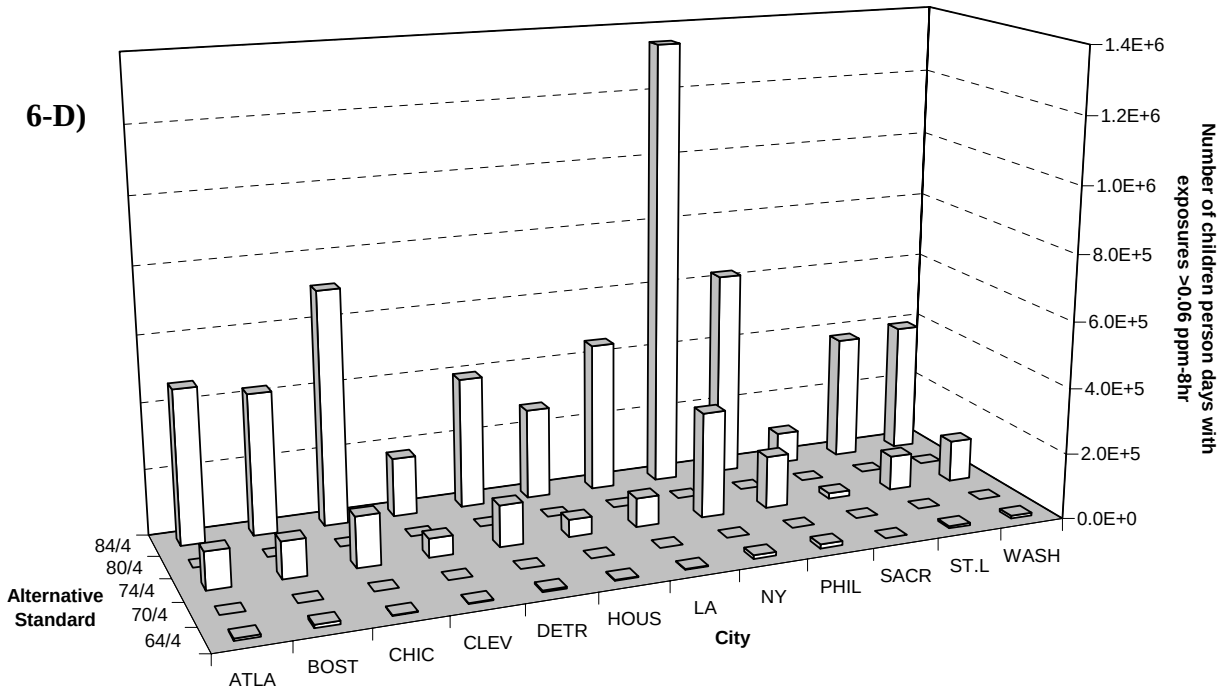
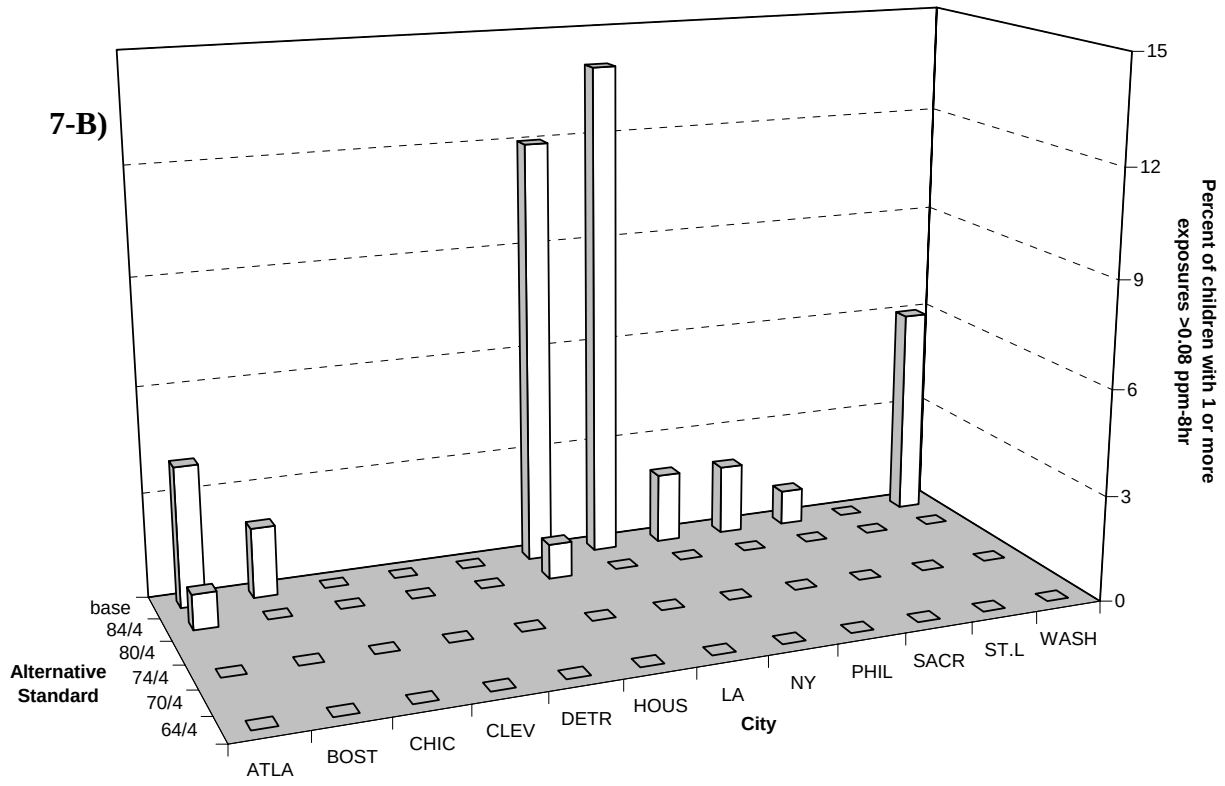

1.4E+6
1.2E+6
1.0E+6
8.0E+5
6.0E+5
4.0E+5
2.0E+5
0.0E+0
Number of children person days with exposures >0.06 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 7. Summary of exposure metrics regarding estimated exceedances of 0.080 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2004 air quality. A) Table of percent of children with at least one exposure above 0.080 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.080 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.080 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.080 ppm 8-hr.

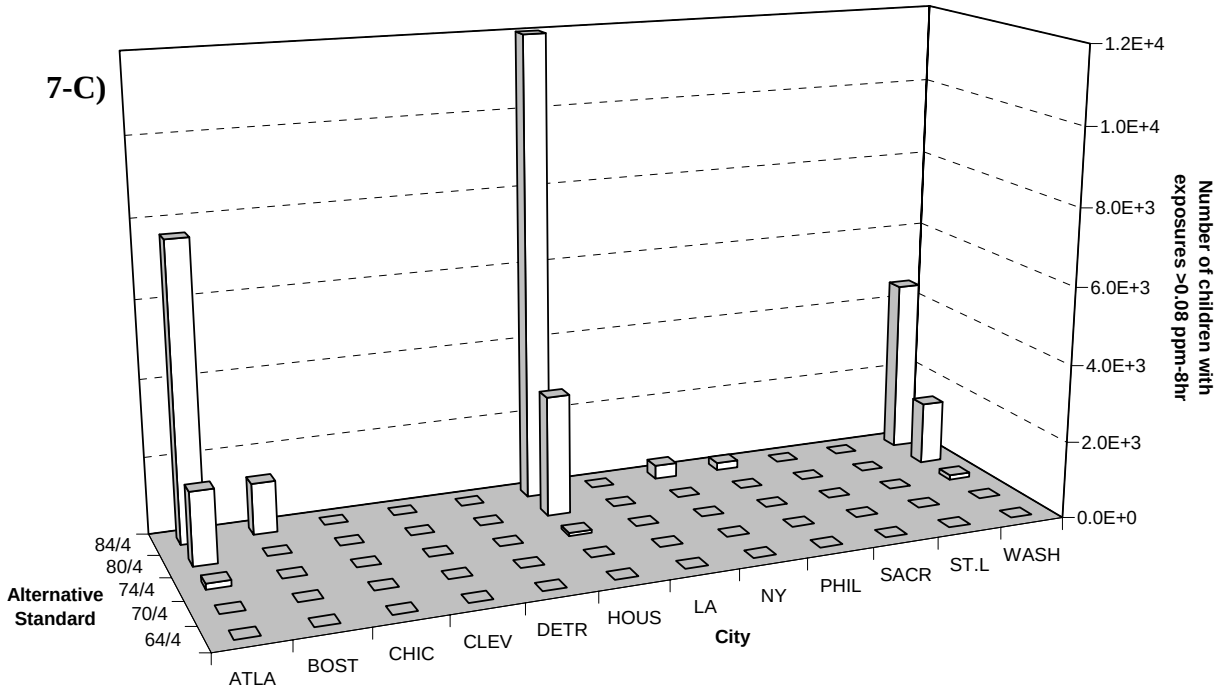
7-A) Percent of children with exposures > 0.080 ppm, moderate exertion, 2004 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	4%	1%	1%	0%	0%	0%	0%	0%	0%
Boston	2%	0%	0%	0%	0%	0%	0%	0%	0%
Chicago	0%	0%	0%	0%	0%	0%	0%	0%	0%
Cleveland	0%	0%	0%	0%	0%	0%	0%	0%	0%
Detroit	0%	0%	0%	0%	0%	0%	0%	0%	0%
Houston	12%	1%	0%	0%	0%	0%	0%	0%	0%
Los Angeles	14%	0%	0%	0%	0%	0%	0%	0%	0%
New York	2%	0%	0%	0%	0%	0%	0%	0%	0%
Philadelphia	2%	0%	0%	0%	0%	0%	0%	0%	0%
Sacramento	1%	0%	0%	0%	0%	0%	0%	0%	0%
St. Louis	0%	0%	0%	0%	0%	0%	0%	0%	0%
Washington	6%	0%	0%	0%	0%	0%	0%	0%	0%

7-B)



7-C)


1.2E+4
1.0E+4
8.0E+3
6.0E+3
4.0E+3
2.0E+3
0.0E+0
Number of children with exposures >0.08 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

7-D)

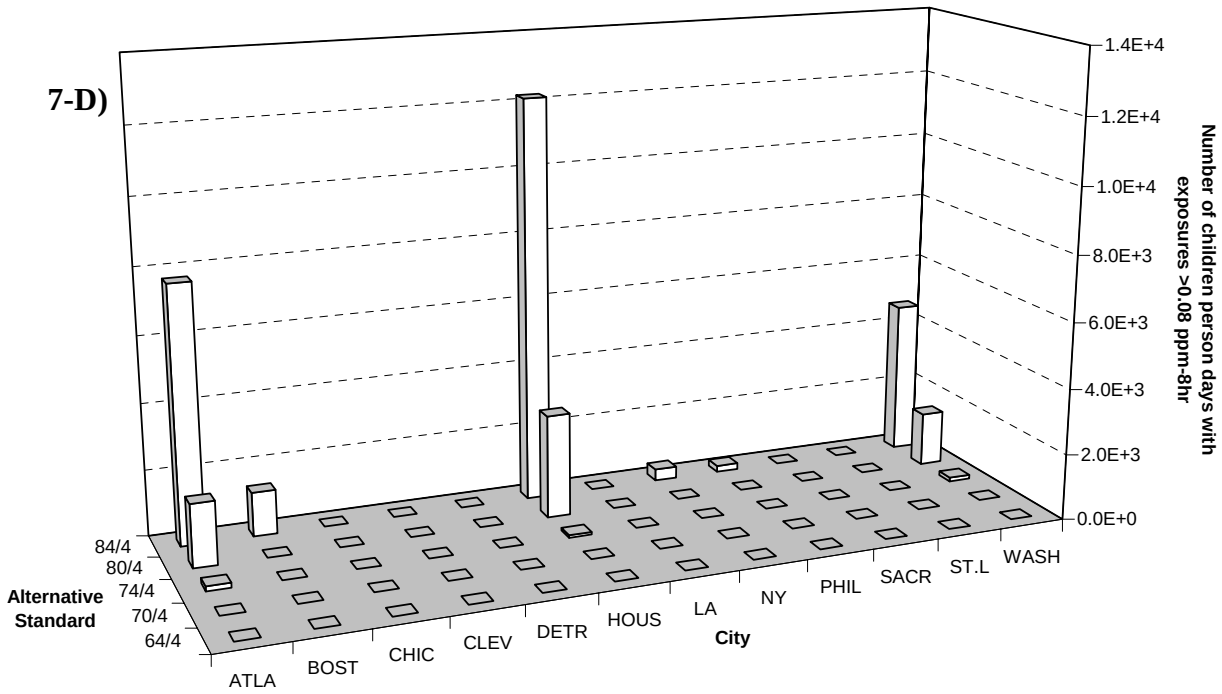
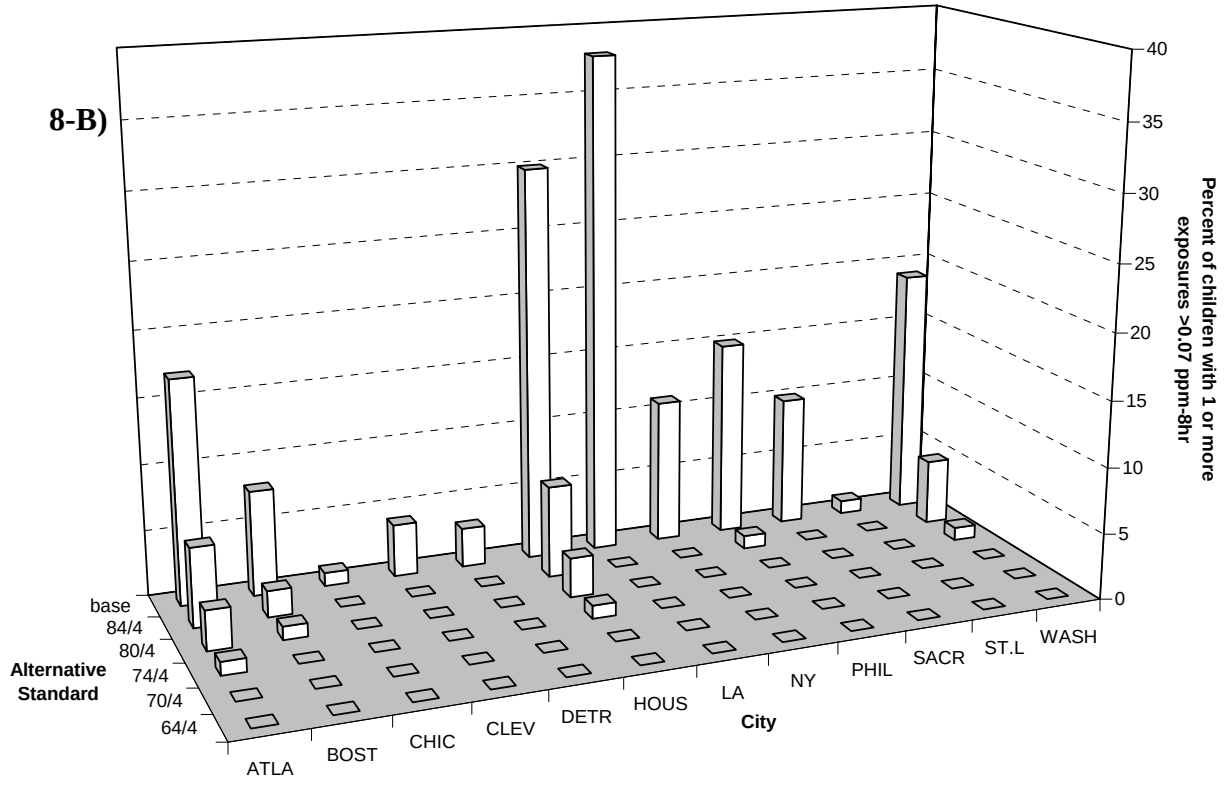

1.4E+4
1.2E+4
1.0E+4
8.0E+3
6.0E+3
4.0E+3
2.0E+3
0.0E+0
Number of children person days with exposures >0.08 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 8. Summary of exposure metrics regarding estimated exceedances of 0.070 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2004 air quality. A) Table of percent of children with at least one exposure above 0.070 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.070 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.070 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.070 ppm 8-hr.

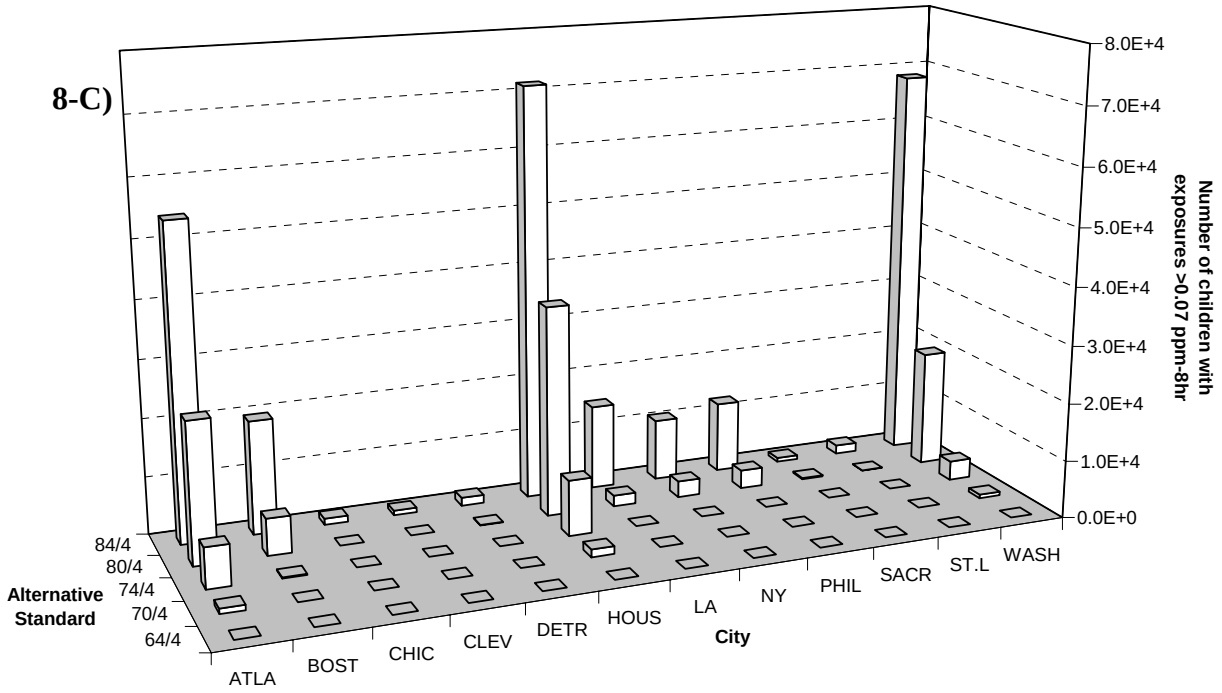
8-A) Percent of children with exposures > 0.070 ppm, moderate exertion, 2004 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	17%	6%	5%	3%	1%	1%	1%	0%	0%
Boston	8%	2%	1%	1%	0%	0%	0%	0%	0%
Chicago	1%	0%	0%	0%	0%	0%	0%	0%	0%
Cleveland	4%	0%	0%	0%	0%	0%	0%	0%	0%
Detroit	3%	0%	0%	0%	0%	0%	0%	0%	0%
Houston	30%	7%	4%	3%	1%	1%	0%	0%	0%
Los Angeles	38%	0%	0%	0%	0%	0%	0%	0%	0%
New York	11%	0%	0%	0%	0%	0%	0%	0%	0%
Philadelphia	15%	1%	0%	0%	0%	0%	0%	0%	0%
Sacramento	10%	0%	0%	0%	0%	0%	0%	0%	0%
St. Louis	1%	0%	0%	0%	0%	0%	0%	0%	0%
Washington	19%	5%	1%	1%	0%	0%	0%	0%	0%

8-B)



8-C)


8.0E+4
7.0E+4
6.0E+4
5.0E+4
4.0E+4
3.0E+4
2.0E+4
1.0E+4
0.0E+0
Number of children with exposures >0.07 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

8-D)

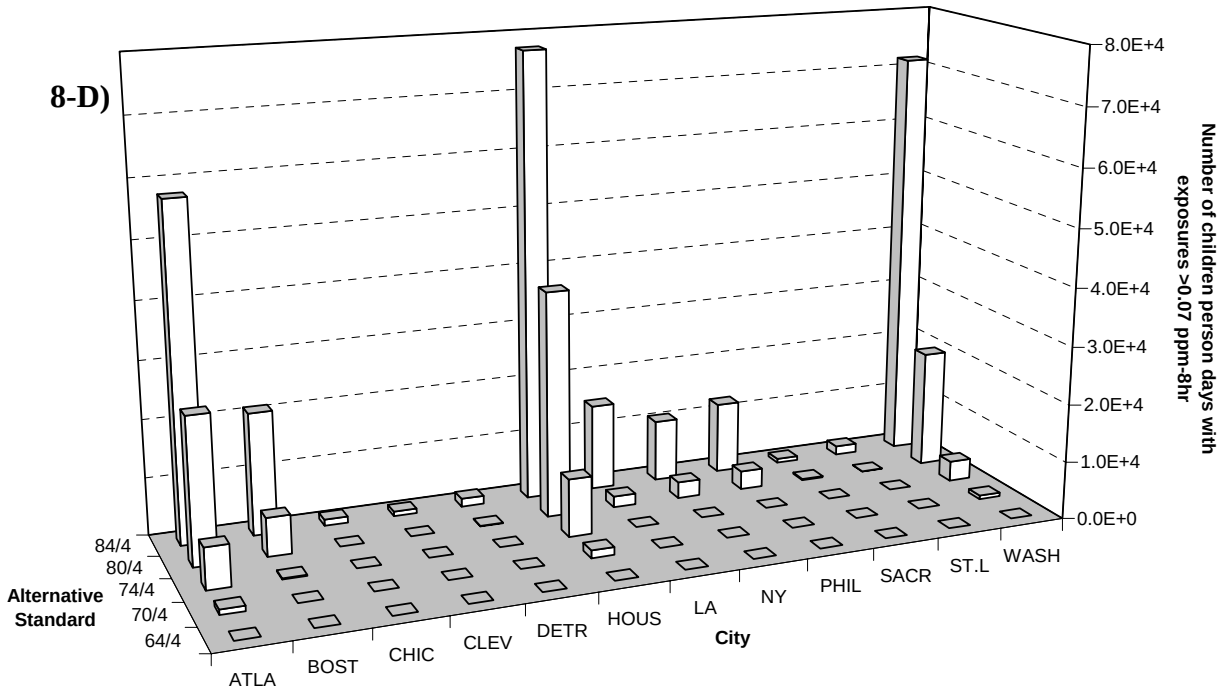
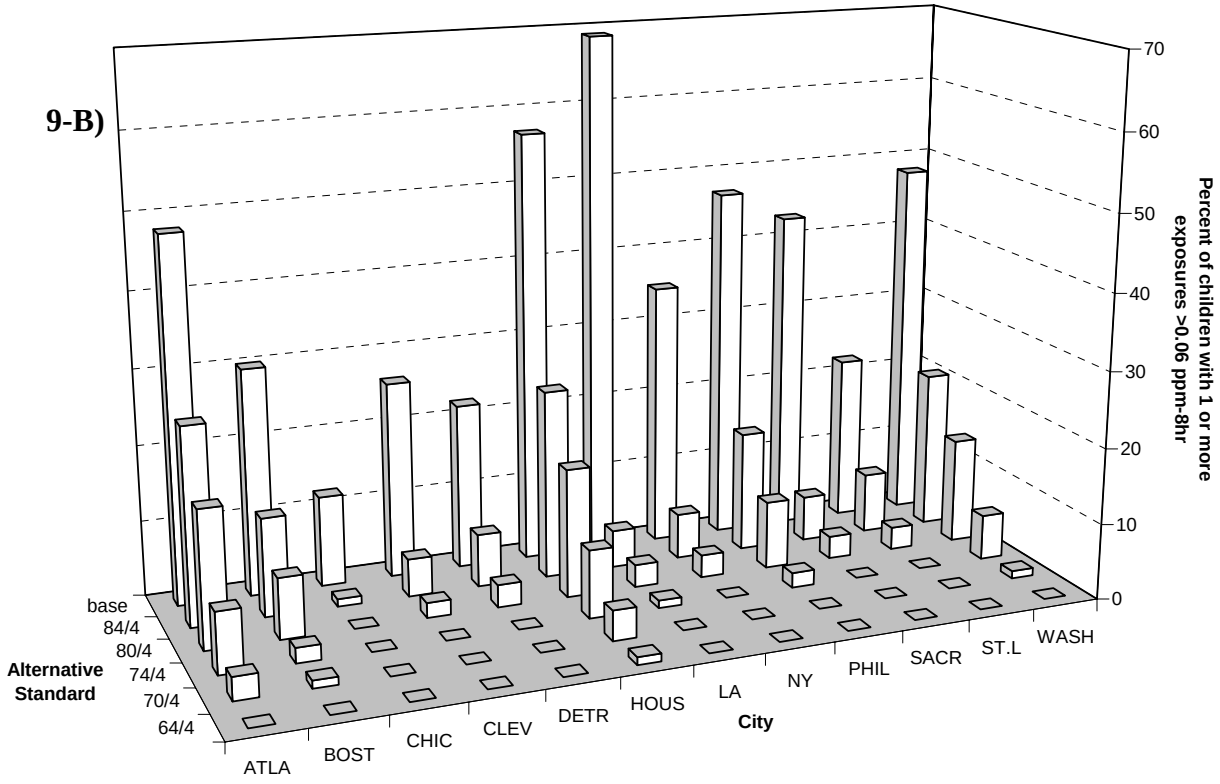

8.0E+4
7.0E+4
6.0E+4
5.0E+4
4.0E+4
3.0E+4
2.0E+4
1.0E+4
0.0E+0
Number of children person days with exposures >0.07 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

Exhibit 9. Summary of exposure metrics regarding estimated exceedances of 0.060 ppm 8-hr ozone exposures, concomitant with moderate or greater exertion, based on 2004 air quality. A) Table of percent of children with at least one exposure above 0.060 ppm 8-hr, B) Illustration of percent of children with at least one exposure above 0.060 ppm 8-hr, C) Illustration of the number of children with at least one exposure above 0.060 ppm 8-hr, D) Illustration of the number of children person days with exposure above 0.060 ppm 8-hr.

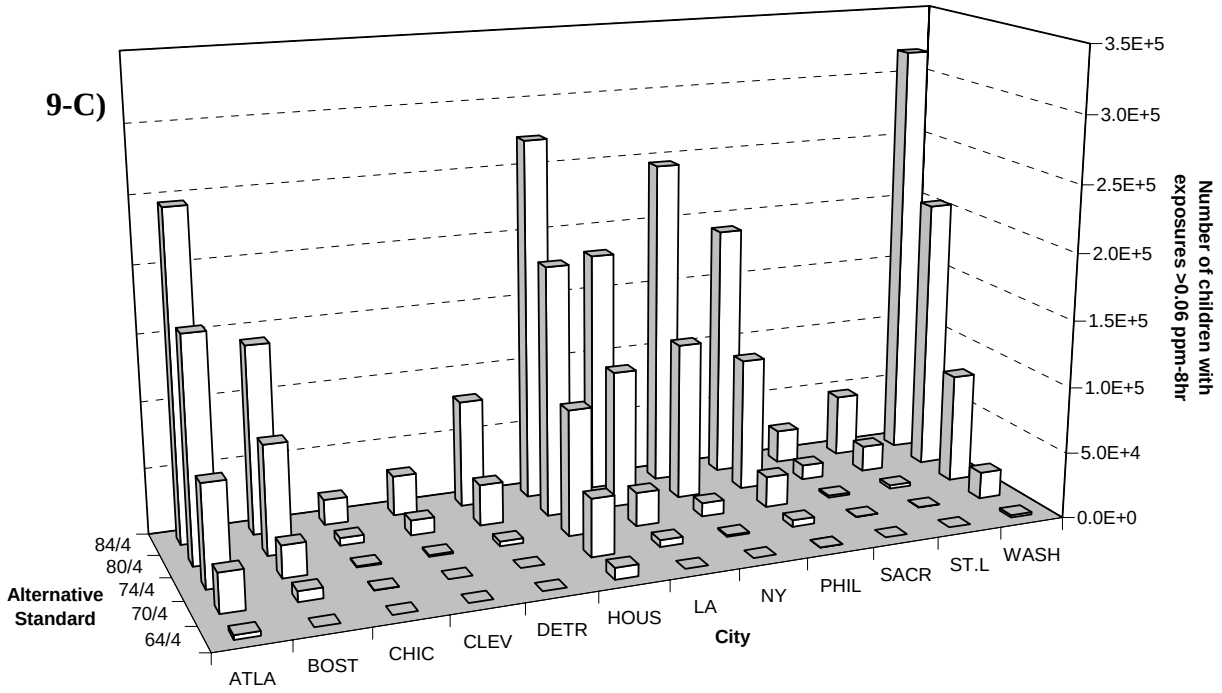
9-A) Percent of children with exposures > 0.060 ppm, moderate exertion, 2004 data

City	base	84/4	84/3	80/4	74/5	74/4	74/3	70/4	64/4
Atlanta	48%	26%	25%	18%	11%	8%	8%	3%	0%
Boston	30%	13%	8%	8%	6%	2%	1%	1%	0%
Chicago	12%	1%	0%	0%	0%	0%	0%	0%	0%
Cleveland	26%	5%	3%	2%	0%	0%	0%	0%	0%
Detroit	22%	7%	4%	3%	2%	0%	0%	0%	0%
Houston	57%	25%	20%	17%	10%	9%	6%	4%	1%
Los Angeles	69%	5%	4%	3%	1%	1%	1%	0%	0%
New York	35%	6%	4%	3%	0%	0%	0%	0%	0%
Philadelphia	47%	16%	11%	9%	3%	2%	1%	0%	0%
Sacramento	43%	6%	4%	3%	1%	0%	0%	0%	0%
St. Louis	22%	8%	5%	3%	1%	0%	0%	0%	0%
Washington	48%	21%	15%	14%	7%	6%	2%	1%	0%

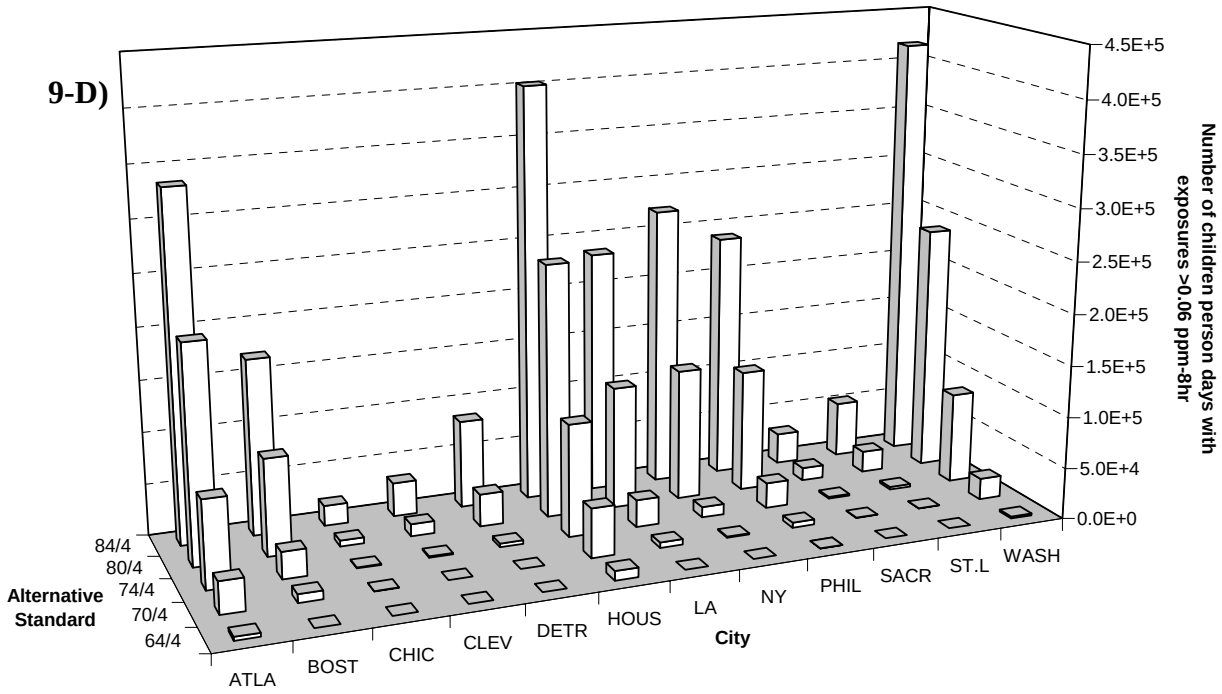
9-B)



9-C)


3.5E+5
3.0E+5
2.5E+5
2.0E+5
1.5E+5
1.0E+5
5.0E+4
0.0E+0
Number of children with exposures >0.06 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

9-D)


4.5E+5
4.0E+5
3.5E+5
3.0E+5
2.5E+5
2.0E+5
1.5E+5
1.0E+5
5.0E+4
0.0E+0
Number of children person days with exposures >0.06 ppm-8hr
84/4
80/4
74/4
70/4
64/4
Alternative Standard
ATLA
BOST
CHIC
CLEV
DETR
HOUS
LA
NY
PHIL
SACR
ST.L
WASH
City

4.7.3 Estimates of Repeated Exposures

As discussed in section 3.6.3, multiple exposures pose a greater health concern than single exposures. However, multiple repeated exposures are underestimated by APEX (discussed previously in the model evaluation section), which should be kept in mind for interpretation of these results. Figures 4-9 through 4-11 illustrate the effect of the current and alternative standards has on the estimated percent of children experiencing 1, 2, and 3 or more repeated exposures above 0.080, 0.070, and 0.060 ppm-8hr, respectively, concomitant with moderate or greater exertion, for each of the urban areas modeled, based on rollback of 2002 O_3 concentrations. When considering the alternative standard scenarios, clear trends are evident. The reduction in both the number of those experiencing exposures above a given benchmark level and the frequency of occurrence of those exceedances is directly correlated with the standard level.

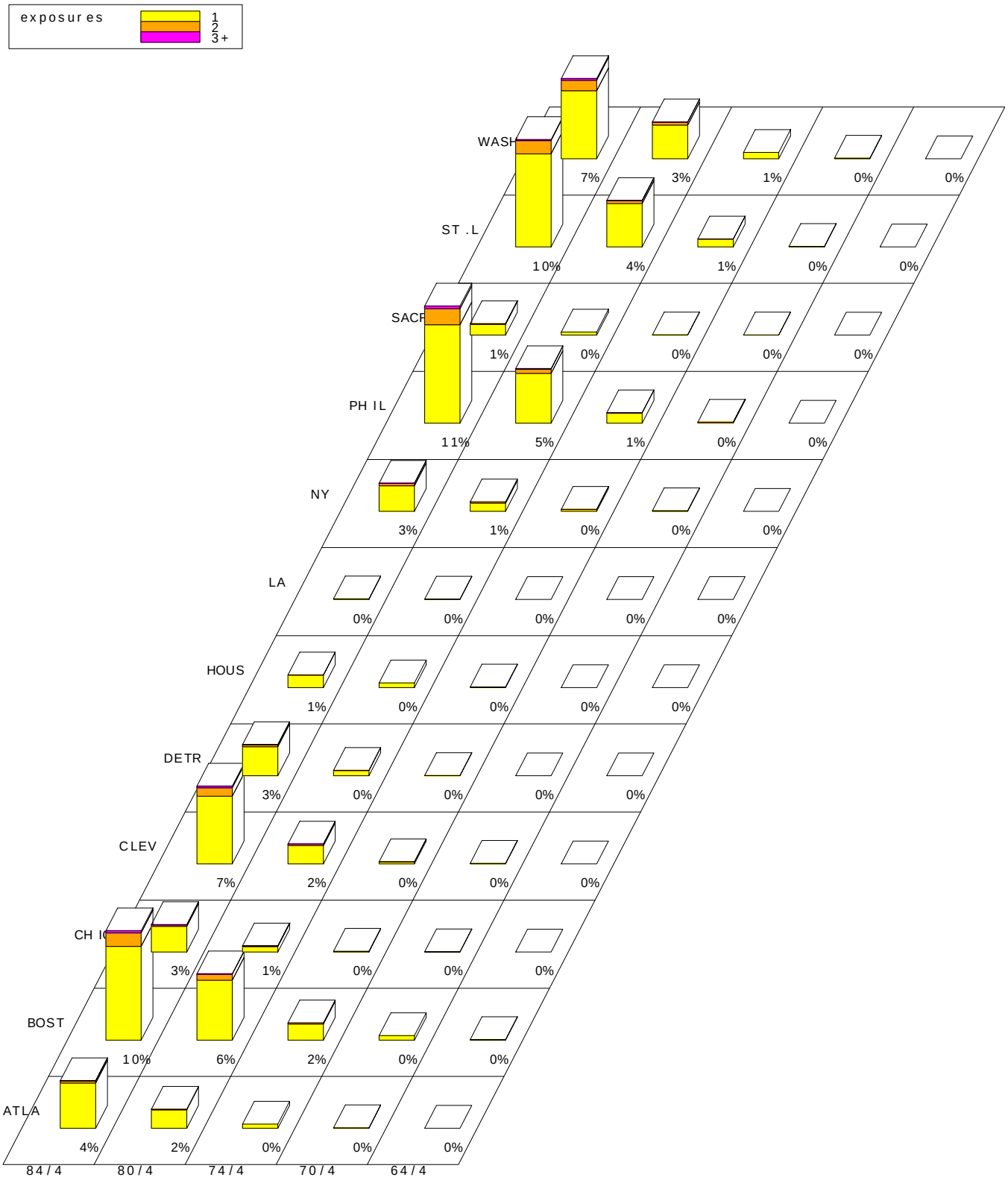


Figure 4-9. Percent of children (ages 5-18) with repeated 8-hr ozone exposures > 0.08 ppm-8hr, for exposures concomitant with moderate or greater exertion, based on 2002 air quality data.

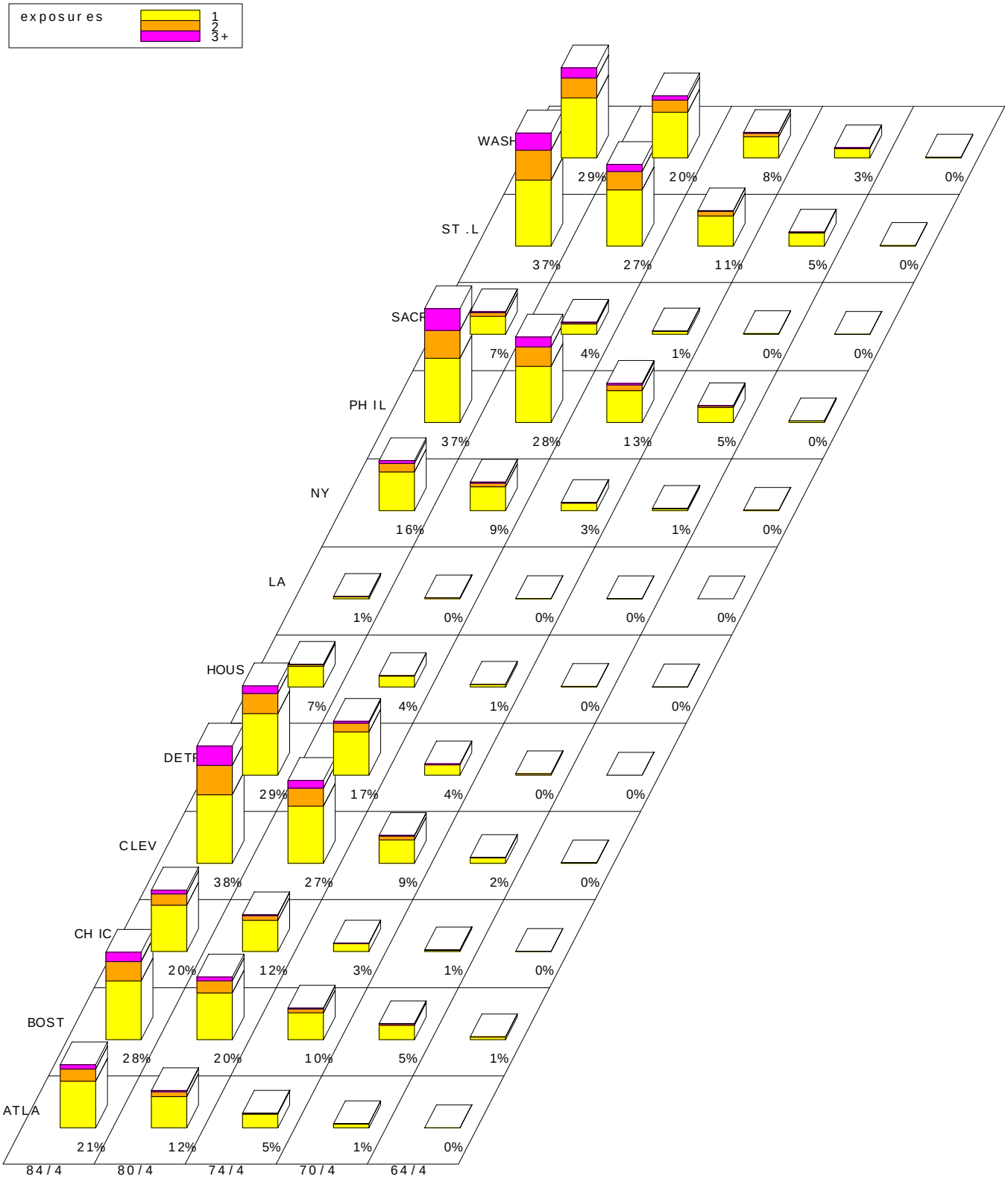


Figure 4-10. Percent of children (ages 5-18) with repeated 8-hr ozone exposures > 0.07 ppm-8hr, for exposures concomitant with moderate or greater exertion, based on 2002 air quality.

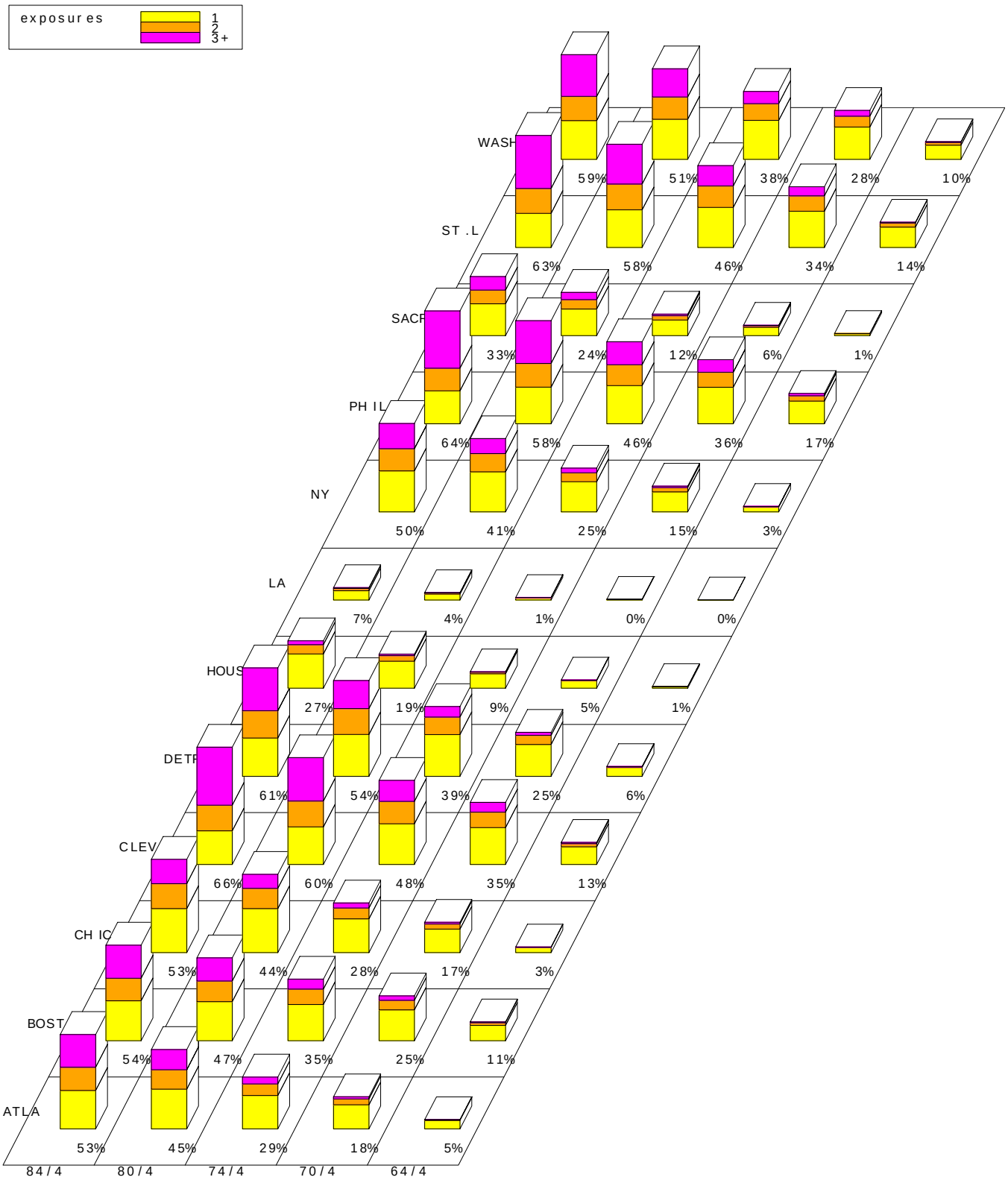


Figure 4-11. Percent of children (ages 5-18) with repeated 8-hr ozone exposures > 0.06 ppm-8hr, for exposures concomitant with moderate or greater exertion, based on 2002 air quality data.

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5. CHARACTERIZATION OF HEALTH RISKS

5.1 INTRODUCTION

This chapter presents information regarding the results from an updated ozone (O_3) health risk assessment that builds upon the methodology used in the assessment conducted as part of the last O_3 NAAQS review. This updated assessment includes estimates of (1) risks of lung function decrements in school age children, respiratory symptoms in asthmatic children, respiratory-related hospital admissions, and respiratory, non-accidental, and cardiorespiratory mortality associated with recent ambient O_3 levels; and (2) risk reductions associated with just meeting the current and several alternative 8-hr O_3 NAAQS. The current risk assessment is more fully described and presented in a technical support document, *Ozone Health Risk Assessment for Selected Urban Areas* (Abt Associates, 2007; henceforth referred to as the Risk Assessment Technical Support Document and cited as Risk Assessment TSD).

The goals of this O_3 risk assessment are: (1) to provide estimates of the potential magnitude of mortality and several morbidity effects associated with current O_3 levels, and with meeting the current 8-hr O_3 NAAQS and alternative O_3 8-hr standards, in specific urban areas; (2) to develop a better understanding of the influence of various inputs and assumptions on the risk estimates; and (3) to gain insights into the distribution of risks and patterns of risk reductions associated with meeting alternative O_3 standards. The risk assessment covers a variety of health effects for which there is adequate information to develop quantitative risk estimates. However, there are several health endpoints (e.g., increased lung inflammation, increased airway responsiveness, impaired host defenses, increased medications use, increased asthma-elated emergency department visits, increased school absences) for which there currently is insufficient information to develop quantitative risk estimates. These additional health endpoints are discussed qualitatively in Chapter 3 of this Staff Paper. We recognize that while there are many sources of uncertainty and variability inherent in the inputs to this assessment, which make the specific estimates uncertain, there is sufficient confidence in the direction and general magnitude of the estimates provided by the assessment, particularly with respect to relative differences between alternative potential standards, for the assessment to serve as a useful input to decisions on the adequacy of the O_3 standard. While some of these uncertainties have been addressed quantitatively in the form of estimated confidence ranges around central risk estimates, other uncertainties and the variability in key inputs are not reflected in these confidence ranges, but rather are addressed through separate sensitivity analyses or are characterized qualitatively.

Following this introductory section, this chapter discusses the scope of the risk assessment, including selection of urban areas and health endpoints and the degree of confidence associated with the various health outcomes that have been associated with ambient O_3

exposures; components of the risk model; characterization of uncertainty and variability associated with the risk estimates; and key results from the assessment. The Risk Assessment TSD provides a more detailed discussion of the risk assessment methodology and includes additional risk estimates beyond those summarized herein.

5.1.1 Overview of Risk Assessment From Last Review

EPA conducted a health risk assessment that produced risk estimates for the number and percent of children and outdoor workers experiencing lung function and respiratory symptoms associated with O_3 exposures for 9 urban areas. This portion of the risk assessment was based on exposure-response relationships developed from analysis of data from several controlled human exposure studies which was combined with exposure estimates developed for children who spent more time outdoors and for outdoor workers. The risk assessment for the last review also included risk estimates for excess respiratory-related hospital admissions related to O_3 concentrations for New York City, based on a concentration-response relationship reported in an epidemiological study (Thurston et al., 1992). Risk estimates for lung function decrements, respiratory symptoms, and hospital admissions were developed associated with recent air quality levels (referred to as "as is" air quality) and for just meeting the existing 1-hr standard and several alternative 8-hr standards. The methodological approach followed in conducting the last risk assessment and risk estimates resulting from that assessment are described in Chapter 6 of the 1996 Staff Paper (U.S. EPA, 1996b) and in several technical reports and publications (Whitfield et al., 1996; Whitfield, 1997; Whitfield et al., 1998).

In the 1997 review of the O_3 NAAQS, the risk estimates played a significant role in both the staff recommendations and in the proposed and final decisions to revise the O_3 standards. CASAC stated (Wolff, 1995) in its advice and recommendations to the Administrator on the O_3 Staff Paper that "EPA's risk assessments must play a central role in identifying an appropriate level," while also noting that "because of the myriad of assumptions that are made to estimate population exposure and risk, large uncertainties exist in these estimates." In the 1997 notice (62 FR 38856) announcing the decision to revise the O_3 standards, EPA indicated that the Administrator considered the results of the exposure and risk analyses and key observations and conclusions from these analyses in putting effects considered to be adverse to individuals into a broader public health perspective and in making judgments about the level of a standard that would be requisite to protect public health with an adequate margin of safety.

5.1.2 Development of Approach for Current Risk Assessment

The health risk assessment described in this chapter and in the Risk Assessment TSD builds upon the methodology and lessons learned from the risk assessment work conducted for the last review. The current risk assessment also is based on the information evaluated in the

final CD. The general approach used in the current risk assessment was described in the draft Health Assessment Plan (U.S. EPA, 2005a), that was released to the CASAC and general public in April 2005 for review and comment and which was the subject of a consultation with the CASAC O_3 Panel on May 5, 2005. The approach used in the current risk assessment reflects consideration of the comments offered by CASAC members and the public on the draft Health Assessment Plan, comments offered by CASAC members and the public on the first and second drafts of the Staff Paper and first and second drafts of the Risk Assessment TSD at and subsequent to a consultation with CASAC on December 8, 2005, and at and subsequent to a review by CASAC on August 24-25, 2006. CASAC comments reflecting both the Ozone Panel's views and additional comments by individual members were provided to the Agency in letters dated February 16, 2006 (Henderson, 2006a), June 5, 2006 (Henderson, 2006b) and October 24, 2006 (Henderson, 2006c). This risk assessment chapter indicates where significant new information has been added since the second draft Staff Paper.

The basic structure of the current risk assessment reflects the two different types of human studies on which the O_3 health risk assessment is based: controlled human exposure studies and epidemiological studies. Controlled human exposure studies involve volunteer subjects who are exposed while engaged in different exercise regimens to specified levels of O_3 under controlled conditions for specified amounts of time. For the current health risk assessment, we are using probabilistic exposure-response relationships based on analysis of individual data that describe the relationship between a measure of personal exposure to O_3 and measures of lung function recorded in the studies. The measure of personal exposure to ambient O_3 is typically some function of hourly exposures – e.g., 1-hr maximum or 8-hr maximum. Therefore, a risk assessment based on exposure-response relationships derived from controlled human exposure study data requires estimates of personal exposure to ambient O_3, typically on a 1-hr or multi-hour basis. Because data on personal hourly exposures to O_3 of ambient origin are not available, estimates of personal exposures to varying ambient concentrations are derived through exposure modeling, as described in Chapter 4. While the quantitative risk assessment based on controlled human exposure studies addresses only lung function responses, it is important to note that other respiratory responses have been found to be related to O_3 exposures in these types of studies, including increased lung inflammation, increased respiratory symptoms, increased airway responsiveness, and impaired host defenses. Chapter 3 of this Staff Paper provides a more complete discussion of these additional health endpoints which are an important part of the overall characterization of risks associated with ambient O_3 exposures.

In contrast to the **exposure-response** relationships derived from controlled human exposure studies, epidemiological studies provide estimated **concentration-response** relationships based on data collected in real world community settings. Ambient O_3

concentrations, measured as the average of monitor-specific measurements, using population-oriented monitors, are used as a surrogate measure of population exposure. It is important to consider that O_3 in ambient air is present in a complex mixture of air pollutants, and that some other components of the mixture may play an important role in some of the health-related effects observed. It is also important to recognize that population health responses included in the quantitative risk assessment for O_3 (i.e., respiratory symptoms in asthmatic children, hospital admissions for respiratory illness, and premature mortality) represent only a portion of the health effects that are associated with O_3 exposures. As discussed more fully in Chapter 3, a wide variety of respiratory and cardiorespiratory effects have been shown to be related to O_3 exposures including increased medication usage in asthmatics, increased doctor's visits and emergency department visits, and increased school absences. As described more fully below, a risk assessment based on epidemiological studies typically requires baseline incidence rates and population data for the risk assessment locations.

The characteristics that are relevant to carrying out a risk assessment based on controlled human exposure studies versus one based on epidemiology studies evaluated in the CD can be summarized as follows:

- The relevant controlled human exposure studies in the CD provide data that can be used to estimate exposure-response functions, and therefore a risk assessment based on these studies requires as input (modeled) personal exposures to ambient O_3. The relevant epidemiological studies in the CD provide concentration-response functions, and, therefore, a risk assessment based on these studies requires as input (actual monitored or adjusted based on monitored) ambient O_3 concentrations, and personal exposures are not required as inputs to the assessment.
- Epidemiological studies are carried out in specific real world locations (e.g., specific urban areas). To minimize uncertainty, a risk assessment based on epidemiological studies has been performed for the locations in which the studies were carried out. Controlled human exposure studies, carried out in laboratory settings, are generally not specific to any particular real world location. A risk assessment based on controlled human exposure studies can therefore appropriately be carried out for any location for which there are adequate air quality and other data on which to base the modeling of personal exposures. There are, therefore, some locations for which a risk assessment based on controlled human exposure studies could appropriately be carried out but a risk assessment based on epidemiological studies would involve greater uncertainty.
- The adequate modeling of hourly personal exposures associated with ambient concentrations for use with exposure-response relationships requires more complete ambient monitoring data than are necessary to estimate average ambient concentrations used to calculate risks based on concentration-response relationships. Therefore, there may be some locations in which an epidemiological studies-based risk assessment could appropriately be carried out, but a controlled human exposure studies-based risk assessment would involve greater uncertainty.

- To derive estimates of risk from concentration-response relationships estimated in epidemiological studies, it is usually necessary to have estimates of the baseline incidences of the health effects involved. Such baseline incidence estimates are not needed in a controlled human exposure studies-based risk assessment.

The scope of the current O_3 risk assessment is described in the next section along with air quality considerations that are relevant to both parts of the risk assessment. Then, the methods for the two parts of the risk assessment – the part based on controlled human exposure studies and the part based on epidemiological and field studies – are discussed in sections 5.3.1 and 5.3.2 below, followed by presentation and discussion of the O_3 risk estimates in section 5.4. Both parts of the risk assessment were implemented within a new probabilistic version of TRIM.Risk, the component of EPA's Total Risk Integrated Methodology (TRIM) model that estimates human health risks.

5.2 SCOPE OF OZONE HEALTH RISK ASSESSMENT

The current O_3 health risk assessment estimates risks of various health effects associated with exposure to ambient O_3 in a number of urban areas selected to illustrate the public health impacts of this pollutant. The short-term exposure related health endpoints selected for the O_3 risk assessment, discussed in section 5.2.1, include those for which the CD concludes that the evidence as a whole supports the general conclusion that O_3, acting alone and/or in combination with other components in the ambient air pollution mix is likely causal.[1]

As discussed in section 3.7, we recognize that there are varying levels of confidence that various health effect endpoints are associated with O_3 at ambient levels. As discussed in section 3.7.5 there is clear evidence of a causal relationship between lung function decrements and O_3 exposures for school age children engaged in moderate exertion for 6 to 8-hours based on the numerous controlled human exposure studies and summer camp field studies conducted by various investigators over the last 30 years. We also judge that there is clear evidence of a causal relationship between increased respiratory symptoms in moderate to severe asthmatic children and O_3 exposures. There also is strong evidence of a causal relationship between increased respiratory-related hospital admissions and O_3 exposure during the warm O_3 season, based on extensive and fairly consistent epidemiological studies as well as evidence from controlled human exposure studies reporting increased lung inflammation and airway responsiveness.

The CD concludes that there is strong evidence which is highly suggestive of a causal relationship between respiratory-related, non-accidental, and cardiorespiratory-related mortality

[1]As discussed in 5.2.1, certain endpoints met this criteria of likely causality, but were not included in the risk assessment for other reasons, such as insufficient exposure-response data or lack of baseline incidence data.

and O_3 exposures during the warm O_3 season. Our judgment with respect to these health outcomes is based on the fairly consistent positive associations found between elevated warm O_3 season levels and these mortality outcomes even when the effect of PM is controlled for, and supporting evidence about potential mechanisms of effects on the cardiovascular system from animal toxicology, human clinical and epidemiological studies. There is certainly greater uncertainty about these outcomes than the other effects discussed above. We also recognize, as discussed in section 3.7.5, that for some of the effects observed in epidemiological studies, such as increased respiratory-related hospital admissions and non-accidental and cardiorespiratory mortality, O_3 may be serving as an indicator for reactive oxidant species in the overall photochemical oxidant mix and that these other constituents may be responsible in whole or part for the observed effects.

The current risk assessment includes risk estimates for 12 urban areas. The basis for selection of these areas is discussed below (section 5.2.2).

Another important aspect of the current risk assessment is that the risks estimated are only those associated with ambient O_3 concentrations exceeding estimated policy-relevant background (PRB) levels (hereafter, referred to as either "background" or "PRB" in this chapter).[2] Risks associated with concentrations above this background are judged to be more relevant to policy decisions about the NAAQS than estimates that include risks potentially attributable to uncontrollable background concentrations.

5.2.1 Selection of Health Endpoint Categories

As noted above, in the last review a significant portion of the health risk assessment involved developing risk estimates for both lung function decrements (≥ 10, ≥ 15, and $\geq 20\%$ changes in FEV_1) and respiratory symptoms in children (age 6 to 18 years old) who spend more time outdoors and outdoor workers with 1-hr exposures at moderate and heavy exertion and 8-hr exposures at moderate exertion. As discussed in section 3.3.1.1 and Chapter 6 of the CD, there is a significant body of controlled human exposure studies reporting lung function decrements and respiratory symptoms in adults associated with 1- and 6 to 8-hr exposures to O_3, as well as similar responses in outdoor workers and others engaged in recreational outdoor activities.

Consistent with the approach used in the last review, we judge that it is reasonable to estimate exposure-response relationships for lung function decrements associated with O_3 exposures in children 5-18 years old based on data from young adult subjects (18-35 years old). As discussed in the 1996 Staff Paper and 1996 CD, findings from other chamber studies

[2]Policy relevant background is defined in section 2.7 of this Staff Paper and development of estimates for policy relevant background for use in the risk assessment is discussed in section 5.2.3.

(McDonnell et al., 1985) for children 8-11 years old at a single exposure level and summer camp field studies in at least six different locations in the U.S. and Canada found lung function decrements in healthy children similar to those observed in healthy adults exposed to O_3 under controlled chamber conditions. The same approach is being used in the current assessment. In the prior risk assessment, staff focused on the risk estimates for lung function decrements associated with 1-hr heavy exertion, 1-hr moderate exertion, and 8-hr moderate exertion exposures in children age 5-18 years of age. Since the 8-hr moderate exertion exposure scenario in children who spend more time outdoors clearly resulted in the greatest health risks in terms of both the magnitude of the lung function decrements and the percent of the population estimated to experience these effects, and since no new information published since the last review suggests any changes that would impact this conclusion, we have included only the lung function decrements ($\geq$ 10, 15, and 20% FEV_1) associated with 8-hr moderate exertion exposures in children and asthmatic children (age 5 to 18 years old) in the current risk assessment.[3] Risk estimates for asthmatic school age children have been added to the risk assessment since the second draft Staff Paper based on comments offered by the CASAC emphasizing the importance of health effects for this population.

While outdoor workers and other adults who engage in moderate exertion for prolonged periods during the day also are clearly at risk for experiencing similar lung function responses when exposed to elevated ambient O_3 concentrations, the exposure and risk assessment conducted during the prior review suggested that school age children are at greatest risk in terms of the number of individuals likely affected. Given the lack of information about the number of individuals who regularly work or exercise outdoors, we chose to focus the current quantitative risk assessment for lung function decrements on all and asthmatic school age children. Therefore, it is important to recognize that the current risk assessment does not account for all of the lung function effects in the general population that would be associated with exposure to O_3 under the various air quality scenarios examined.

Although respiratory symptoms in healthy children were estimated in the last review, we have not included this endpoint in the current quantitative risk assessment. This is because

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[3]Subsequent to completion of the Risk Assessment TSD, EPA analysis of uncertainty of the exposure modeling results uncovered an error in how children are characterized as active. This error resulted in an overestimate of the number of active children included in the exposure estimates which are an input to the lung function risk estimates for active children. Thus, the lung function risk estimates provided for active children in the Risk Assessment TSD are not accurate and we did not include risk estimates for active children in this chapter.

several field studies conducted since the last review did not observe respiratory symptoms in field studies examining responses in healthy children. The CD concludes that "collectively, these studies indicate that there is no consistent evidence of an association between O_3 and respiratory symptoms among healthy children" (CD, p. 7-55). Thus, we decided to limit this portion of the risk assessment to lung function decrements in children and to again base the exposure-response relationships on data obtained for 18-35 year old subjects.

While a number of controlled human exposure studies have reported additional health endpoints associated with short-term exposures to O_3, including airway hyperresponsiveness, inflammation, and immune system effects, there is insufficient exposure-response data at different concentrations to develop quantitative risk estimates for these effects. These important additional effects are discussed in Chapter 3, and we want to emphasize that the current quantitative risk assessment presents only a partial picture of the risks to public health associated with short-term O_3 exposures.

As discussed in the CD and Chapter 3, a significant number of epidemiological studies examining a variety of health effects associated with ambient O_3 concentrations in various locations throughout the U.S., Canada, Europe, and other regions of the world have been published since the last O_3 NAAQS review. Chapter 3 reviews the epidemiological evidence evaluated in Chapter 7 of the CD. In selecting health endpoints to be included in the current quantitative risk assessment, we have focused on health endpoints that are better understood in terms of health consequences (i.e., where there is greater consensus about the degree of response that should be considered as representing an adverse health effect in the population) and endpoint categories for which the weight of the evidence supports the inference of a likely causal relationship between O_3 and the effect category. Certain health endpoints met the criteria of likely causality, but were not included in the risk assessment for other reasons, such as insufficient exposure-response data or lack of baseline incidence data. Based on these considerations, the following endpoints associated with short-term exposures to O_3 during the "warm O_3 season" (April 1 to September 30) have been included:

- Respiratory symptoms in moderate/severe asthmatic children (ages 0 to 12);
- Hospital admissions for respiratory illness and asthma;
- Premature total non-accidental and cardiorespiratory mortality.

As discussed above in section 3.3.1.1.1, the CD also concludes that collectively, the results of epidemiological studies suggest that respiratory symptoms and increased medication use in asthmatic children are associated with acute exposure to O_3. These recent studies provide very strong evidence that asthmatic children experience O_3-related effects.

Large multi-city studies, as well as many studies from individual cities, have reported an association of O_3 concentrations with respiratory-related hospital admissions. Studies with data restricted to the summer or warm season, in general, indicated positive and robust associations between ambient O_3 concentrations and respiratory-related hospital admissions. With respect to acute O_3 effects on mortality, the CD concludes (p.7-175) that, "The majority of the studies suggest an elevated risk of all cause mortality associated with acute exposure to O_3, especially in the summer or warm season when O_3 levels are typically high."

As discussed in Chapter 7 of the CD and in sections 3.3.1.1.1 and 3.3.1.1.6 above, several additional health endpoints including ED visits for respiratory illness and increased school absences have been reported to be associated with short-term O_3 exposures. The current quantitative risk assessment does not include these additional health endpoints. Emergency department visits were excluded from the quantitative risk assessment because of the lack of baseline incidence data for ED visits, as well as the more limited and less consistent database. We also judge that the data reporting an association between short-term O_3 exposures and school absences is too limited to include in the current risk assessment.

Table 5-1 provides a summary of the health effects and the corresponding populations for each health effect which were included in the quantitative risk assessment. Table 5-2 lists some of the health effects that have been associated with elevated O_3 exposures which were not included in the quantitative risk assessment. Chapter 3 provides additional discussion of the health effects not included in this risk assessment.

5.2.2 Selection of Study Areas

The criteria and considerations that went into selection of urban areas for the O_3 risk assessment included the following:

- The overall set of urban locations should represent a range of geographic areas, urban population demographics, and climatology and be focused on areas that do not meet the current 8-hr O_3 NAAQS.
- The largest areas with major O_3 nonattainment problems should be included.
- There must be sufficient air quality data for a recent three year period.
- An area should be the same or close to the location where at least one concentration-response function for the health endpoints included in the assessment has been estimated by a study that satisfies the study selection criteria (see below). If the study was a hospital admissions study, then relatively recent location-specific baseline incidence data had to be available.

Table 5-1. Health Effects and Associated Population Groups Addressed in Quantitative Risk assessment

Health Effect	Population
Lung function decrements (FEV_1)	All school age children (age 5-18) School age asthmatic children (age 5-18)
Respiratory symptoms (chest tightness, shortness of breath, wheeze)	Asthmatic children (age 0 - 12) in Boston
Hospital admissions: -For respiratory illness -Asthma-related -Pneumonia	 -Age 30+ in Los Angeles, Age 65+ in Cleveland, All ages in New York -All ages in New York -Age 65+ in Detroit
Mortality: -Total (not including accidental) -Cardiorespiratory	All ages

Table 5-2. Health Endpoints and Associated Population Groups Not Included in the Quantitative Risk Assessment*

Health Effect	Population
Lung function decrements	Adults (outdoor workers, recreational exercisers, athletes)
Respiratory symptoms (cough, chest discomfort)	Adults (outdoor workers, recreational exercisers, athletes)
School absences for respiratory illness	Children
Asthma-related emergency department visits	Asthmatics
Doctors visits	Adults and children
Lung inflammation	Adults and children
Increased medication usage	Asthmatic children and adults
Decreased resistance to infection, impaired host defense	Adults and children

*The list of health endpoints and populations not included in the risk assessment is not a comprehensive list, but rather provides a general indication of the types of health endpoints that are associated with exposures to ozone but not included in the quantitative risk assessment.

- Locations in which more health endpoints have been assessed were preferred to those with fewer.

Since the exposure-response functions for lung function decrements based on the controlled human exposure studies were based on controlled laboratory conditions, the location of these studies played no role in selecting urban locations for the risk assessment.

Based on the selection criteria and considerations listed above, the following urban areas were included in the risk assessment:

- Atlanta
- Boston
- Chicago
- Cleveland
- Detroit
- Houston
- Los Angeles
- New York City
- Philadelphia
- Sacramento
- St. Louis
- Washington, D.C.

As discussed in Chapter 4, for the purposes of estimating population exposure and the risk of lung function decrements associated with these population exposure estimates, the 12 urban areas have been defined based on consolidated statistical areas (CSAs). The population estimates for these 12 urban area CSAs are given in Table 4-3. About 40% of the total U.S. urban population (88.5 million persons) resides in these 12 urban areas including 18.3 million school age children (ages 5 to 18). In contrast to the risk assessment for lung function decrements, for the risk estimates for premature mortality and excess hospital admissions, the urban areas have been defined to be generally consistent with the geographic boundaries used in the epidemiological studies which were the source of the concentration-response functions used in this risk assessment. In most cases the epidemiological studies only included the core urban county or a limited number of counties in one or more of the 12 urban areas. In addition, estimates of respiratory symptoms in asthmatic children were developed for one urban area (the Boston CSA).

5.2.3 Air Quality Considerations

Both the portion of the risk assessment based on controlled human exposure and the portion based on epidemiological studies include risk estimates for a recent year of air quality (labeled "as is" air quality in the Risk Assessment TSD) and for air quality adjusted so that it simulates just meeting the current and several alternative 8-hr O_3 NAAQS based on a recent

three-year period (2002-2004). This period was selected to represent the most recent air quality data for which complete data were available when the risk assessment was conducted.

In order to estimate health risks associated with just meeting the current and alternative 8-hr O_3 NAAQS, it is necessary to estimate the distribution of hourly O_3 concentrations that would occur under any given standard. Since compliance with the current O_3 standard is based on a 3-year average, air quality data from 2002 to 2004 have been used to determine the amount of reduction in O_3 concentrations required to meet the current standard. Estimated design values[4] are used to determine the adjustment necessary to just meet the current 8-hr daily maximum standard. The amount of control has then been applied to each year of data (2002, 2003, and 2004) to estimate risks for a single O_3 season or single warm O_3 season, depending on the health effect, in each of these individual years. As described in section 4.5.6 and in more detail in Rizzo (2006), after considering several approaches, we concluded that the Quadratic air quality adjustment procedure generally best represented the pattern of reductions across the O_3 air quality distribution observed over the last decade. The Quadratic air quality adjustment procedure was applied in each of the 12 urban areas to the 2002 and 2004 O_3 air quality data and in a subset of 5 urban areas (Atlanta, Chicago, New York, Houston, and Los Angeles) to the 2003 O_3 air quality data, based on the 3-year period (2002-2004) O_3 design values, to generate new time series of hourly O_3 concentrations for 2002, 2003, and 2004 that reflect air quality levels that just meet the current 8-hr O_3 standard over this three year period. Risk estimates associated with 2003 O_3 monitoring data and 2003 air quality adjusted to just meet the current and alternative 8-hr standards have been added to the assessment since the second draft Staff Paper.

We note that since compliance with the current standard is based on the 3-year average of the 4th-highest daily maximum 8-hr values, the air quality distribution in each of the 3 years can, and generally does, vary. As a consequence, the risk estimates associated with air quality just meeting the current standard also will vary depending on the year chosen for the analysis. We include assessments involving adjustment of both 2002 and 2004 air quality data to illustrate the magnitude of this year-to-year variability in the risk estimates. The year 2002 generally had meteorology that was very conducive to producing O_3 over the eastern half of the U.S. and this resulted in the highest O_3 levels over the 2002-2004 time period in the vast majority of the 12 urban study areas. In contrast, 2004 was a year associated with an unusually cool and rainy

[4]A design value is a statistic that describes the air quality status of a given area relative to the level of the NAAQS. Design values are often based on multiple years of data, consistent with the specification of the NAAQS in Part 50 of the CFR. For example, for the current O_3 NAAQS, the 3-year average of the annual 4th-highest daily maximum 8-hr average concentrations, based on the monitor within an urban area yielding the highest 3-yr average, is the design value.

summer in the eastern half of the U.S. and this contributed to the fact that the lowest O_3 levels over this same three-year period were observed in this year in most of the urban areas included in the assessment. The lower O_3 levels observed in 2004 also were lower, in part, as a result of reductions in nitrogen oxides (NO_x) emissions associated with implementation of additional regional controls on large power plants in the eastern half of the U.S. Differences in meteorology were less evident in Texas and California and these latter areas also were not impacted by the recent additional regional controls imposed on large power plants. Thus, its not surprising that the daily maximum 8-hr levels observed in Houston in 2004 were somewhat higher than those observed in 2002 and that 8-hr levels were similar in Los Angeles between these two years. The risk results for 2002 and 2004, thus, provide generally lower-end and upper-end estimates of the annual risks that can occur over a three-year period when alternative standards are just met. Daily maximum 1-hr and 8-hr O_3 levels in 2003 generally fell somewhere between 2002 and 2004 levels in most of the 12 urban areas.

As noted earlier, the risk estimates developed for both the recent air quality scenario and just meeting the current 8-hr standard represent risks associated with O_3 levels in excess of estimated background concentrations. The results of the global tropospheric O_3 model GEOS-CHEM have been used to estimate average background O_3 levels for different geographic regions across the U.S. These GEOS-CHEM simulations include a background simulation in which North American anthropogenic emissions of NO_x, non-methane volatile organic compounds, and carbon monoxide are set to zero, as described in Fiore et al. (2003). We estimated monthly background concentrations for each of the 12 urban areas based on the GEOS-CHEM simulations, including daily diurnal profiles which were fixed for each day of each month during the O_3 season (See Appendix 2-A of this Staff Paper for plots of these estimated background values). The CD and section 2.7 above indicate that background O_3 concentrations at the surface are generally predicted to be in the range of 0.015 to 0.035 ppm in the afternoon, and they decline under conditions conducive to O_3 episodes. They are highest during spring and decline into summer. This range is lower than the estimated range of 0.03 to 0.05 ppm for typical summertime background levels included in the 1996 CD and the single value of 0.04 ppm used for background in the prior risk assessment.

As discussed in section 2.7, the CD summarizes its evaluation of the validity of the GEOS-CHEM model, and states "in conclusion, we estimate that the PRB ozone values reported by Fiore et al. (2003) for afternoon surface air over the United States are likely 10 ppbv too high in the southeast in summer, and accurate within 5 ppbv in other regions and seasons" (CD, p.3-

53). These error estimates are based on comparison of model output with observations for conditions which most nearly reflect those given in the PRB definition (i.e., at the lower end of the probability distribution). For O_3 (cf. Figures 8 and 9 of Fiore et al. (2003) for the southeast and Figure 3 of Fiore et al. (2002) for the northeast) it can be seen that GEOS-CHEM overestimates O_3 for the southeast and underestimates it for the northeast. Sensitivity analyses examining the impact of alternative estimates for background on lung function and mortality risk estimates have been developed since the second draft Staff Paper and are presented in section 5.4.3. As discussed in section 5.4.3, estimated risk reductions associated with alternative standards relative to just meeting the current standard are generally unaffected by the estimated PRB levels within the range of alternative estimates examined.

5.3 COMPONENTS OF THE RISK MODEL

As noted above in section 5.1.2, there are two parts to the health risk assessment: one based on combining information from controlled human exposure studies with modeled population exposure and the other based on combining information from community epidemiological studies with either monitored or adjusted ambient concentrations levels. Section 5.3.1 below discusses the portion of the current risk assessment related to effects reported in controlled human exposure studies and section 5.3.2 below discusses the portion of the current risk assessment related to health effects reported in community epidemiological studies.

5.3.1 Assessment of Risk Based on Controlled Human Exposure Studies

5.3.1.1 General Approach

The major components of the portion of the health risk assessment based on data from controlled human exposure studies are illustrated in Figure 5-1. As shown in Figure 5-1, under this portion of the risk assessment, exposure estimates for a number of different air quality scenarios (i.e, recent year of air quality, just meeting the current 8-hr standard, just meeting alternative standards, and background) are combined with probabilistic exposure-response relationships derived from the controlled human exposure studies to develop risk estimates associated with recent air quality and just meeting the current and alternative standards in excess of background. As discussed above, the health effect included in this portion of the risk assessment is lung function decrement, as measured by FEV_1 in school aged children engaged in moderate exertion for 8 hours. The air quality and exposure analysis components that are integral to this portion of the risk assessment are discussed in greater detail in Chapter 4 and in the Exposure Assessment TSD.

Several risk measures were generated for this portion of the risk assessment. In addition to the estimates of the number of all and asthmatic school age children experiencing one or more occurrences of a lung function decrement $\geq$10, $\geq$15, and $\geq$20% in an O_3 season, risk estimates have been developed for the total number of occurrences of these lung function decrements in these same population groups. The mean number of occurrences per child has been calculated to provide an indicator of the average number of times that a responder would experience the specified effect during an O_3 season. The population sizes for all and asthmatic school age children for each of the 12 urban areas used in this part of the risk assessment are given in Table 4-3 of this Staff Paper. We note that the asthmatic school age children subpopulation is a subset of all school age children, and thus the risk estimates presented for these two groups should not be combined.

A population risk estimate for a given lung function decrement (e.g., $\geq$15% change in FEV_1) is an estimate of the expected number of people who will experience that lung function decrement. Since we are interested in risk estimates associated with O_3 concentrations in excess of background concentrations, the following steps were taken to estimate the risk associated with recent conditions in excess of background: (1) expected risk given the personal exposures associated with recent ambient O_3 concentrations was estimated, (2) expected risk given the personal exposures associated with estimated background ambient O_3 concentrations was estimated, and (3) the latter was subtracted from the former. As shown in Equation 5-1 below, the population risk is then calculated by multiplying the resulting expected risk by the number of people in the relevant population. See Appendix 5B.2 for additional information concerning notation and the derivation of the risk estimate algorithms used in this risk assessment. Because response rates are calculated for 21 fractiles (i.e., 0.01, 0.05, 0.10, 0.15 … 0.50, 0.55, … 0.90, 0.95, 0.99), estimated population risks are similarly fractile-specific.

The risk (i.e., expected fractional response rate) for the k^{th} fractile, R_k is:

$$R_k = \sum_{j=1}^{N} P_j \; x \; (RR_k \mid e_j) \quad - \quad \sum_{i=1}^{N_b} P_i^b \; x \; (RR_k \mid e_i^b) \quad \text{(Equation 5-1)}$$

where:

e_j = (the midpoint of) the jth category of personal exposure to O_3, given recent ambient O_3 concentrations;

e_i^b = (the midpoint of) the ith category of personal exposure to O_3, given background ambient O_3 concentrations;

P_j = the fraction of the population having personal exposures to O_3 concentration of e_j ppm, given recent ambient O_3 concentrations;

P_i^b = the fraction of the population having personal exposures to O_3 concentration of e_i^b ppm, given background ambient O_3 concentrations;

$RR_k \mid e_j$ = k-fractile response rate at O_3 concentration e_j;

$RR_k \mid e_i^b$ = k-fractile response rate at O_3 concentration e_i^b; and

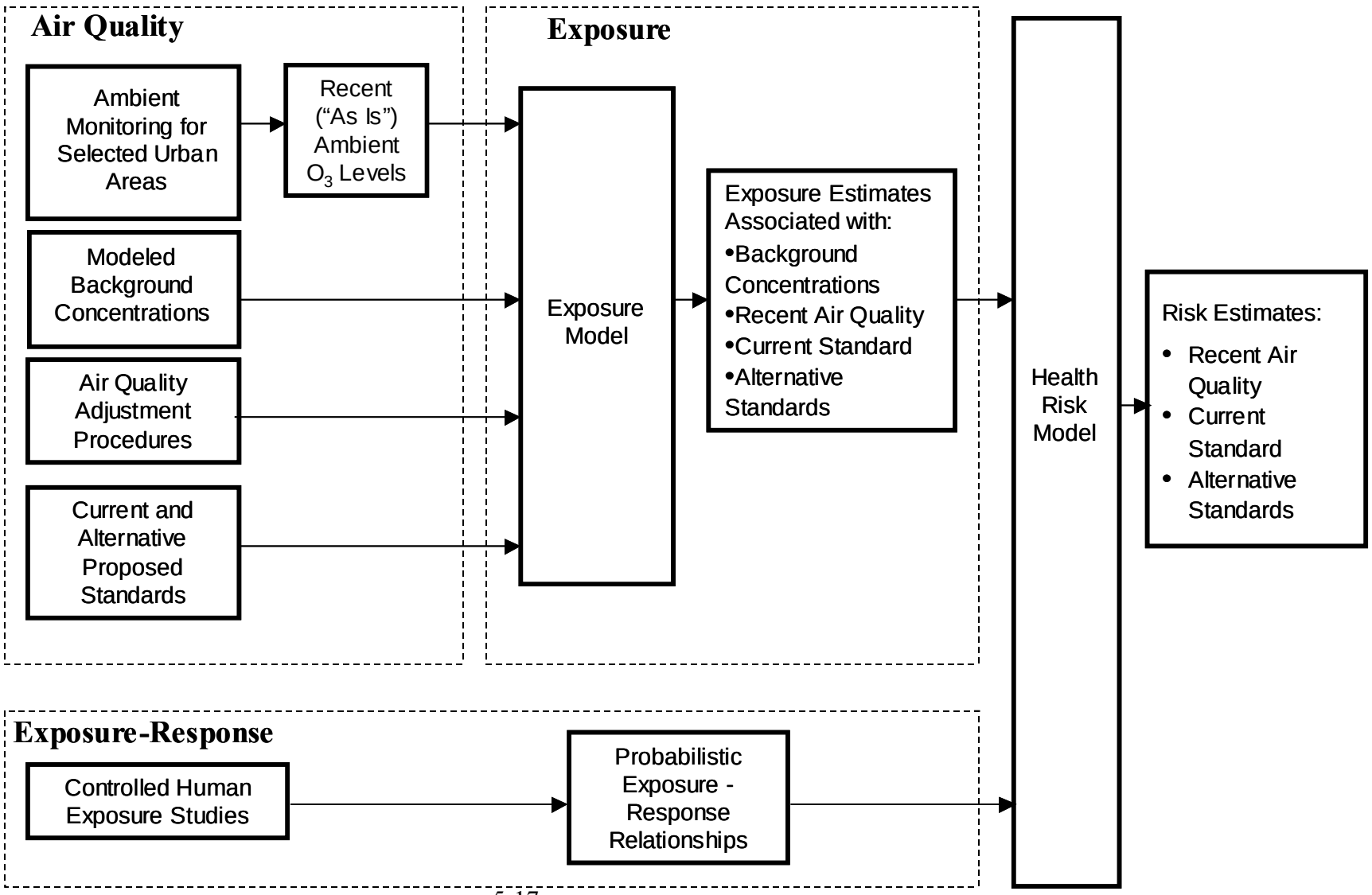
N = number of intervals (categories) of O_3 personal exposure concentration, given recent ambient O_3 concentrations; and

N_b = number of intervals of O_3 personal exposure concentration, given background ambient O_3 concentrations.

For example, if the median expected response rate for recent ambient concentrations is 0.065 (i.e., the median expected fraction of the population responding is 6.5%) and the median expected response rate for background ambient concentrations is 0.001 (i.e., the median expected fraction of the population responding is 0.1%), then the median expected response rate associated with recent ambient concentrations above background concentrations is 0.065 – 0.001 = 0.064. If there are 300,000 people in the relevant population, then the population risk is 0.064 x 300,000 = 19,200.[5]

[5]A normalization procedure had to be applied to the number of responders (or the number of occurrences of response) given personal exposures associated with recent ambient O_3 concentrations (or concentrations rolled back to simulate just meeting a standard) because the population size used in the exposure runs for background concentrations were not identical to those used in the exposure runs conducted for the recent air quality and alternative standard scenarios. This normalization procedure is described in section 3.3.1 of the Risk Assessment TSD.

Figure 5-1. Major Components of Ozone Health Risk Assessment Based on Controlled Human Exposure Studies



5.3.1.2 Exposure Estimates

Exposure estimates used in this portion of the risk assessment were obtained from running TRIM.Expo for each of the 12 urban areas for the various air quality scenarios (i.e., for 2004 and 2002 air quality representing recent years, for 2004 and 2002 air quality adjusted to just meet the current and several potential alternative 8-hr standards, for 2003 air quality and 2003 air quality adjusted to just meet several potential alternative standards in 5 urban areas, and for air quality levels representing background based on estimates from the GEOS-CHEM model).[6] Chapter 4 and the Exposure Assessment TSD (U.S. EPA, 2007) provide additional details about the inputs and methodology used to estimate population exposure in the 12 urban areas. Exposure estimates for all and asthmatic school age children (ages 5 to 18) were separately combined with probabilistic exposure-response relationships for lung function decrements associated with 8-hr exposure while engaged in moderate exertion. Individuals engaged in activities that resulted in an average equivalent ventilation rate (EVR) for the 8-hr period at or above 13 l/min-m^2 were included in the exposure estimates for 8-hr moderate or greater exertion. This range was selected to match the EVR for the group of subjects in the controlled human exposure studies that were the basis for the exposure-response relationships used in this portion of the risk assessment.

5.3.1.3 Exposure-Response Functions

As described in section 3.1.2 of the Risk Assessment TSD, a Bayesian Markov Chain Monte Carlo approach was used to estimate probabilistic exposure-response relationships for lung function decrements associated with 8-hr moderate exertion exposures using the WinBUGS software (Spiegelhalter et al., 1996).[7] The combined data set including the data from the Folinsbee et al. (1988), Horstman et al. (1990), and McDonnell et al. (1991) studies used previously and the more recent data from Adams (2002, 2003, 2006) have been used to estimate exposure-response relationships for 8-hr exposures under moderate exertion for each of the three measures of lung function decrement listed above. Table 5-3 presents a summary of the study-specific results based on correcting all individual responses for the effect on lung function decrements of exercise in clean air. The previously used studies were all conducted in EPA's facility in Chapel Hill, while the Adams studies were conducted at the University of California at

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[6] As noted in chapter 4, a small error was detected in the exposure model in January 2007 that resulted in small increases in the exposure estimates. The lung function risk estimates presented in this final Staff Paper reflect corrected exposure estimates and are generally slightly higher than the original estimates presented in the January 2007 version of the Staff Paper.

[7] See Gelman et al. (1995) or Gilks et al. (1996) for an explanation of these methods.

Table 5-3. Study-Specific Exposure-Response Data for Lung Function Decrements

Study	Protocol	Change in $FEV_1 \geq 10\%$		Change in $FEV_1 \geq 15\%$		Change in $FEV_1 \geq 20\%$	
		Number Exposed	Number Responding	Number Exposed	Number Responding	Number Exposed	Number Responding
0.04 ppm O_3							
Adams (2006)	Triangular	30	0	30	0	30	0
Adams (2002)	Square-wave, face mask	30	2	30	0	30	0
0.06 ppm O_3							
Adams (2006)	Square-wave	30	2	30	0	30	0
	Triangular	30	2	30	2	30	0
0.08 ppm O_3							
Adams (2006)	Square-wave	30	7	30	2	30	1
	Triangular	30	9	30	3	30	1
Adams (2003)	Square-wave, chamber	30	6	30	2	30	1
	Square-wave, face mask	30	9	30	3	30	1
	Variable levels (0.08 ppm avg), chamber	30	6	30	1	30	1
	Variable levels (0.08 ppm avg), face mask	30	5	30	3	30	0
Adams (2002)	Square-wave, face mask	30	6	30	5	30	2
F-H-M*	Square-wave	60	18	60	11	60	5
0.1 ppm O_3							
F-H-M	Square-wave	32	13	32	9	32	5
0.12 ppm O_3							
Adams (2002)	Square-wave, chamber	30	17	30	12	30	10
	Square-wave, face mask	30	21	30	13	30	7
F-H-M	Square-wave	30	15	30**	15**	30	6

*F-H-M includes combined data from Folinsbee et al. (1988), Horstman et al. (1990), and McDonnell et al. (1991) .

**This data point was sufficiently inconsistent with the other data from the F-H-M combined data set that it was considered an outlier and was not included in the analysis.

Davis. Data from these controlled human exposure studies were corrected for the effect of exercise in clean air to remove any systematic bias that might be present in the data attributable to an exercise effect. Generally, this correction for exercise in clean air was small relative to the total effects measures in the O_3-exposed cases.

For the risk assessment conducted during the last O_3 NAAQS review, there were data for only 3 exposure levels (0.08, 0.10, and 0.12 ppm)[8] and a linear exposure-response relationship was estimated for use in the risk assessment. With the addition of data from three more recent Adams studies[9] that included 0.04, 0.06, and/or 0.08 ppm, 6.6 hour exposures, the combined data set appears to be more S-shaped, although there is still considerable uncertainty about the overall functional form, given the limited data at exposure levels below 0.08 ppm. Consistent with advice from the CASAC O_3 Panel in its October 24, 2006 letter (Henderson, 2006c), EPA considered both linear and logistic functional forms in estimating the exposure-response relationship and revised this aspect of the assessment by adopting a Bayesian Markov Chain Monte Carlo approach. This Bayesian estimation approach incorporated both model uncertainty and uncertainty due to sampling variability.

We chose a 90 percent logistic/10 percent linear split as the base case for the current risk assessment based on the following considerations: 1) the prior 1997 risk assessment had used a linear form consistent with the advice from the CASAC O_3 Panel at the time that a linear model reasonably fit the available lung function response data at 0.08, 0.10, and 0.12 ppm from three 6.6 hour exposure studies, 2) with the addition of data at 0.06 and 0.04 ppm, a logistic model provided a very good fit to the data, and 3) as members of the current CASAC O_3 Panel have noted there is only very limited data at the two lowest exposure levels and, therefore, a linear model cannot entirely be ruled out. We have included a sensitivity analysis that examines the impact on the lung function risk estimates of two alternative choices , an 80 percent logistic/20% linear split and 50% logistic/50% linear split (see section 5.4.3.2).

For each of the three measures of lung function decrement, we assumed for the base case a 90% probability that the exposure-response function has the following 3-parameter logistic form:[10,11]

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[8]The studies conducted in EPA's facility in Chapel Hill that are considered in the lung function risk assessment measured O_3 concentrations to within +/- 5% or +/- 0.004 ppm at the 0.08 ppm exposure level.

[9]These studies reported O_3 concentrations to be accuate within +/- 0.003 ppm over the range of concentrations included in these studies.

[10]As noted in Whitfield et al., 1996, the response data point associated with 0.12 ppm for the response measure $FEV_1 \geq 15\%$ appeared to be inconsistent with the other data points (see Whitfield et al., 1996, Table 10, footnote c). Because of this, we estimated the probability of a response of $FEV_1 \geq 15\%$ at an O_3 concentration of 0.12 ppm by interpolating between the $FEV_1 \geq 10\%$ and $FEV_1 \geq 20\%$ response rates at that O_3 concentration.

$$y(x;\alpha,\beta,\gamma) = \frac{\alpha * e^{\gamma}(1-e^{\beta x})}{(1+e^{\gamma})(1+e^{\beta x+\gamma})}, \qquad \text{(Equation 5-2)}$$

where x denotes the O_3 concentration (in ppm) to which the individual is exposed, y denotes the corresponding response (decrement in $FEV_1 \geq 10\%$, $\geq 15\%$ or $\geq 20\%$), and α, β, and γ are the three parameters whose values are estimated.

We assumed for the base case a 10 percent probability that the exposure-response function has the following linear (hockeystick) form:

$$y(x;\alpha,\beta) = \begin{cases} \alpha + \beta x, & for\ \alpha + \beta x > 0 \\ 0, & for\ \alpha + \beta x < 0 \end{cases} \qquad \text{(Equation 5-3)}$$

We assumed that the number of responses, S, out of N subjects exposed to a given concentration, x, has a binomial distribution with response probability given by model (5-1) with 90 percent probability and response probability given by model (5-2) with 10 percent probability. In some of the controlled human exposure studies, subjects were exposed to a given O_3 concentration more than once – for example, using a square-wave exposure pattern in one protocol and a triangular exposure pattern in another protocol. However, because there were insufficient data to estimate subject-specific response probabilities, we assumed a single response probability (for a given definition of response) for all individuals and treated the repeated exposures for a single subject as independent exposures in the binomial distribution.

For each of the two functional forms (logistic and linear), we derived a Bayesian posterior distribution using this binomial likelihood function in combination with prior distributions for each of the unknown parameters. We assumed lognormal priors with maximum likelihood estimates of the means and variances for the parameters of the logistic function, and normal priors, similarly with maximum likelihood estimates for the means and variances, for the parameters of the linear function. For each of the two functional forms considered, we used 1000 iterations as the "burn-in" period followed by 9,000 iterations for the estimation. Each iteration corresponds to a set of values for the parameters of the (logistic or linear) exposure-response function. We then combined the 9,000 sets of values from the logistic model runs with the last 1,000 sets of values from the linear model runs to get a single combined distribution of 10,000 sets of values reflecting the 90 percent/10 percent assumption stated above. As noted

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[11]The 3-parameter logistic function is a special case of the 4-parameter logistic, in which the function is forced to go through the origin, so that the probability of response to 0.00 ppm is 0.

above, sensitivity analyses examining the impact on lung function risk estimates of two alternative choices are presented in section 5.4.3.2.

For any O_3 concentration, *x*, we could then derive the n^{th} percentile response value, for any n, by evaluating the exposure-response function at *x* using each of the10,000 sets of parameter values (9,000 of which were for a logistic model and 1,000 of which were for a linear model). The resulting 2.5^{th} percentile, median (50^{th} percentile), and 97.5^{th} percentile exposure-response functions for changes in $FEV_1 \geq 10\%$, $\geq 15\%$ and $\geq 20\%$ are shown separately in Figure 5-2a, b, and c along with the response data to which they were fit.

We note that the fraction of the population experiencing $FEV_1 \geq 10$, 15, and $\geq 20\%$ associated with 0.08 ppm O_3 exposures was generally lower in the three Adams studies compared to the combined data set based on the studies by Folinsbee et al. (1991), Horstmann et al. (1990), and McDonell et al. (1991). For example, the fraction of the population experiencing FEV_1 decrements $\geq$15% associated with 0.08 ppm O_3 exposures ranged from 3.3 to16.7% in the three Adams studies compared to 18.3% in the combined data set from the Chapel Hill studies. The 0.08 ppm level is the only common level tested in both sets of studies. This observed difference may be due to differences in sensitivity of the subjects tested, random variability due to the relatively small number of subjects tested, and/or possibly greater attenuation of response for subjects living in or near Davis, California (where the Adams studies were conducted) compared to subjects living in or near Chapel Hill, NC (where the other studies were conducted). Adams notes in his studies that they were conducted over a 6-month period when the 0.09 ppm, 1-hr California standard was not exceeded in the area where his subjects resided. The difference in observed responses between these two sets of studies is an additional uncertainty that should be considered.

As noted above, the Risk Assessment TSD includes risk estimates for all three measures of lung function response (i.e., $\geq$ 10, 15, and 20% decrements in FEV_1). However, we are focusing on FEV_1 decrements $\geq$15% for all school age children and $\geq$10% FEV_1 decrements for asthmatic school age children in this Staff Paper, consistent with the advice from CASAC expressed in its October 24, 2006 letter (Henderson, 2006c) that these levels of response represent indicators of adverse health effects for these populations.

5.3.1.4 Characterizing Uncertainty and Variability

An important issue associated with any population health risk assessment is the characterization of uncertainty and variability. *Uncertainty* refers to the lack of knowledge regarding both the actual values of model input variables (parameter uncertainty) and the physical systems or relationships (model uncertainty – e.g., the shapes of concentration-response functions). In any risk assessment, uncertainty is, ideally, reduced to the maximum extent

possible, but significant uncertainty often remains. It can be reduced by improved measurement and improved model formulation. In addition, the degree of uncertainty can be characterized, sometimes quantitatively. For example, the statistical uncertainty surrounding the estimated O_3 coefficients in the exposure-response functions is reflected in the credible intervals provided for the risk estimates in this chapter and in the Risk Assessment TSD.

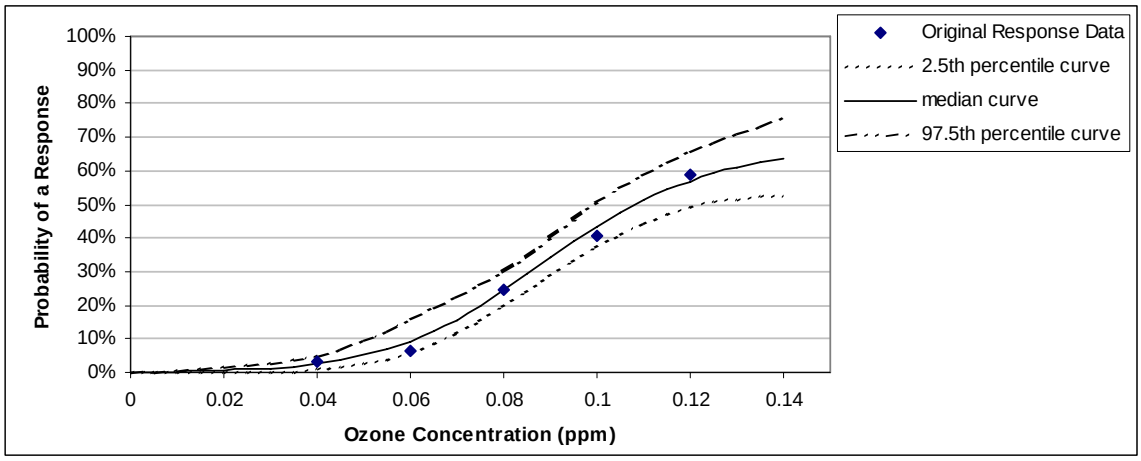
As described in section 3.1.3 of the Risk Assessment TSD and section 5.3.1.3 above, we have revised the approach used in the second draft Staff Paper and have now used a Bayesian Markov Chain Monte Carlo approach to characterize uncertainty attributable to sampling error. Using this approach, we could derive the n^{th} percentile response value, for any n, for any O_3 concentration, x, as described above. Because the exposure estimates were generated at the midpoints of 0.01 ppm intervals (i.e., for 0.005 ppm, 0.015 ppm, etc), we derived 2.5^{th} percentile, 50^{th} percentile (median), and 97.5^{th} percentile response estimates for O_3 concentrations at these midpoint values. As illustrated in Figure 5-2a, b, and c, for each of the lung function response definitions, the 2.5^{th} percentile and 97.5^{th} percentile response estimates comprise the lower and upper bounds of the credible interval around each point estimate (median estimate) of response.

As noted above, the exposure-response relationships shown in Figures 5-2a, b, and c are based on the assumption that the relationship between exposure and response has a logistic form with 90 percent probability and a linear (hockeystick) form with 10 percent probability. The resulting 2.5^{th} percentile, median (50^{th} percentile), and 97.5^{th} percentile exposure-response functions for decrements in $FEV_1 \geq 10\%$ and $\geq 15\%$ for the base case and two alternative exposure-response functions, based on an 80 percent logistic/20 percent linear split and a 50 percent logistic/50 percent linear split are shown in Figures 5-3a and b To aid comparison between the base case and the two alternative exposure-response functions, Figures 5-4a and b show the median exposure-response relationships in the same graph for these same two FEV_1 endpoint definitions for the base case and two alternative exposure-response functions. Section 5.4.4 presents results from a sensitivity analysis that examines the impact of using these two alternative exposure-response relationships on the lung function risk estimates for all and asthmatic school age children.

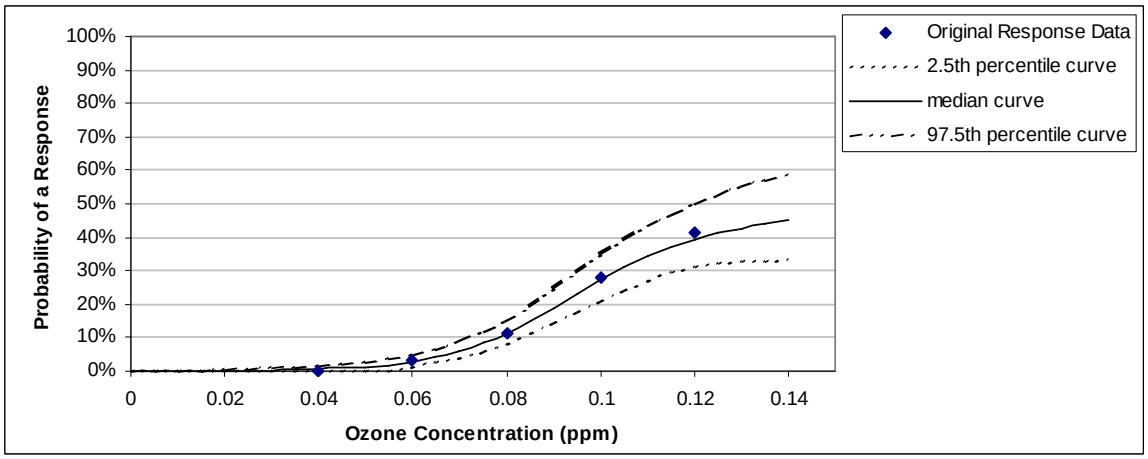
In addition to uncertainties arising from sampling variability considerations, there are other uncertainties associated with the use of the exposure-response relationships for lung function responses. For example, while we have used the combined data set for the current risk assessment, as it represents the best available data, we believe that the observed differences in response between the Adams studies and the Chapel Hill studies contribute to additional uncertainty about the exact shape of the exposure-response relationship, especially for levels at

Figure 5-2. a, b, c. Probabilistic Exposure-Response Relationships for FEV_1 Decrement ≥ 10%, ≥ 15%, and ≥ 20% for 8-Hour Exposures Under Moderate Exertion*

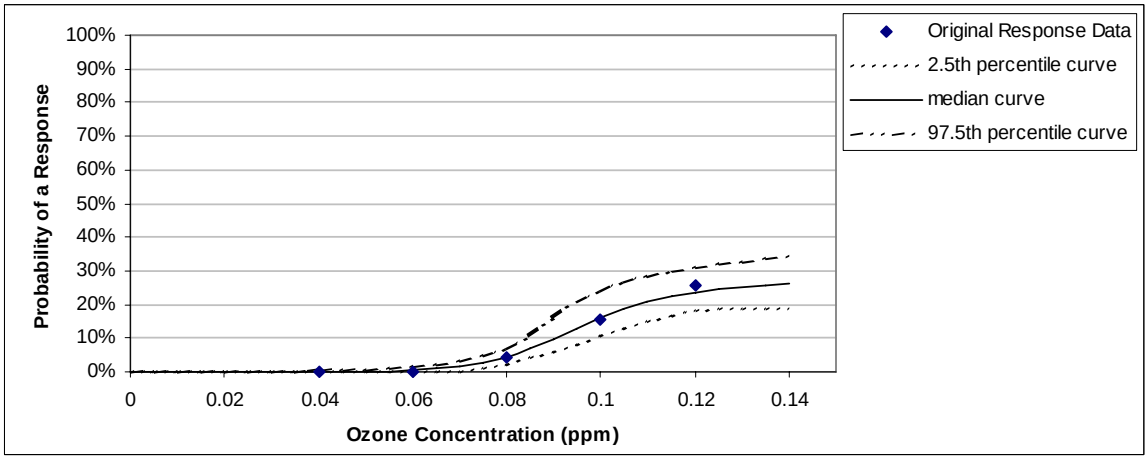
a) FEV_1 Decrement ≥ 10%



b) FEV_1 Decrement ≥ 15%



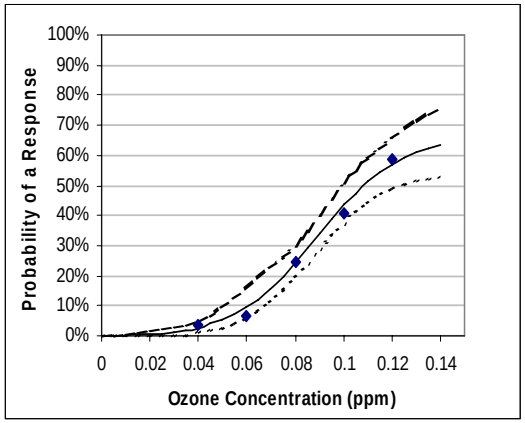
c) FEV_1 Decrement ≥ 20%



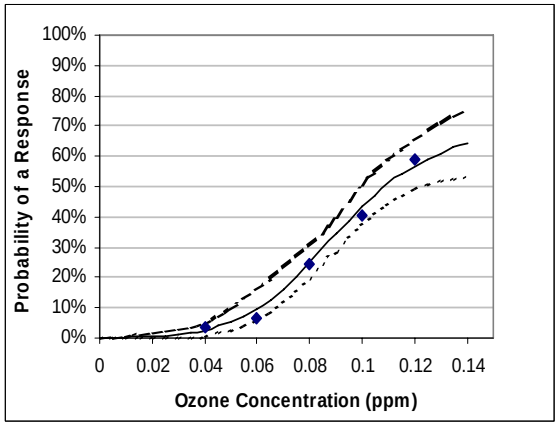
*Derived from Folinsbee et al., 1988; Horstman et al. 1990; McDonnell et al., 1991; Adams 2002, 2003, 2006). Each curve assumes a 90% probability that the form of the exposure-response relationship is logistic and 10% probability that the form is linear (see text above).

Figure 5-3a, b, c. Probabilistic Exposure-Response Relationships for FEV_1 Decrement ≥ 10% and ≥ 15% for 8-Hour Exposures Under Moderate Exertion: Comparison of 90% Logistic/10% Linear (Hockeystick) Split and 80% Logistic/20% Linear (Hockeystick) and 50% Logistic/50% Linear (Hockeystick) Splits in Assumed Relationship Between Exposure and Response*

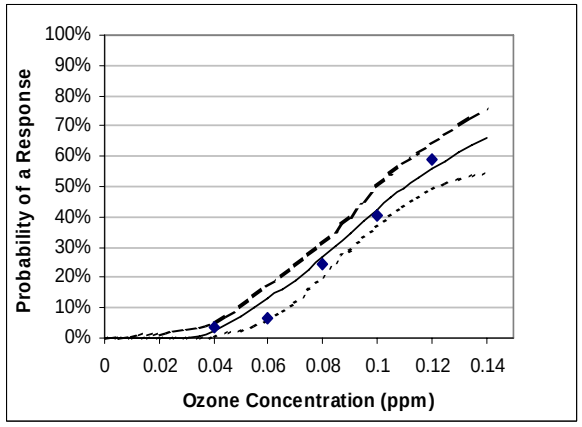
FEV_1 Decrement ≥ 10%: 90% Logistic/10% Linear



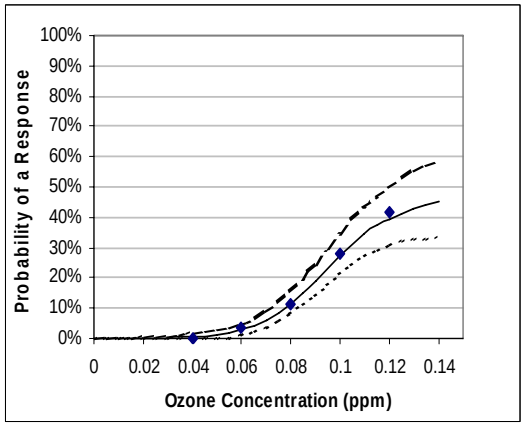
FEV_1 Decrement ≥ 10%: 80% Logistic/20% Linear



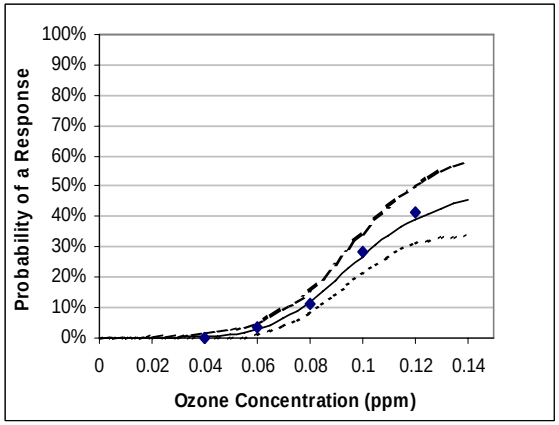
FEV_1 Decrement ≥ 10%: 50% Logistic/50% Linear



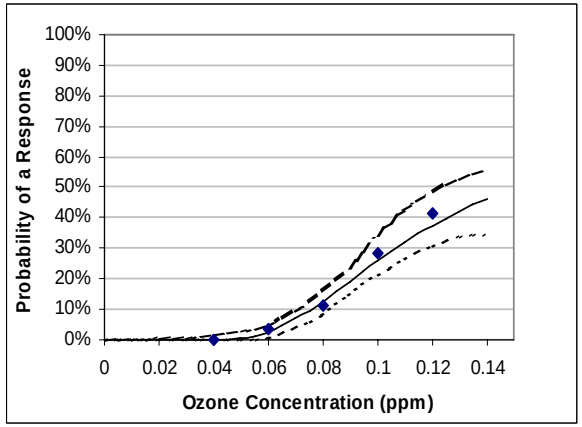
FEV_1 Decrement ≥ 15%: 90% Logistic/10% Linear



FEV_1 Decrement ≥ 15%: 80% Logistic/20% Linear



FEV_1 Decrement ≥ 15%: 50% Logistic/50% Linear



*Derived from Folinsbee et al., 1988; Horstman et al. 1990; McDonnell et al., 1991; Adams 2002, 2003, 2006.

Figure 5-4. Median Exposure-Response Functions Using Three Different Combinations of Logistic and Linear (Hockeystick) Models

Figure 5-4a. FEV_1 Decrements $\geq$ 10%

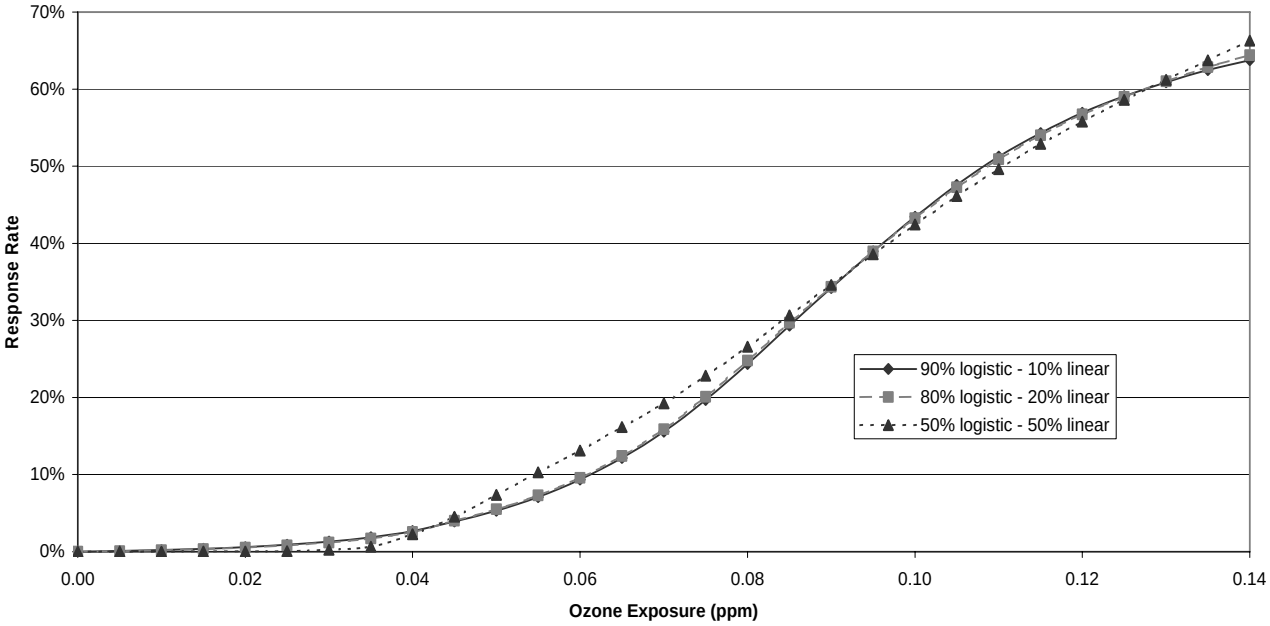
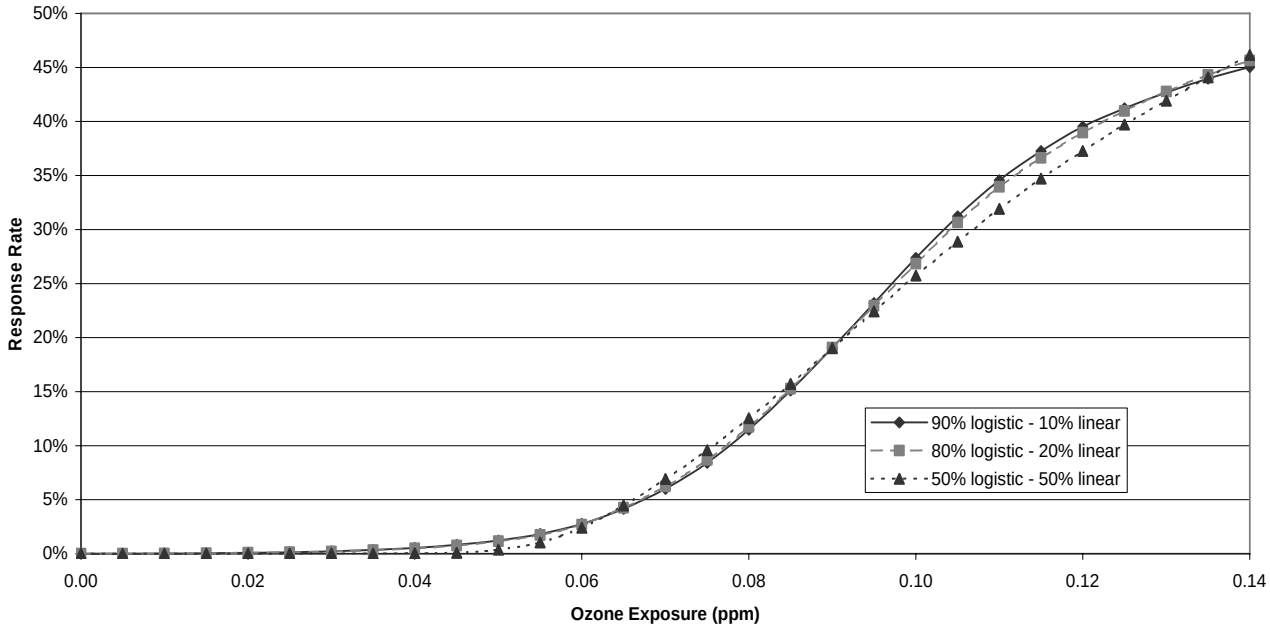


Figure 5-4b. FEV_1 Decrements $\geq$ 15%



or below 0.08 ppm. Additional uncertainties with respect to the estimated exposure-response relationships are briefly summarized below.[12] These additional uncertainties include:

- Length of exposure. The 8-hr moderate exertion risk estimates are based on a combined data set from six controlled human exposure studies conducted using 6.6-hr exposures. The use of these data to estimate responses associated with an 8-hr exposure are reasonable, in our judgment, because lung function response appears to level off after exposure for 4 to 6 hours. It is unlikely that the exposure-response relationships would have been appreciably different had the studies been conducted over an 8-hr period.
- Extrapolation of exposure-response relationships. It was necessary to estimate responses at O_3 levels below the lowest exposure levels used in the controlled human studies (i.e., 0.04 ppm) down to background levels.
- Reproducibility of O_3-induced responses. The risk assessment assumed that the O_3-induced responses for individuals are reproducible. This assumption is supported by the evaluation in the CD (see section AX6.4) which cites studies by McDonnell et al. (1985b) and Hazucha et al. (2003) as showing significant reproducibility of response.
- Age and lung function response. As in the prior review, exposure-response relationships based on controlled human exposure studies involving 18-35 year old subjects were used in the risk assessment to estimate responses for school age children (ages 5-18). This approach is supported by evaluation in the CD (see section AX6.4) which cites the findings of McDonnell et al. (1985a) who reported that children 8-11 years old experienced FEV_1 responses similar to those observed in adults 18-35 years old when both groups were exposed to concentrations of 0.12 ppm at an EVR of 35 L/min/m^2. In addition, a number of summer camp studies of school age children exposed in outdoor environments in the northeast also showed O_3-induced lung function changes similar to and in some cases somewhat larger than those observed in controlled human exposure studies.
- Exposure history. The risk assessment assumed that the O_3-induced response on any given day is independent of previous O_3 exposures. As discussed in Chapter 3 and in the CD, O_3-induced responses can be enhanced on the second day of exposure or attenuated after more than 2 consecutive days of exposure. The possible impact of recent exposure history on the risk estimates is an additional source of uncertainty that is not quantified in this assessment. We note that the three Adams studies which were conducted in Davis, California reported a smaller fraction of the subjects experiencing FEV_1 decrements $\geq$15 and 20% associated with O_3 exposures to 0.08 ppm for 6.6 hours than the Folinsbee/Horstman/McDonnell studies conducted in Chapel Hill, NC at this same level and exposure period. While Adams indicates in each of these studies that O_3 levels did not exceed the 0.09 ppm, 1-hr California standard, we do not know whether the exposures outside the chamber played any role in the differences observed between these

[12]Additional uncertainties with respect to the exposure inputs to the risk assessment are described in Chapter 4 of this Staff Paper, in the Exposure Assessment TSD, and in Langstaff (2007).

two sets of studies or whether the differences might reflect differential sensitivity among the pools of subjects tested.

- Exposure-response relationships for all and asthmatic school age children. The risk assessment used the same O_3 exposure-response relationships developed from data on "healthy" subjects, for all and asthmatic school age children. Based on evidence from epidemiological studies, it is likely that moderate to severe asthmatic children would experience greater lung function decrements than other children without these conditions. This would tend to lead to underestimating the lung function decrements for asthmatic children in the current risk assessment. One consideration working in the opposite direction is that the activity patterns used in the exposure analysis to estimate exposures for asthmatic children were not specific to asthmatic individuals. To the extent that asthmatic children, especially those with moderate to severe asthma, are less active or spend less time outdoors than other children of the same age, the estimates of their 8-hr exposures to O_3 under moderate exertion may be overstated. This factor would tend to lead to overestimates of risks for lung function decrements in the asthmatic school age population.
- Interaction between O_3 and other pollutants. Because the controlled human exposure studies used in the risk assessment involved only O_3 exposures, it was assumed that estimates of O_3-induced health responses would not be affected by the presence of other pollutants (e.g., SO_2, $PM_{2.5}$). Some evidence exists that other pollutants may enhance the respiratory effects associated with exposure to O_3, but the evidence is not consistent across studies.

Variability refers to the heterogeneity in a population or variable of interest that is inherent and cannot be reduced through further research. The current controlled human exposure studies portion of the risk assessment incorporates some of the variability in key inputs to the analysis by using location-specific inputs for the exposure analysis (e.g., location-specific population data, air exchange rates, air quality and temperature data). Although spatial variability in these key inputs across all U.S. locations has not been fully characterized, variability across the selected locations is embedded in the analysis by using, to the extent possible, inputs specific to each urban area. The extent to which there is variability in the exposure-response relationships for the populations included in the risk assessment across different geographic areas is currently unknown. Temporal variability also is more difficult to address, because the risk assessment focuses on some unspecified time in the future. To minimize the degree to which values of inputs to the analysis may be different from the values of those inputs at that unspecified time, we have used the most current inputs available – for example, year 2002, 2003, and 2004 air quality data for the urban locations, and the most recent available population data (from the 2000 Census). However, future changes in inputs have not been predicted (e.g., future population levels).

5.3.2 Assessment of Risk Based on Epidemiological Studies

As discussed above, the current quantitative risk assessment based on epidemiological studies includes risk estimates for respiratory symptoms in moderate to severe asthmatic children, respiratory-related hospital admissions, and total non-accidental and cardiorespiratory mortality associated with short-term O_3 exposures in selected urban locations in the U.S. We want to emphasize that there is considerable evidence that O_3 exposures also results in additional respiratory-related effects beyond those included in the quantitative risk assessment. These effects include, e.g., increased school absences, increased asthma-related emergency department visits, and increased medication usage in asthmatics. These additional effects are discussed in Chapter 3 and considered in the overall risk characterization presented in Chapter 6. The methods used in the epidemiological portion of the quantitative risk assessment are described below.

5.3.2.1 General Approach

The general approach used in this part of the risk assessment relies upon concentration-response functions which have been estimated in epidemiological studies evaluated in the CD. Since these studies estimate concentration-response functions using ambient air quality data from fixed-site, population-oriented monitors, the appropriate application of these functions in a risk assessment similarly requires the use of ambient air quality data at fixed-site, population-oriented monitors. In order to estimate the incidence of a particular health effect associated with recent conditions in a specific county or set of counties attributable to ambient O_3 exposures in excess of background, as well as the change in incidence of the health effect in that county or set of counties corresponding to a given change in O_3 levels resulting from just meeting the current or alternative 8-hr O_3 standards, the following three elements are required:

- **Air quality information** including: (1) recent air quality data for O_3 from population-oriented monitors in the assessment location, (2) estimates of background O_3 concentrations appropriate to this location, and (3) recent concentrations adjusted to reflect patterns of air quality estimated to occur when the area just meets the specified standards. (These air quality inputs are discussed in more detail in section 4.5.6)
- **Concentration-response function(s)** which provide an estimate of the relationship between the health endpoint of interest and ambient O_3 concentrations, preferably derived in the assessment location, as use of functions estimated in other locations increases uncertainty.
- **Seasonal baseline health effects incidence rate and population**. The baseline incidence rate provides an estimate of the incidence rate in the assessment location corresponding to recent O_3 levels in that location.

Figure 5-5 provides a broad schematic depicting the role of these components in this part of the risk assessment. Each of the key components (i.e., air quality information, estimated concentration-response functions, and baseline incidence and population data) is discussed below, highlighting those points at which judgments have been made.

These inputs are combined to estimate health effect incidence changes associated with specified changes in O_3 levels. Although some epidemiological studies have estimated linear or logistic concentration-response functions, by far the most common form is the exponential (or log-linear) form:

$$y = Be^{\beta x}, \qquad \text{(Equation 5-4)}$$

where x is the ambient O_3 level, y is the incidence of the health endpoint of interest at O_3 level x, β is the coefficient of ambient O_3 concentration (describing the extent of change in y with a unit change in x), and B is the incidence at x=0, i.e., when there is no ambient O_3. The relationship between a specified ambient O_3 level, x_0, for example, and the incidence of a given health endpoint associated with that level (denoted as y_0) is then

$$y_0 = Be^{\beta x_0}. \qquad \text{(Equation 5-5)}$$

Because the log-linear form of concentration-response function (equation (5-4)) is by far the most common form, we use this form to illustrate the derivation of the "health impact function" used in this portion of the risk assessment.[13]

If we let x_0 denote the baseline (upper) O_3 level, and x_1 denote the lower O_3 level, and y_0 and y_1 denote the corresponding incidences of the health effect, we can derive the following relationship between the change in x, $\Delta x = (x_0 - x_1)$, and the corresponding change in y, Δy, from equation (5-4)[14]:

$$\Delta y = (y_0 - y_1) = y_0[1 - e^{-\beta \Delta x}]. \qquad \text{(Equation 5-6)}$$

Alternatively, the difference in health effects incidence can be calculated indirectly using relative risk. Relative risk (RR) is a measure commonly used by epidemiologists to characterize

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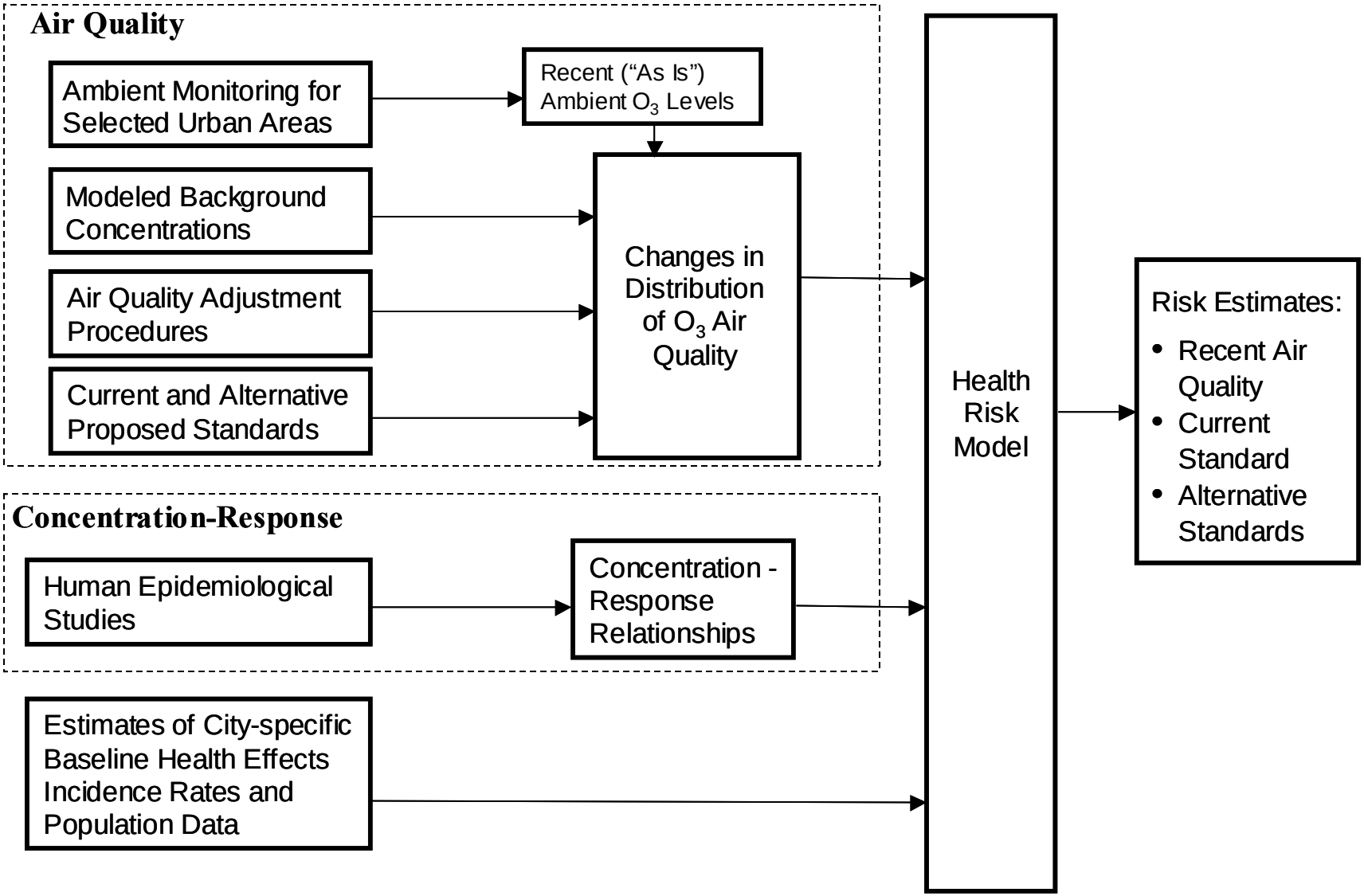
[13]The derivations of the health impact functions from concentration-response functions for all three functional forms found in the epidemiological literature and used in this risk assessment – the log-linear, linear, and logistic – are given in section B.2 of Appendix B in the Risk Assessment TSD.

[14]If $\Delta x < 0$ – i.e., if $\Delta x = (x_1 - x_0)$ – then the relationship between Δx and Δy can be shown to be $\Delta y = (y_1 - y_0) = y_0[e^{\beta \Delta x} - 1]$. If $\Delta x < 0$, Δy will similarly be negative. However, the *magnitude* of Δy will be the same whether $\Delta x > 0$ or $\Delta x < 0$ – i.e., the absolute value of Δy does not depend on which equation is used.

Figure 5-5. Major Components of Ozone Health Risk Assessment Based on Epidemiological Studies



the comparative health effects associated with a particular air quality comparison. The risk of mortality at ambient O_3 level x_0 relative to the risk of mortality at ambient O_3 level x_1, for example, may be characterized by the ratio of the two mortality rates: the mortality rate among individuals when the ambient O_3 level is x_0 and the mortality rate among (otherwise identical) individuals when the ambient O_3 level is x_1. This is the RR for mortality associated with the difference between the two ambient O_3 levels, x_0 and x_1. Given a concentration-response function of the form shown in equation (5-4) and a particular difference in ambient O_3 levels, Δx, the RR associated with that difference in ambient O_3, denoted as $RR_{\Delta x}$, is equal to $e^{\beta \Delta x}$. The difference in health effects incidence, Δy, corresponding to a given difference in ambient O_3 levels, Δx, can then be calculated based on this $RR_{\Delta x}$:

$$\Delta y = (y_0 - y_1) = y_0[1 - (1/RR_{\Delta x})] \quad \text{(Equation 5-7)}$$

Equations (5-6) and (5-7) are simply alternative ways of expressing the relationship between a given difference in ambient O_3 levels, Δx, and the corresponding difference in health effects incidence, Δy. These health impact equations are the key equations that combine air quality information, concentration-response function information, and baseline health effects incidence information to estimate ambient O_3 health risk.

5.3.2.2 Air Quality Considerations

As illustrated in Figure 5-5, and noted earlier, air quality information required to conduct the O_3 risk assessment includes: (1) recent air quality data for O_3 from suitable monitors for each selected location, (2) estimates of background concentrations for each selected location, and (3) air quality adjustment procedures to modify the recent data to reflect changes in the distribution of hourly O_3 air quality estimated to occur when an area just meets a given O_3 standard. The approach used to adjust air quality data to simulate just meeting alternative 8-hr standards is discussed in more detail in Chapter 4 and in Rizzo (2005, 2006).

We retrieved O_3 ambient air quality data for the years 2002 through 2004 from EPA's Air Quality System (AQS). Although the O_3 season varies somewhat for different regions of the country, for much of the country the season coincides with spring and summer. To allow comparison across locations, and because O_3 effects observed in epidemiological studies have been more clearly and consistently shown for warm season analyses, all analyses for this portion of the risk assessment were carried out for the same time period, April through September. Because O_3 concentrations varied substantially over the 3-year period from 2002 through 2004, separate analyses were carried out using air quality data from 2002, in which O_3 concentrations were relatively higher in most locations over this 3-year period, and air quality data from 2004,

in which O_3 concentrations were relatively lower in most locations for this 3-year period. These two years provide generally upper- and lower-end cases within this 3-year period. However, two of the 12 urban areas, Houston and Los Angeles, had similar or higher O_3 concentrations in 2004 compared to 2002. In addition, a more limited set of analyses, focusing only on mortality in a subset of five urban areas (Atlanta, Chicago, Houston, Los Angeles, and New York), was carried out since the second draft Staff Paper, using air quality data from 2003.

To estimate the change in incidence of a health effect associated with a change in O_3 concentrations from recent levels to background levels in an assessment location, two time series of O_3 concentrations are needed for that location: (1) hourly O_3 concentrations from a recent year for the period April 1 through September 30, and (2) hourly background O_3 concentrations for the same time period. In order to be consistent with the approach generally used in the epidemiological studies that estimated O_3 concentration-response functions, the (spatial) average ambient O_3 concentration on each hour for which measured data are available is deemed most appropriate for the risk assessment. A composite monitor data set was created for each assessment location based on averaging each hourly value from all monitors eligible for comparison with the current standard for each hour of the day. Table 4-7 provides a summary of the design values for the 12 urban study areas. Appendix 5A.1 to this chapter provides more detailed information on ambient O_3 concentrations for these locations.

Different exposure metrics have been used in epidemiological O_3 studies, including the 24-hr average and the daily 1-hr and 8-hr maximum. Therefore, daily changes at the composite monitor in the O_3 exposure metric appropriate to a given concentration-response function were calculated for use in the risk assessment (see Tables 5A-13 and 5A-14, Appendix 5A.1 for summary statistics for the composite monitor O_3 concentrations in the 12 urban locations for 2002, 2003, and 2004). For example, if a concentration-response function related daily mortality to daily 1-hr maximum O_3 concentrations, the daily changes in 1-hr maximum O_3 concentrations at the composite monitor were calculated. In the first part of the epidemiology-based risk assessment, in which risks associated with the recent levels of O_3 above background levels were estimated, this required the following steps:

- Using the monitor-specific input streams of hourly O_3 concentrations from a recent year, calculate a stream of hourly O_3 concentrations at the composite monitor. The recent O_3 concentration at the composite monitor for a given hour on a given day is the average of the monitor-specific O_3 concentrations for that hour on that day.
- Using this stream of hourly O_3 concentrations from a recent year at the composite monitor, calculate the 1-hr maximum O_3 concentration for each day at the composite monitor.
- Using the monitor-specific input streams of hourly background O_3 concentrations, calculate a stream of hourly background O_3 concentrations at the composite monitor.

- Using this stream of background hourly O_3 concentrations at the composite monitor, calculate the 1-hr maximum background O_3 concentration for each day at the composite monitor.
- For each day, calculate Δx = (the 1-hr maximum O_3 concentration for that day at the composite monitor) - (the 1-hr maximum background O_3 concentration for that day at the composite monitor).

The calculations for the second part of the epidemiology-based risk assessment, in which risks associated with estimated O_3 levels that just meet the current and potential alternative 8-hr standards above background levels, were done analogously. For this case the series of monitor-specific adjusted hourly concentrations were used rather than the series of monitor-specific recent monitored hourly concentrations. Similarly, calculations for concentration-response functions that used a different exposure metric (e.g., the 8-hr daily maximum or 24-hr average) were done analogously, using the exposure metric appropriate to the concentration-response function.

5.3.2.3 Concentration-Response Functions

As indicated in Figure 5-5, another key component in the risk model based on epidemiological studies is the set of concentration-response functions which provide estimates of the relationships between each health endpoint of interest and ambient concentrations. As discussed above, the health endpoints that have been included in the O_3 risk assessment include respiratory symptoms in moderate-to-severe asthmatic children, respiratory-related hospital admissions, and premature mortality associated with short-term exposures. For those health endpoints, the assessment includes all estimates of response magnitude from studies judged suitable for inclusion in this assessment, including those that did not yield statistically significant results. Effect estimates that are not statistically significant are used from studies judged suitable for inclusion in this assessment to avoid introducing bias into the estimate of the magnitude of the effect. Table 5-4 summarizes the studies included in this part of the risk assessment for each of the urban locations.

Studies often report more than one estimated concentration-response function for the same location and health endpoint. Sometimes models including different sets of co-pollutants are estimated in a study; sometimes different lags are estimated. In some cases, two or more studies estimated a concentration-response function for O_3 and the same health endpoint in the same location (this is the case, for example, with O_3 and mortality associated with short-term exposures). For some health endpoints, there are studies that estimated multi-city and single-city O_3 concentration-response functions, while other studies estimated only single-city functions.

All else being equal, a concentration-response function estimated in the assessment location is preferable to a function estimated elsewhere, since it avoids uncertainties related to potential differences due to geographic location. That is why the urban areas selected this part of the O_3 risk assessment are, generally, those locations in which concentration-response functions have been estimated. There are several advantages, however, to using estimates from multi-city studies versus studies carried out in single cities. These advantages include, but are not limited to: (1) more precise effect estimates due to larger data sets, reducing the uncertainty around the estimated coefficient, (2) greater consistency in data handling and model specification that can eliminate city- to-city variation due to study design, and (3) less likelihood of publication bias or exclusion of reporting of negative or nonsignificant findings. Multi-city studies are applicable to a variety of settings, since they estimate a central tendency across multiple locations. Because single-city and multi-city studies have different advantages, where both are available for a given location, risk estimates have been developed for both functions.

As discussed in the CD and section 3.3.2.1 of this Staff Paper, O_3 epidemiological studies have reported relationships based on single pollutant models and/or multi-pollutant models (i.e., where PM, NO_2, SO_2, or CO were entered into the health effects model along with O_3. To the extent that any of the co-pollutants present in the ambient air may have contributed to the health effects attributed to O_3 in single pollutant models, risks attributed to O_3 might be overestimated where concentration-response functions are based on single pollutant models. However, if co-pollutants are highly correlated with O_3, their inclusion in an O_3 health effects model can lead to misleading conclusions in identifying a specific causal pollutant. When collinearity exists, inclusion of multiple pollutants in models often produces unstable and statistically insignificant effect estimates for both O_3 and the co-pollutants. Given that single and multi-pollutant models each have both potential advantages and disadvantages, with neither type clearly preferable over the other in all cases, risk estimates based on both single- and multi-pollutant models have been included in the risk assessment where both are available.

Epidemiological studies have reported effect estimates associated with varying lag periods, but for the reasons discussed in the CD and summarized in section 3.4.5 above the CD focuses on effect estimates from models using 0- or 1-day lag periods, with some consideration of multi-day lag effects (CD, p. 7-11). For quantitative assessments, we conclude that it is appropriate to use results from lag period analyses consistent with those reported in the CD, focusing on single day lag periods of 0-1 days for associations with mortality or respiratory hospitalization, depending on availability of results (CD, p. 8-59). If the effect of O_3 on health outcomes persists over several days, distributed lag model results can provide more accurate effect estimates for quantitative assessment than an effect estimate for a single lag period (CD, p. 7-10). Therefore, we have used distributed lag models when they are available. Where only

Table 5-4. Locations, Health Endpoints, and Epidemiological Studies Included in the O_3 Risk Assessment*

Urban Area	Premature Mortality	Hospital Admissions for Respiratory Illnesses	Respiratory Symptoms in Asthmatic Children
Atlanta	Bell et al. (2004) Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities**		
Boston	Bell et al. (2004) – 95 cities		Gent et al. (2003)
Chicago	Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities** Schwartz (2004) Schwartz (2004) – 14 cities		
Cleveland	Bell et al. (2004) Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities**	Schwartz et al. (1996)	
Detroit	Bell et al. (2004) Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities** Schwartz (2004) Schwartz (2004) – 14 cities Ito (2003)	Ito (2003)	
Houston	Bell et al. (2004) Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities** Schwartz (2004) Schwartz (2004) – 14 cities		
Los Angeles	Bell et al. (2004) Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities**	Linn et al. (2000)	
New York	Bell et al. (2004) – 95 cities Huang et al. (2004)** Huang et al. (2004) – 19 cities**	Thurston et al. (1992)	
Philadelphia	Bell et al. (2004) – 95 cities Huang et al. (2004) ** Huang et al. (2004) – 19 cities** Moolgavkar et al. (1995)		
Sacramento	Bell et al. (2004) Bell et al. (2004) – 95 cities		
St. Louis	Bell et al. (2004) Bell et al. (2004) – 95 cities		
Washington, D.C.	Bell et al. (2004) – 95 cities		

*Where a study indicates "14 cities," "19 cities," or "95 cities," a multi-city concentration-response function was used in the risk assessment and the assessment location was one of the cities included in the original epidemiological study.

**This study estimated concentration-response functions for cardiorespiratory mortality.

single day lags are available we have focused on single day lag periods of 0-1 days for associations with mortality or respiratory hospitalization, depending on availability of effect estimates (CD, p. 8-59).

In summary:

- if a single-city concentration-response function was estimated in a risk assessment location and a multi-city function which includes that location was also available for the same health endpoint, both functions were included for that location in the risk assessment;
- risk estimates based on both single- and multi-pollutant models were used when both were available;
- distributed lag models were used, when available; when a study reported several single lag models for a health effect, the initial selection of the appropriate lag structure for the health effect was based on the overall assessment in the CD, considering all studies reporting concentration-response functions for that health effect.

The locations, health endpoints, studies, and concentration-response functions included in that portion of the risk assessment based on epidemiological studies are summarized in Tables 5B-1 through 5B-12 in Appendix 5B.1.

5.3.2.4 Baseline Health Effects Incidence and Population Estimates

As illustrated in Equation 5-4, the most common health risk model based on epidemiological studies expresses the reduction in health risk (Δy) associated with a given reduction in O_3 concentrations (Δx) as a percentage of the baseline incidence (y). To accurately assess the impact of changes in O_3 air quality on health risk in the selected urban areas, information on the baseline incidence of health effects in each location is therefore needed. For this assessment, baseline incidence is the incidence under recent air quality conditions. Population sizes, for both total population and various age ranges used in the risk assessment were obtained for the year 2000 (U.S. Census) and are summarized in Table 5-5. Where possible, county-specific incidence or incidence rates have been used in the assessment. County specific mortality incidences were available for the year 2002 from CDC Wonder (CDC, 2005), an interface for public health data dissemination provided by the Centers for Disease Control (CDC). The baseline mortality rates for each risk assessment location are provided in Table 5-6 and are expressed as a rate per 100,000 population.

County-specific rates for respiratory hospital discharges, and various subcategories (e.g., asthma, pneumonia) have been obtained, where possible, from state, local, and regional health

Table 5-5. Relevant Population Sizes for O_3 Risk Assessment Locations*

City	Counties	Population (in millions)*			
		Total	Ages ≥30	Ages ≥ 65	Children, Ages ≤ 12, with asthma**
Atlanta	Fulton, DeKalb	1.5	---	---	---
Boston	Suffolk	0.7	---	---	
Boston	Essex, Middlesex, Norfolk, Suffolk, Worcester	---	---	---	0.025
Chicago	Cook	5.4	---	---	---
Cleveland	Cuyahoga	1.4	---	0.2	
Detroit	Wayne	2.1	---	---	---
Houston	Harris	3.4	---	---	---
Los Angeles	Los Angeles	9.5	---	---	---
Los Angeles	Los Angeles, Riverside, San Bernardino, Orange	---	8.4	---	---
New York	Bronx, Kings, Queens, New York, Richmond, Westchester	8.9	---	---	---
New York	Bronx, Kings, Queens, New York, Richmond	8.0	---	---	---
Philadelphia	Philadelphia	1.5	---	---	---
Sacramento	Sacramento	1.2	---	---	---
St. Louis	St. Louis City	0.3	---	---	---
Washington, D.C.	Washington, D.C.	0.6	---	---	---

* Total population and age-specific population estimates taken from the 2000 U.S. Census. Populations are rounded to the nearest 0.1 million. The urban areas given in this table are those considered in the studies used in the risk assessment, with the exception of the larger Boston area, which is the CSA for Boston (since the study that estimated a concentration-response function in moderate and severe asthmatic children (ages 0 – 12) was conducted in Springfield, MA and CT).
** Population derived as follows: The populations of children <5 and 5 - 12 in the counties listed were multiplied by corresponding percents of children [in each age group] in New England with "current asthma" -- 5.1% and 10.7% for the two age groups, respectively (see "The Burden of Asthma in New England." Asthma Regional Council. March 2006. Table S-2. www.asthmaregionalcouncil.org). These estimated numbers of asthmatic children were then multiplied by the estimated percent of asthmatic children using maintenance medications (40%) (obtained via email 4-05-06 from Jeanne Moorman, CDC) and the results were summed.

Table 5-6. Baseline Mortality Rates (per 100,000 Population) Used in the O_3 Risk Assessment*

City	Counties	Type of Mortality (ICD-9 Codes)		
		Non-accidental (<800)	Cardiorespiratory (390-448; 490-496; 487; 480-486; 507)	Respiratory (460-519)
Atlanta	Fulton, DeKalb	623	131	---
Boston	Suffolk	736	---	---
Chicago	Cook	781	189	---
Cleveland	Cuyahoga	1,058	268	---
Detroit	Wayne	913	234	76
Houston	Harris	533	123	---
Los Angeles	Los Angeles	569	155	---
New York	Bronx, Kings, Queens, New York, Richmond, Westchester	704	199	---
Philadelphia	Philadelphia	1,057	242	---
Sacramento	Sacramento	686	---	---
St. Louis	St. Louis City	1147	---	---
Washington, D.C.	Washington, D.C.	942	---	---
National	---	790	196	80

*Data for the year 2002 from United States Department of Health and Human Services (US DHHS), Centers for Disease Control and Prevention (CDC), National Center for Health Statistics (NCHS), Compressed Mortality File (CMF) compiled from CMF 1968-1988, Series 20, No. 2A 2000, CMF 1989-1998, Series 20, No. 2E 2003 and CMF 1999-2002, Series 20, No. 2H 2004 on CDC WONDER On-line Database. See http://wonder.cdc.gov/.

departments and hospital planning commissions for each of the risk assessment locations.[15] Baseline hospitalization rates used in each risk assessment location are summarized in Table 5-7 and are expressed as a rate per 100,000 relevant population.

Baseline rates of symptoms among asthmatic children who used maintenance medications in the Boston area were estimated by using the median rates of the respiratory symptoms reported in Table 3 of Gent et al. (2003). Each symptom rate, the percentage of days on which the symptom occurred, was calculated for each subject by dividing the number of days of the symptom by the number of days of participation in the study and then multiplying by 100. Median symptom rates among maintenance medication users for wheeze, chest tightness, and shortness of breath were 2.8%, 1.2%, and 1.5% of days, respectively.

5.3.2.5 Characterizing Uncertainty and Variability

Section 5.3.1.4 previously defined what is meant by *uncertainty* and *variability* in the context of this risk assessment. For the portion of the risk assessment based on epidemiological studies, the statistical uncertainty surrounding the estimated O_3 coefficients in the reported concentration-response functions is reflected in the confidence or credible intervals provided for the risk estimates in this chapter and in the Risk Assessment TSD. Additional uncertainties have been addressed quantitatively through sensitivity analyses and/or have been discussed throughout section 5.3.

With respect to variability within this portion of the risk assessment, there may be variability among concentration-response functions describing the relation between O_3 and mortality across urban areas. This variability may be due to differences in population (e.g., age distribution), population activities that affect exposure to O_3 (e.g., use of air conditioning), levels and composition of co-pollutants, and/or other factors that vary across urban areas.

The current risk assessment incorporates some of the variability in key inputs to the analysis by using location-specific inputs (e.g., location-specific concentration-response functions, baseline incidence rates, and air quality data). Although spatial variability in these key inputs across all U.S. locations has not been fully characterized, variability across the selected locations is imbedded in the analysis by using, to the extent possible, inputs specific to each urban area. Temporal variability is more difficult to address, because the risk assessment

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[15]The data were annual hospital discharge data, which were used as a proxy for hospital admissions. Hospital discharges are issued to all people who are admitted to the hospital, including those who die in the hospital. Use of the annual or seasonal discharge rate is based on the assumption that the admissions at the end of the year (or season) that carry over to the beginning of the next year (or season), and are therefore not included in the discharge data, are offset by the admissions in the previous year (or season) that carry over to the beginning of the current year (or season).

Table 5-7. Baseline Rates for Hospital Admissions Used in the O_3 Risk Assessment

Relevant Population	Rate per 100,000 Relevant Population*			
	Los Angeles[1]	New York[2]	Detroit[3]	Cleveland[4]
	Ages 30+	All Ages	Ages 65+	Ages 65+
Admissions for:				
Pulmonary illness (DRG Codes 75 – 101) – spring	208	---	---	---
Pulmonary illness (DRG Codes 75 – 101) – summer	174	---	---	---
Respiratory illness (ICD codes 466, 480-486, 490, 491, 492, 493)	---	800	---	---
Asthma (ICD code 493)	---	327	---	---
Pneumonia (ICD codes 480-486)	---	---	2,068	---
Respiratory illness ((ICD codes 460-519)	---	---	---	3,632

[1] Rates of unscheduled hospital admissions were calculated from patient discharge data for 1999, obtained from California's Office of Statewide Health Planning and Development, which also provided records of hospital admissions for the study by Linn et al. (2000).
[2]Rates of unscheduled hospital admissions were calculated from patient discharge data for 2001, obtained from the New York Statewide Planning and Research Cooperative.
[3]Rates were calculated from hospitalization data for Wayne County for the year 2000, obtained from the Michigan Health and Hospital Association in April 2002. EPA expressly understands that the Michigan Health and Hospital Association has not performed an analysis of the hospitalization data obtained or warranted the accuracy of this information and, therefore, it cannot be held responsible in any manner for the outcome.
[4]Based on mean daily hospital admissions for ages 65+ for ICD-9 codes 460-519 -- Table 1 in Schwartz et al. (1996).

focuses on some unspecified time in the future. To minimize the degree to which values of inputs to the analysis may be different from the values of those inputs at that unspecified time, we have used recent input data – for example, years 2002 through 2004 air quality data for all of the urban locations, and recent mortality baseline incidence rates (from 2002). However, future changes in inputs have not been predicted (e.g., future population levels or possible changes in baseline incidence rates).

A number of important sources of uncertainty were addressed where possible. Section 4.1.9 in the Risk Assessment TSD discusses in greater detail the uncertainties and variability present in the health risk assessment. The following is a brief discussion of the major sources of uncertainty and variability in the epidemiological portion of the risk assessment and how they are dealt with or considered in the risk assessment:

- Causality. There is uncertainty about whether each of the estimated associations between O_3 indicators and the various health endpoints included in this risk assessment actually reflect a causal relationship. Our judgment, drawing on the conclusions in the CD and as discussed in more detail in Chapter 3 (section 3.7.5), is that for the health effects included in the risk assessment (i.e, increased respiratory symptoms in moderate to severe asthmatic children, increased respiratory-related hospital admissions, total non-accidental mortality, and cardiorespiratory mortality) there is, at a minimum, a likely causal relationship with either short-term O_3 exposure itself or with O_3 serving as an indicator for itself and other components of the photochemical oxidant mix, especially during the warm O_3 season.
- Empirically estimated concentration-response relationships. In estimating the concentration-response relationships, there are uncertainties: (1) surrounding estimates of O_3 coefficients in concentration-response functions used in the assessment, (2) concerning the specification of the concentration-response model (including the shape of the relationship) and whether or not a population threshold or non-linear relationship exists within the range of concentrations examined in the studies , (3) related to the extent to which concentration-response relationships derived from studies in a given location and time when O_3 levels were higher or behavior and/or housing conditions were different provide accurate representations of the relationships for the same locations with lower air quality distributions and different behavior and/or housing conditions, and (4) concerning the possible role of co-pollutants which also may have varied between the time of the studies and the current assessment period. The approach taken to characterize uncertainties in the concentration-response functions arising from sample size considerations is discussed below. With respect to the shape of the function and whether or not a population threshold may exist, as discussed in Chapter 3, the CD states that in those studies that provide suggestive evidence of thresholds, the potential thresholds are at low concentrations at or approaching background levels (CD, p. 7-159). As discussed in Chapter 3 and in the CD (CD, p.7-175), results from recent large U.S. multi-city time-series studies and meta-analyses also show effect estimates that are consistent across studies and robust to control for potential confounders.

- Adequacy of ambient O_3 monitors as surrogate for population exposure. The extent to which there are differences in the relationship between spatial variation in ambient O_3 concentrations and ambient exposures in the original epidemiology studies compared to more recent ambient O_3 data introduces additional uncertainty in the risk estimates. We recognize that ambient concentrations at central monitors may not provide a good representation of personal exposures. The CD identifies the following three components to exposure measurement error: (1) the use of average population rather than individual exposure data; (2) the difference between average personal ambient exposure and ambient concentrations at central monitoring sites; and (3) the difference between true and measured ambient concentrations (CD, p. 7-7). The CD notes that "these components are expected to have different effects, with the first and third likely not causing bias in a particular direction ("nondifferential error") but increasing the standard error, while the second component may result in downward bias, or attenuation of the risk estimate" (CD, pp. 7-7 to 7-8). While a concentration-response function may understate the effect of personal exposures to O_3 on the incidence of a health effect, it will give an unbiased estimate of the effect of ambient concentrations on the incidence of the health effect if the ambient concentrations at monitoring stations provide an unbiased estimate of the ambient concentrations to which the population is exposed. A more comprehensive discussion of exposure measurement is given in section 3.4.2.1 of this Staff Paper.
- Adjustment of air quality distributions to simulate just meeting the current standard. The shape of the distribution of hourly O_3 concentrations that would result upon just meeting the current or alternative 8-hr standards is unknown. Based on an analysis of historical data, we believe that the Quadratic air quality adjustment procedure provides reasonable estimates of the shape of the distribution; however, there is greater uncertainty for those urban areas that have air quality well above the current standard (e.g., Los Angeles, Houston). As noted previously, there is considerable year-to-year variability in O_3 concentrations over a three-year period in many of the urban areas examined. This leads to substantial year-to-year variability in risk estimates associated with O_3 concentrations when air quality is simulated to just meet the current and potential alternative standards.
- Estimated background concentrations for each location. The calculation of risk associated with recent air quality in excess of background requires as an input estimates of background concentrations for each location throughout the period of the assessment. The estimated background concentrations for each location have been estimated based on runs of the GEOS-CHEM global model (see section 2.7) for all hours of an "average day" in a given month, for each of the months from April through September. As discussed in section 2.7, evaluation of the GEOS-CHEM suggests that the model is generally within 5 ppb in most regions of the country and that it may be 10 ppb too high in the southeast. Section 5.4.3 presents results from a sensitivity analysis that characterizes the impact of the uncertainty about background concentrations on the non-accidental mortality risk estimates associated with recent air quality and just meeting the current and several alternative 8-hr standards.
- Baseline incidence rates and population data. There are uncertainties related to: (1) the extent to which baseline incidence rates, age distributions, and other relevant demographic variables that impact the risk estimates vary for the year(s) when the actual

epidemiological studies were conducted, the recent year of air quality used in this assessment, and some unspecified future year when air quality is adjusted to simulate just meeting the current or alternative standards and (2) the use of annual or seasonal incidence rate data to develop daily health effects incidence data. Spatial variability in baseline incidence and population data is taken into account by use of city-specific data in most cases.

One of the most critical elements in the risk assessment is the concentration-response relationships used in the assessment. The uncertainty resulting from the statistical uncertainty associated with the estimate of the O_3 coefficient in the concentration-response function was characterized either by confidence intervals or by Bayesian credible intervals around the corresponding point estimates of risk. Confidence and credible intervals express the range within which the true risk is likely to fall if the only uncertainty surrounding the O_3 coefficient involved sampling error. Other uncertainties, such as differences in study location, time period, and model uncertainties are not represented by the confidence or credible intervals presented.

Two large scale multi-city mortality studies, Bell et al. (2004) and Huang et al. (2004), reported both multi-location and single-location concentration-response functions, using a Bayesian two-stage hierarchical model. In these cases, the single-location estimates can be adjusted to make more efficient use of the data from all locations. The resulting "shrinkage" estimates are so called because they "shrink" the location-specific estimates towards the overall mean estimate (the mean of the posterior distribution of the multi-location concentration-response function coefficient). The greater the uncertainty about the estimate of the location-specific coefficient relative to the estimate of between-study heterogeneity, the more the location-specific estimate is "pulled in" towards the overall mean estimate. Bell et al. (2004) calculated these shrinkage estimates, which were presented in Figure 2 of that paper. These location-specific shrinkage estimates, and their adjusted standard errors were provided by the study authors and were used in the risk assessment.

The location-specific estimates reported in Table 1 of Huang et al. (2004) are not "shrinkage" estimates. However, the study authors provided the posterior distribution for the heterogeneity parameter, τ, for their distributed lag model, shown in Figure 4(b) of their paper. Given this posterior distribution, and the original location-specific estimates presented in Table 1 of their paper, we calculated location-specific "shrinkage" estimates using a Bayesian method described in DuMouchel (1994) (see section 5B.3 in Appendix 5B of this Staff Paper). As with the shrinkage estimates presented in Bell et al. (2004), the resulting Bayesian shrinkage estimates use the data from all of the locations considered in the study more efficiently than do the original location-specific estimates. The calculation of these shrinkage estimates is thus one way to address the relatively large uncertainty surrounding estimates of coefficients in location-specific concentration-response functions.

With respect to model form, most of the epidemiological studies estimated O_3 coefficients using log-linear models. However, there still is substantial uncertainty about the correct functional form of the relationship between O_3 and various health endpoints, especially at the low end of the range of observed concentrations. While there are likely biological thresholds in individuals for specific health responses, as discussed in section 3.4.6 available studies have found little evidence for population thresholds. For example, in a recent study, Bell et al. (2006), applied several statistical models to data on air pollution, weather, and mortality for the 98 NMMAPS communities to evaluate whether a threshold level exists for premature mortality. The results suggested that even low levels of tropospheric O_3, well below 0.08 ppm, are associated with premature mortality. However, as discussed in section 3.4.6 and in the CD, the use of ambient O_3 concentrations may obscure the presence of thresholds in epidemiological studies (CD p. 7-158). In those studies that provide suggestive evidence of thresholds, the potential thresholds are at low concentrations at or approaching background levels (CD, p. 7-159).

The CD finds that no definitive conclusion can be reached with regard to the existence of thresholds in epidemiological studies (CD, p. 8-44). We recognize, however, the possibility that thresholds for individuals may exist for reported associations at fairly low levels within the range of air quality observed in the studies, but not be detectable as population thresholds in epidemiological analyses. Based on the CD's conclusions, we judge that there is insufficient evidence to support use of potential threshold levels in the quantitative risk assessment, but we do recognize there is increasing uncertainty about the concentration-response relationship at lower concentrations that is not captured by the characterization of the statistical uncertainty due to sampling error. Therefore, as discussed later in this chapter, the risk estimates for premature mortality, respiratory symptoms in moderate to severe asthmatic children, and respiratory-related hospital admissions associated with exposure to O_3 must be considered in the light of uncertainties about whether or not these O_3-related effects occur in the population at very low concentrations.

Several recent meta-analyses (Bell et al. 2005; Levy et al., 2005; and Ito et al., 2005) have addressed the impact of various factors on estimates of mortality associated with short-term exposures to O_3. We reviewed these meta-analyses for additional information that might be used to assist in characterizing the uncertainties associated with concentration-response functions for this health outcome. As discussed in Chapter 3, the CD observes common findings across all three analyses, in that all reported that effect estimates were larger in warm season analyses, reanalysis of results using default GAM criteria did not change the effect estimates, and there was no strong evidence of confounding by PM (CD, p. 7-97). Bell et al. (2005) and Ito et al. (2005) both provided suggestive evidence of publication bias, but O_3-mortality associations

remained after accounting for that potential bias. The results from these meta-analyses, as well as several single- and multiple-city studies, also indicate that copollutants generally do not appear to substantially confound the association between O_3 and mortality.

As discussed in Chapter 3, while concluding that O_3-health associations are found to be generally consistent, the recent O_3-mortality meta-analyses indicate that some heterogeneity exists across studies (CD, pp. 7-96 – 7-97). The CD discusses a number of factors that could result in heterogeneity in associations between different geographic areas, focusing particularly on variables that can affect exposure to ambient O_3. For example, the use of air conditioning can reduce ambient exposures during the warm season, while increased outdoor activity can increase exposure.

5.4 OZONE RISK ESTIMATES

We present risk estimates associated with several air quality scenarios, including three recent years of air quality as represented by 2002, 2003, and 2004 monitoring data in section 5.4.1. In section 5.4.2 we summarize risk estimates associated with air quality adjusted to simulate just meeting the current and several potential alternative 8-hr standards. In section 5.4.3 we present and discuss the results of sensitivity analyses examining the influence of alternative estimates of background O_3 concentrations and alternative assumptions about the shape of the exposure-response relationship for lung function decrements in all and asthmatic school age children. In section 5.4.4 we discuss and compare the risk estimates developed for the current review with the risk estimates developed for the prior O_3 NAAQS review completed in July 1997. Finally, in section 5.4.5 we present key observations from the health risk assessment.

Throughout this section the uncertainty surrounding risk estimates resulting from the statistical uncertainty of the O_3 coefficients in the concentration- and exposure-response functions used is characterized by ninety-five percent confidence or credible intervals around estimates of incidence, incidence per 100,000 population, and the percent of total incidence that is O_3-related. In some cases, the lower bound of a confidence or credible interval falls below zero. This does not imply that additional exposure to O_3 has a beneficial effect, but only that the estimated O_3 coefficient in the concentration- or exposure-response function was not statistically significantly different from zero. Lack of statistical significance could reflect insufficient statistical power to detect a relationship that exists or that a relationship does not exist. Conversely, the fact that a study reports statistical significant associations does not prove causation. The judgments about whether a causal relationship likely exists between O_3 and various health endpoints rests on a variety of types of supporting evidence and involves a weight of the evidence judgment, as discussed in Chapter 3.

5.4.1 Recent Air Quality

In the prior 1997 risk assessment, risks for lung function decrements associated with 1-hr heavy exertion, 1-hr moderate exertion, and 8-hr moderate exertion exposures were estimated. Since the 8-hr moderate exertion exposure scenario for children clearly resulted in the greatest health risks in terms of lung function decrements, we have chosen to include only the 8-hr moderate exertion exposures in the current risk assessment for this health endpoint. Thus, the risk estimates presented here are most useful for making relative comparisons across alternative air quality scenarios and do not represent the total risks for lung function decrements in children or other groups within the general population associated with any of the air quality scenarios. Thus, some outdoor workers and adults engaged in moderate exertion over multi-hour periods (e.g., 6-8 hr exposures) also would be expected to experience similar lung function decrements. However, the percentage of each of these other subpopulations expected to experience these effects is expected to be smaller than all school age children who tend to spend more hours outdoors while active based on the exposure analyses conducted during the prior review.

We have included risk estimates for all and asthmatic school age children in this section. As noted previously, risk estimates for asthmatic school age children have been added to the assessment since the second draft Staff Paper. Risk estimates associated with recent air quality (2002 and 2004) for up to 12 urban locations are presented in this section. Additional risk estimates developed since the second draft Staff Paper associated with 2003 air quality for a subset of five locations (Atlanta, Chicago, Houston, Los Angeles, and New York) are presented in the Risk Assessment TSD and for all 12 urban areas in a recent memo (Post, 2007).

Tables 5-8 and 5-9 display the risk estimates for all school age children (ages 5-18) associated with 2004 and 2002 O_3 concentrations for $\geq$ 15% lung function decrement responses for the 12 urban areas. Tables 5-8 and 5-9 also include risk estimates associated with air quality adjusted to simulate just meeting the current 0.08 ppm, 8-hr standard, which will be discussed in the next section. Consistent with CASAC's advice contained in its October 24, 2006 letter (Hendeson, 2006), we have focused on FEV_1 decrements $\geq$ 15% for all school age children since this level of response is judged to be an indicator of adverse health effects for healthy children. Similar estimates for $\geq$ 10 and $\geq$ 20% decrement in lung function for all school age children can be found in Chapter 3 of the Risk Assessment TSD. All estimates in both tables reflect responses associated with exposure to O_3 in excess of exposures associated with background O_3 concentrations. Table 5-8 shows the number and percent of all school age children estimated to have at least 1 lung function response (defined as $FEV_1 \geq 15\%$) during the O_3 season. Table 5-9 displays the total number of occurrences for this same lung function response during the O_3 season. As illustrated by the estimates shown in these two tables, a child may experience multiple occurrences of a lung function response during the O_3 season. For example, in Atlanta

the median estimate is that 34,000 school age children experienced an FEV_1 decrement $\geq$ 15% during the O_3 season with a median estimate of about 191,000 occurrences of this same response in this population for 2004 air quality data. Thus, for this example on average each child is estimated to have over 5 occurrences of this lung function response during the O_3 season.

As shown in Table 5-8, across the 12 urban areas, the ranges in median estimates of the percent of all school age children estimated to experience at least one FEV_1 decrement $\geq$ 15% during the O_3 season are 1.4 to 6.0% for 2004 and 5.3 to 9.9% for 2002. In terms of total occurrences of FEV_1 decrement $\geq$ 15% during the O_3 season, Table 5-9 shows a range of median estimates from 69,000 to nearly 1.5 million responses in 2004 and from about 145,000 to over 1.5 million responses in 2002 for all school age children across the 12 urban areas associated with O_3 concentrations. Both Tables 5-8 and 5-9 also include 95% credible intervals for the lung function decrement risk estimates based on sample size considerations. These credible intervals only represent part of the uncertainty associated with these risk estimates. Additional uncertainties are summarized in section 5.3.2.5 and should be kept in mind as one considers the risk estimates in these tables.

Comparable tables to those discussed above for lung function responses in all school age children are presented in Tables 5-10 and 5-11 for asthmatic school age children for 5 urban areas that are a subset of the 12 urban areas included for all children. Again, the risk estimates associated with just meeting the current 8-hr standard presented in these tables will be discussed in the next section. For asthmatic children a lung function response defined in terms of FEV_1 decrement $\geq$ 10% is shown, consistent with CASAC's advice (Henderson, 2006c) that this level of response serves as an indicator of adverse health effects for this population. As shown in Table 5-10, across the 5 urban areas, the ranges in median estimates of the number of asthmatic school age children estimated to experience at least one FEV_1 decrement $\geq$ 10% during the O_3 season are 12,000 to 62,000 for 2004 air quality and 17,000 to about 118,000 for 2002 air quality. These median ranges represent 4.9 to 13.6% of asthmatic school age children for 2004 air quality and 12.5 to 18.3% of asthmatic school age children for 2002 air quality. In terms of total occurrences of FEV_1 decrement $\geq$ 10% associated with O_3 concentrations during the O_3 season, Table 5-11 shows a range of median estimates from 109,000 to about 660,000 responses in 2004 and from about 96,000 to 834,000 responses in 2002 for asthmatic school age children across the 5 urban areas. Dividing the estimated total number of occurrences by the number of asthmatic children estimated to experience this lung function response, results in each child being estimated to have on average between roughly 5 and 10 occurrences of this lung function response during the O_3 season depending on the urban area and year of air quality analyzed.

Table 5-8. Number and Percent of All School Age Children Estimated to Experience Lung Function Responses ($FEV_1 \geq 15\%$) Associated with 8-Hour O_3 Exposure While Engaged in Moderate Exertion for Location-Specific O_3 Seasons*

Location (O_3 Season)	Number of Children (Ages 5-18) Having at Least 1 Lung Function Response ($FEV_1 \geq 15\%$) Associated with 8-Hour O_3 Exposure**							
	Recent Air Quality (2004)		Just Meeting Current 8-Hour Standard (based on adjusting 2004 air quality)		Recent Air Quality (2002)		Just Meeting Current 8-Hour Standard (based on adjusting 2002 air quality)	
	Number (1000s)	Percent	Number (1000s)	Percent	Number (1000s)	Percent	Number (1000s)	Percent
Atlanta	34 (19 - 51)	3.6% (2% - 5.4%)	20 (8 - 34)	2.2% (0.9% - 3.6%)	59 (40 - 81)	6.3% (4.2% - 8.6%)	36 (21 - 54)	3.8% (2.2% - 5.7%)
Boston	26 (12 - 42)	2.4% (1.1% - 3.8%)	15 (4 - 27)	1.4% (0.4% - 2.5%)	84 (58 - 112)	7.6% (5.3% - 10.3%)	52 (33 - 74)	4.7% (3% - 6.8%)
Chicago	27 (6 - 49)	1.4% (0.3% - 2.5%)	15 (1 - 31)	0.8% (0% - 1.6%)	123 (83 - 169)	6.3% (4.2% - 8.7%)	71 (41 - 106)	3.6% (2.1% - 5.4%)
Cleveland	12 (5 - 20)	2% (0.8% - 3.3%)	6 (1 - 12)	1% (0.1% - 2%)	56 (40 - 74)	9.4% (6.7% - 12.4%)	30 (19 - 43)	5.1% (3.3% - 7.2%)
Detroit	20 (7 - 35)	1.8% (0.6% - 3.1%)	12 (2 - 24)	1.1% (0.2% - 2.1%)	76 (51 - 103)	6.8% (4.6% - 9.3%)	47 (29 - 69)	4.3% (2.6% - 6.3%)
Houston	57 (37 - 79)	5.2% (3.4% - 7.3%)	23 (10 - 37)	2.1% (0.9% - 3.4%)	58 (38 - 80)	5.3% (3.5% - 7.4%)	24 (11 - 38)	2.2% (1% - 3.5%)
Los Angeles	220 (149 - 298)	6% (4.1% - 8.1%)	34 (5 - 62)	0.9% (0.1% - 1.7%)	220 (150 - 297)	6% (4.1% - 8.1%)	35 (7 - 62)	0.9% (0.2% - 1.7%)
New York	112 (55 - 176)	2.7% (1.3% - 4.2%)	43 (6 - 84)	1% (0.2% - 2%)	346 (244 - 462)	8.3% (5.9% - 11.2%)	142 (79 - 216)	3.4% (1.9% - 5.2%)
Philadelphia	38 (21 - 59)	3.2% (1.8% - 4.9%)	19 (5 - 33)	1.6% (0.4% - 2.8%)	118 (85 - 155)	9.9% (7.2% - 13.1%)	63 (41 - 89)	5.4% (3.5% - 7.5%)
Sacramento	11 (6 - 17)	2.7% (1.4% - 4.1%)	4 (1 - 7)	1% (0.2% - 1.8%)	24 (16 - 32)	5.8% (3.9% - 7.9%)	10 (5 - 15)	2.3% (1.1% - 3.5%)
St. Louis	10 (3 - 18)	1.8% (0.6% - 3.1%)	7 (1 - 14)	1.3% (0.2% - 2.4%)	41 (28 - 56)	7.1% (4.8% - 9.6%)	30 (20 - 43)	5.2% (3.4% - 7.4%)
Washington, DC	57 (33 - 84)	3.8% (2.2% - 5.6%)	28 (10 - 48)	1.9% (0.7% - 3.2%)	125 (88 - 167)	8.4% (5.9% - 11.2%)	68 (42 - 98)	4.6% (2.9% - 6.6%)

*Risks are estimated for exposures in excess of policy relevant background.

**Numbers are median (0.5 fractile) numbers of children. Numbers in parentheses below the median are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient. Numbers are rounded to the nearest 1000. Percents are rounded to the nearest tenth.

Table 5-9. Number of Occurrences of Lung Function Responses ($FEV_1 \geq 15\%$) Among All School Age Children Associated with 8-Hour O_3 Exposure While Engaged in Moderate Exertion for Location-Specific O_3 Seasons*

Location	Occurrences of Lung Function Response ($FEV_1 \geq 15\%$) Associated with 8-Hour O_3 Exposure Among Children (Ages 5-18), in Thousands**			
	Recent Air Quality (2004)	Just Meeting Current 8-Hour Standard (based on adjusting 2004 air quality)	Recent Air Quality (2002)	Just Meeting Current 8-Hour Standard (based on adjusting 2002 air quality)
Atlanta (March-October)	191 (29 - 456)	131 (10 - 344)	290 (88 - 593)	196 (39 - 442)
Boston (April-September)	125 (16 - 315)	86 (5 - 238)	311 (115 - 611)	210 (56 - 458)
Chicago (April-September)	167 (6 - 460)	110 (1 - 328)	511 (171 - 1015)	325 (68 - 727)
Cleveland (April-October)	69 (6 - 179)	43 (1 - 125)	259 (110 - 473)	153 (43 - 320)
Detroit (April-September)	111 (8 - 296)	79 (2 - 227)	333 (119 - 649)	226 (56 - 488)
Houston (All year)	230 (63 - 465)	110 (13 - 253)	209 (62 - 419)	99 (13 - 223)
Los Angeles (All year)	1470 (393 - 3073)	371 (6 - 1044)	1265 (355 - 2642)	315 (9 - 869)
New York (April-September)	563 (77 - 1383)	296 (7 - 851)	1522 (585 - 2885)	753 (140 - 1727)
Philadelphia (April-October)	218 (35 - 509)	130 (6 - 345)	570 (239 - 1037)	335 (92 - 696)
Sacramento (All year)	86 (11 - 206)	41 (1 - 114)	145 (39 - 305)	72 (8 - 179)
St. Louis (April-October)	69 (4 - 181)	54 (1 - 148)	183 (65 - 356)	141 (40 - 292)
Washington, DC (April-October)	268 (50 - 622)	164 (12 - 432)	565 (209 - 1086)	345 (82 - 752)

*Risks are estimated for exposures in excess of policy relevant background.

**Numbers are median (0.5 fractile) numbers of occurrences in thousands. Numbers in parentheses below the median are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient. Numbers are rounded to the nearest 1000

The risk estimates associated with 2004 and 2002 O_3 concentrations for morbidity health endpoints based on epidemiological studies are shown in Tables 5-12 and 5-13 for respiratory symptoms in moderate to severe asthmatic children for the Boston urban area and in Tables 5-14 and 5-15 for excess hospital admissions for total respiratory illness and asthma (which is a subset of total respiratory illness admissions) for the New York City urban area. Additional hospital admission estimates for three other locations are provided in the Risk Assessment TSD. All results for morbidity health endpoints based on epidemiological studies are for health risks associated with short-term exposures to O_3 concentrations in excess of background levels from April through September for 2004 and 2002, respectively.

As discussed previously, risk estimates were developed for several respiratory symptoms in asthmatic children ages 0 to 12 who use maintenance medications based on the concentration-response functions provided in Gent et al. (2003). These estimates were developed only for the Boston urban area which was near the location of the original epidemiological study. Tables 5-12 and 5-13 show risk estimates for three different respiratory symptoms (i.e., chest tightness, shortness of breath, and wheeze) for the Boston area associated with O_3 levels above background for April through September of 2004 and 2002, respectively. The risk estimates are expressed in terms of cases, cases per 100,000 relevant population, and percent of total incidence

Tables 5-14 and 5-15 show risk estimates of unscheduled hospital admissions for respiratory illness in the New York City area associated with O_3 levels above background for April through September of 2004 and 2002, respectively. The risk estimates are expressed in terms of cases, cases per 100,000 relevant population, and percent of total incidence.

Tables 5-16 and 5-17 show risk estimates for non-accidental mortality associated with O_3 levels above background for April through September of 2004 and 2002, respectively. Similar tables for cardiorespiratory mortality are included in the Risk Assessment TSD. The risk estimates are presented in terms of estimated incidence, incidence per 100,000 relevant population, and percent of total incidence.

As discussed in section 5.3.2.5,, Bell et al. (2004) reported both multi-location and single-location concentration-response functions in a variety of locations, using a Bayesian two-stage hierarchical model. Thus, where available, risk estimates are included in Tables 5-16 and 5-17 based on both single-city and multi-city functions. The ranges shown in these tables are based either on the 95 percent confidence intervals around those estimates (if the coefficients were estimated using classical statistical techniques) or on the 95 percent credible intervals (if the coefficients were estimated using Bayesian statistical techniques).

Table 5-10. Number and Percent of Asthmatic School Age Children Estimated to Experience Lung Function Responses ($FEV_1 \geq 10\%$) Associated with 8-Hour O_3 Exposure While Engaged in Moderate Exertion for Location Specific O_3 Seasons*

Location (O_3 Season)	Asthmatic Children (Ages 5-18) Having at Least 1 Lung Function Response ($FEV_1 \geq 10\%$) Associated with 8-Hour O_3 Exposure Under Moderate Exertion**							
	Recent Air Quality (2004)		Just Meeting Current 8-Hour Standard (based on adjusting 2004 air quality)		Recent Air Quality (2002)		Just Meeting Current 8-Hour Standard (based on adjusting 2002 air quality)	
	Number (1000s)	Percent	Number (1000s)	Percent	Number (1000s)	Percent	Number (1000s)	Percent
Atlanta (March-October	12 (9 - 17)	9.9% (7.4% - 14.2%)	8 (6 - 12)	6.9% (4.8% - 10.6%)	18 (14 - 23)	15.2% (12.2% - 19.8%)	13 (10 - 18)	10.9% (8.3% - 15.3%)
Chicago (April-September)	14 (9 - 22)	4.9% (3.1% - 7.8%)	9 (5 - 14)	3.2% (1.8% - 5.1%)	40 (32 - 53)	14.5% (11.6% - 18.9%)	27 (20 - 39)	9.8% (7.3% - 14%)
Houston (All year)	17 (14 - 23)	12.6% (10% - 16.8%)	9 (6 - 14)	6.7% (4.7% - 10.1%)	17 (13 - 23)	12.5% (9.9% - 16.7%)	9 (6 - 14)	6.7% (4.8% - 10.1%)
Los Angeles (All year)	62 (52 - 81)	13.6% (11.4% - 17.7%)	16 (11 - 25)	3.5% (2.5% - 5.5%)	61 (51 - 79)	13.3% (11.1% - 17.2%)	16 (11 - 24)	3.4% (2.5% - 5.3%)
New York (April-September)	51 (37 - 76)	8% (5.8% - 11.8%)	26 (16 - 42)	4.1% (2.5% - 6.6%)	118 (97 - 147)	18.3% (15.1% - 22.9%)	63 (47 - 91)	9.8% (7.3% - 14.1%)

*Risks are estimated for exposures in excess of policy relevant background.

**Numbers are median (0.5 fractile) numbers of children. Numbers in parentheses below the median are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient. Numbers are rounded to the nearest 1000. Percents are rounded to the nearest tenth.

Table 5-11. Number of Occurrences of Lung Function Responses ($FEV_1 \geq 10\%$) Among Asthmatic School Age Children Associated with 8-Hour O_3 Exposure While Engaged in Moderate Exertion for Location Specific O_3 Seasons*

Location	Occurrences of Lung Function Response ($FEV_1 \geq 10\%$) Associated with 8-Hour O_3 Exposure Among Asthmatic Children (Ages 5-18) While Engaged in Moderate Exertion, in Thousands**			
	Recent Air Quality (2004)	Just Meeting Current 8-Hour Standard (based on adjusting 2004 air quality)	Recent Air Quality (2002)	Just Meeting Current 8-Hour Standard (based on adjusting 2002 air quality)
Atlanta (March-October)	109 (38 - 196)	82 (22 - 151)	145 (68 - 244)	109 (44 - 190)
Chicago (April-October)	114 (27 - 214)	80 (12 - 154)	257 (125 - 427)	186 (75 - 324)
Houston (All year)	110 (51 - 181)	61 (22 - 103)	96 (45 - 158)	52 (20 - 88)
Los Angeles (All year)	660 (308 - 1108)	219 (49 - 405)	561 (255 - 942)	182 (42 - 335)
New York (April-September)	399 (131 - 720)	240 (46 - 458)	834 (435 - 1356)	509 (200 - 894)

*Risks are estimated for exposures in excess of policy relevant background.

**Numbers are median (0.5 fractile) numbers of occurrences. Numbers in parentheses below the median are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient. Numbers are rounded to the nearest 1000.

Table 5-12. Estimated Respiratory Symptoms Associated with Recent (April - September, 2004) O_3 Concentrations Above Background in Boston, MA

Health Effects*	Study	Ages	Lag	Exposure Metric	Other Pollutants in Model	Health Effects Associated with O_3 Above Policy Relevant Background Levels**		
						Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	1-day lag	1 hr max.	none	5300 (800 - 9200)	20700 (3300 - 36300)	9.4% (1.5% - 16.5%)
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	0-day lag	1 hr max.	PM2.5	8400 (3800 - 12400)	33100 (14900 - 49100)	15.1% (6.8% - 22.3%)
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	1-day lag	1 hr max.	PM2.5	7700 (3000 - 11800)	30400 (11800 - 46800)	13.8% (5.4% - 21.3%)
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	1-day lag	8 hr max.	none	5400 (1700 - 8700)	21400 (6900 - 34500)	9.7% (3.1% - 15.7%)
Respiratory symptoms among asthmatic medication-users -- shortness of breath	Gent et al. (2003)	0 - 12	1-day lag	1 hr max.	none	5700 (700 - 10200)	22500 (2700 - 40200)	8.2% (1% - 14.7%)
Respiratory symptoms among asthmatic medication-users -- shortness of breath	Gent et al. (2003)	0 - 12	1-day lag	8 hr max.	none	6300 (1200 - 10800)	24700 (4800 - 42500)	9% (1.8% - 15.5%)
Respiratory symptoms among asthmatic medication-users -- wheeze	Gent et al. (2003)	0 - 12	0-day lag	1 hr max.	PM2.5	15400 (5500 - 24200)	60800 (21800 - 95600)	11.9% (4.3% - 18.7%)

*Health effects are associated with short-term exposures to O_3.

**Incidence was quantified down to estimated policy relevant background levels. Incidences of respiratory symptom-days and respiratory symptom-days per 100,000 relevant population are rounded to the nearest 100. Percents are rounded to the nearest tenth.

Note: Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

Table 5-13. Estimated Respiratory Symptoms Associated with Recent (April - September, 2002) O_3 Concentrations Above Background in Boston, MA

Health Effects*	Study	Ages	Lag	Exposure Metric	Other Pollutants in Model	Health Effects Associated with O_3 Above Policy Relevant Background Levels**		
						Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	1-day lag	1 hr max.	none	6900 (1100 - 11800)	27200 (4500 - 46600)	12.4% (2% - 21.2%)
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	0-day lag	1 hr max.	PM2.5	10800 (5000 - 15700)	42700 (19700 - 62100)	19.5% (9% - 28.3%)
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	1-day lag	1 hr max.	PM2.5	10000 (4000 - 15000)	39400 (15700 - 59400)	17.9% (7.1% - 27%)
Respiratory symptoms among asthmatic medication-users -- chest tightness	Gent et al. (2003)	0 - 12	1-day lag	8 hr max.	none	7200 (2400 - 11400)	28400 (9300 - 44900)	12.9% (4.2% - 20.5%)
Respiratory symptoms among asthmatic medication-users -- shortness of breath	Gent et al. (2003)	0 - 12	1-day lag	1 hr max.	none	7500 (900 - 13200)	29500 (3700 - 52000)	10.8% (1.3% - 19%)
Respiratory symptoms among asthmatic medication-users -- shortness of breath	Gent et al. (2003)	0 - 12	1-day lag	8 hr max.	none	8300 (1700 - 14000)	32800 (6600 - 55300)	11.9% (2.4% - 20.2%)
Respiratory symptoms among asthmatic medication-users -- wheeze	Gent et al. (2003)	0 - 12	0-day lag	1 hr max.	PM2.5	20100 (7400 - 31000)	79200 (29000 - 122300)	15.5% (5.7% - 23.9%)

*Health effects are associated with short-term exposures to O_3.

**Incidence was quantified down to estimated policy relevant background levels. Incidences of respiratory symptom-days and respiratory symptom-days per 100,000 relevant population are rounded to the nearest 100. Percents are rounded to the nearest tenth.

Note: Numbers in parentheses are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

Table 5-14. Estimated Hospital Admissions Associated with Recent (April - September, 2004) O_3 Concentrations in NY, NY**

Health Effects*	Study	Ages	Lag	Exposure Metric	Other Pollutants in Model	Health Effects Associated with O_3 Above Policy Relevant Background Levels*		
						Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Hospital admissions (unscheduled), respiratory illness	Thurston et al. (1992)***	all	3-day lag	1 hr max.	none	447 (108 - 786)	5.6 (1.4 - 9.8)	1.3% (0.3% - 2.2%)
Hospital admissions (unscheduled), asthma	Thurston et al. (1992)***	all	1-day lag	1 hr max.	none	382 (81 - 683)	4.8 (1 - 8.5)	2.9% (0.6% - 5.2%)

*Incidences are rounded to the nearest whole number; incidences per 100,000 relevant population and percents are rounded to the nearest tenth.
**New York in this study is defined as the five boroughs of New York City.

Note: Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

Table 5-15. Estimated Hospital Admissions Associated with Recent (April - September, 2002) O_3 Concentrations in NY, NY**

Health Effects	Study	Ages	Lag	Exposure Metric	Other Pollutants in Model	Health Effects Associated with O_3 Above Policy Relevant Background Levels*		
						Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Hospital admissions (unscheduled), respiratory illness	Thurston et al. (1992)***	all	3-day lag	1 hr max.	none	608 (147 - 1068)	7.6 (1.8 - 13.3)	1.7% (0.4% - 3%)
Hospital admissions (unscheduled), asthma	Thurston et al. (1992)***	all	1-day lag	1 hr max.	none	519 (110 - 928)	6.5 (1.4 - 11.6)	4% (0.8% - 7.1%)

*Incidences are rounded to the nearest whole number; incidences per 100,000 relevant population and percents are rounded to the nearest tenth.
**New York in this study is defined as the five boroughs of New York City.

Note: Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

Table 5-16. Estimated Non-Accidental Mortality Associated with Recent (April - September, 2004) O3 Concentrations*

Location	Study	Lag	Exposure Metric	Non-Accidental Mortality Associated with O_3 Above Policy Relevant Background Levels**		
				Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Atlanta	Bell et al. (2004)	distributed lag	24 hr avg.	6 (-26 - 38)	0.4 (-1.8 - 2.6)	0.1% (-0.6% - 0.8%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	12 (4 - 20)	0.8 (0.3 - 1.4)	0.3% (0.1% - 0.4%)
Boston	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	7 (2 - 12)	1.0 (0.3 - 1.7)	0.3% (0.1% - 0.5%)
Chicago	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	49 (16 - 81)	0.9 (0.3 - 1.5)	0.2% (0.1% - 0.4%)
	Schwartz (2004)	0-day lag	1 hr max.	394 (125 - 658)	7.3 (2.3 - 12.2)	1.9% (0.6% - 3.1%)
	Schwartz -- 14 US Cities (2004)	0-day lag	1 hr max.	148 (46 - 250)	2.8 (0.9 - 4.6)	0.7% (0.2% - 1.2%)
Cleveland	Bell et al. (2004)	distributed lag	24 hr avg.	27 (-17 - 69)	1.9 (-1.2 - 5)	0.4% (-0.2% - 0.9%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	17 (6 - 28)	1.2 (0.4 - 2)	0.2% (0.1% - 0.4%)
Detroit	Bell et al. (2004)	distributed lag	24 hr avg.	33 (-11 - 76)	1.6 (-0.5 - 3.7)	0.4% (-0.1% - 0.8%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	17 (6 - 28)	0.8 (0.3 - 1.4)	0.2% (0.1% - 0.3%)
	Schwartz (2004)	0-day lag	1 hr max.	128 (-21 - 274)	6.2 (-1 - 13.3)	1.4% (-0.2% - 2.9%)
	Schwartz -- 14 US Cities (2004)	0-day lag	1 hr max.	70 (22 - 117)	3.4 (1.1 - 5.7)	0.7% (0.2% - 1.2%)
	Ito (2003)	0-day lag	24 hr avg.	40 (-37 - 116)	2.0 (-1.8 - 5.6)	0.4% (-0.4% - 1.2%)
Houston	Bell et al. (2004)	distributed lag	24 hr avg.	35	1.0	0.4%

Location	Study	Lag	Exposure Metric	Non-Accidental Mortality Associated with O_3 Above Policy Relevant Background Levels**		
				Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
				(2 - 67)	(0.1 - 2)	(0% - 0.7%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	17 (6 - 28)	0.5 (0.2 - 0.8)	0.2% (0.1% - 0.3%)
	Schwartz (2004)	0-day lag	1 hr max.	93 (9 - 176)	2.7 (0.3 - 5.2)	1% (0.1% - 1.9%)
	Schwartz -- 14 US Cities (2004)	0-day lag	1 hr max.	78 (24 - 130)	2.3 (0.7 - 3.8)	0.9% (0.3% - 1.4%)
Los Angeles	Bell et al. (2004)	distributed lag	24 hr avg.	62 (-149 - 271)	0.6 (-1.6 - 2.8)	0.2% (-0.5% - 1%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	133 (45 - 221)	1.4 (0.5 - 2.3)	0.5% (0.2% - 0.8%)
New York	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	60 (20 - 100)	0.7 (0.2 - 1.1)	0.2% (0.1% - 0.3%)
Philadelphia	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	23 (8 - 38)	1.5 (0.5 - 2.5)	0.3% (0.1% - 0.5%)
	Moolgavkar et al. (1995)	1-day lag	24 hr avg.	82 (52 - 112)	5.4 (3.4 - 7.4)	1% (0.6% - 1.4%)
Sacramento	Bell et al. (2004)	distributed lag	24 hr avg.	12 (-36 - 59)	1.0 (-3 - 4.8)	0.3% (-0.9% - 1.4%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	18 (6 - 29)	1.4 (0.5 - 2.4)	0.4% (0.1% - 0.7%)
St Louis	Bell et al. (2004)	distributed lag	24 hr avg.	3 (-6 - 13)	1.0 (-1.7 - 3.6)	0.2% (-0.3% - 0.6%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	3 (1 - 5)	0.9 (0.3 - 1.5)	0.2% (0.1% - 0.3%)
Washington, D.C.	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	8 (3 - 14)	1.5 (0.5 - 2.4)	0.3% (0.1% - 0.5%)

*All results are for mortality (among all ages) associated with short-term exposures to O_3. All results are based on single-pollutant models.
**Incidence was quantified down to estimated policy relevant background levels. Incidences are rounded to the nearest whole number; incidences per 100,000 relevant population and percents are rounded to the nearest tenth.
Note: Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

Table 5-17. Estimated Non-Accidental Mortality Associated with Recent (April - September, 2002) O_3 Concentrations*

Location	Study	Lag	Exposure Metric	Non-Accidental Mortality Associated with O_3 Above Policy Relevant Background Levels**		
				Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Atlanta	Bell et al. (2004)	distributed lag	24 hr avg.	9 (-37 - 54)	0.6 (-2.5 - 3.6)	0.2% (-0.8% - 1.2%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	17 (6 - 29)	1.2 (0.4 - 1.9)	0.4% (0.1% - 0.6%)
Boston	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	10 (3 - 17)	1.5 (0.5 - 2.5)	0.4% (0.1% - 0.7%)
Chicago	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	69 (23 - 115)	1.3 (0.4 - 2.1)	0.3% (0.1% - 0.5%)
	Schwartz (2004)	0-day lag	1 hr max.	505 (161 - 840)	9.4 (3 - 15.6)	2.4% (0.8% - 4%)
	Schwartz -- 14 US Cities (2004)	0-day lag	1 hr max.	191 (60 - 321)	3.6 (1.1 - 6)	0.9% (0.3% - 1.5%)
Cleveland	Bell et al. (2004)	distributed lag	24 hr avg.	61 (-38 - 157)	4.3 (-2.7 - 11.3)	0.8% (-0.5% - 2.1%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	38 (13 - 64)	2.8 (0.9 - 4.6)	0.5% (0.2% - 0.9%)
Detroit	Bell et al. (2004)	distributed lag	24 hr avg.	57 (-18 - 131)	2.8 (-0.9 - 6.3)	0.6% (-0.2% - 1.4%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	29 (10 - 48)	1.4 (0.5 - 2.3)	0.3% (0.1% - 0.5%)
	Schwartz (2004)	0-day lag	1 hr max.	181 (-30 - 385)	8.8 (-1.4 - 18.7)	1.9% (-0.3% - 4.1%)
	Schwartz -- 14 US Cities (2004)	0-day lag	1 hr max.	99 (31 - 165)	4.8 (1.5 - 8)	1% (0.3% - 1.8%)
	Ito (2003)	0-day lag	24 hr avg.	69 (-64 - 198)	3.4 (-3.1 - 9.6)	0.7% (-0.7% - 2.1%)

Location	Study	Lag	Exposure Metric	Non-Accidental Mortality Associated with O_3 Above Policy Relevant Background Levels**		
				Incidence	Incidence per 100,000 Relevant Population	Percent of Total Incidence
Houston	Bell et al. (2004)	distributed lag	24 hr avg.	29 (2 - 57)	0.9 (0.1 - 1.7)	0.3% (0% - 0.6%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	14 (5 - 24)	0.4 (0.1 - 0.7)	0.2% (0.1% - 0.3%)
	Schwartz (2004)	0-day lag	1 hr max.	85 (8 - 161)	2.5 (0.2 - 4.7)	0.9% (0.1% - 1.8%)
	Schwartz -- 14 US Cities (2004)	0-day lag	1 hr max.	71 (22 - 119)	2.1 (0.7 - 3.5)	0.8% (0.2% - 1.3%)
Los Angeles	Bell et al. (2004)	distributed lag	24 hr avg.	51 (-124 - 224)	0.5 (-1.3 - 2.4)	0.2% (-0.5% - 0.8%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	110 (37 - 184)	1.2 (0.4 - 1.9)	0.4% (0.1% - 0.7%)
New York	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	105 (35 - 174)	1.2 (0.4 - 2)	0.3% (0.1% - 0.6%)
Philadelphia	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	37 (12 - 62)	2.4 (0.8 - 4.1)	0.5% (0.2% - 0.8%)
	Moolgavkar et al. (1995)	1-day lag	24 hr avg.	132 (83 - 180)	8.7 (5.5 - 11.9)	1.6% (1% - 2.2%)
Sacramento	Bell et al. (2004)	distributed lag	24 hr avg.	16 (-48 - 78)	1.3 (-3.9 - 6.4)	0.4% (-1.1% - 1.9%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	23 (8 - 39)	1.9 (0.6 - 3.2)	0.6% (0.2% - 0.9%)
St Louis	Bell et al. (2004)	distributed lag	24 hr avg.	6 (-11 - 23)	1.9 (-3.1 - 6.7)	0.3% (-0.5% - 1.2%)
	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	6 (2 - 10)	1.7 (0.6 - 2.8)	0.3% (0.1% - 0.5%)
Washington, D.C.	Bell et al. -- 95 US Cities (2004)	distributed lag	24 hr avg.	15 (5 - 25)	2.6 (0.9 - 4.4)	0.6% (0.2% - 0.9%)

*All results are for mortality (among all ages) associated with short-term exposures to O_3. All results are based on single-pollutant models.
**Incidence was quantified down to estimated policy relevant background levels. Incidences are rounded to the nearest whole number; incidences per 100,000 relevant population and percents are rounded to the nearest tenth.
Note: Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

We observe from Tables 5-16 and 5-17 that estimates of O_3-related non-accidental mortality reported by Schwartz (2004) for Chicago, Detroit, and Houston, based on both single city and multi-city concentration-response functions, tend to be higher than other estimates for these locations. This is mainly due to the use of the 1-hr maximum O_3 concentration in Schwartz (2004), rather than the 24-hr average, as the exposure metric. The changes from recent (2004 or 2002)) 1-hr maximum to background 1-hr maximum O_3 concentrations were generally larger in the assessment locations than the corresponding changes from recent 24-hr average to background 24-hr average O_3 concentrations. For example, for 2004 air quality the estimated O_3-related (non-accidental) mortality in Detroit based on Bell et al. (2004), which used a 24-hr average indicator, ranged from 0.2% (based on 95 city model) to 0.4% of total incidence (based on single-city model). In contrast, the estimated O_3-related (non-accidental) mortality in Detroit based on Schwartz (2004), which used a 1-hr maximum O_3 concentration as the indicator, ranged from 0.7% (based on 14 city model) to 1.4% (based on single-city model).

Figures 5-6a and b show the estimated annual percent of non-accidental mortality associated with short-term exposure to O_3 concentrations within specified ranges for the warm O_3 season (April 1 to September 30) in two recent years. While the current O_3 standard is expressed in terms of an 8-hr daily maximum inidicator, the large multicity non-accidental (Bell et al. (2004) and cardiorespiratory (Huang et al. (2004) mortality studies reported concentration-response relationships for 24-hr average O_3 levels. Thus, the intervals shown in this figure are for 24-hr average concentrations. To provide some perspective on the 24-hr intervals shown, scatter plots comparing 8-hr daily maximum concentrations at the highest monitor with the average of the 24-hr average over all monitors within an urban area were developed and are included in Appendix 5A.2. These scatter plots show that 8-hr daily maximum concentrations on average are roughly twice the observed 24-hr average levels, although there is considerable variability in this relationship from day-to- day within an urban area.

As shown in Figure 5-6a, in 2004, all O_3-related non-accidental mortality was associated with O_3 concentrations less than 0.06 ppm, 24 hr average, and most of that was associated with O_3 concentrations less than 0.04 ppm, 24-hr average. As shown in Figure 5-6b, in 2002, all O_3-related non-accidental mortality was associated with O_3 concentrations less than 0.08 ppm, 24-hr average and the great majority was associated with O_3 concentrations less than 0.05 ppm, 24-hr average. The results for cardiorespiratory mortality follow a similar pattern and are shown in Figure 4-15 in the Risk Assessment TSD.

Figure 5-6. Estimated Annual Percent of Non-Accidental Mortality Associated with Short-Term Exposure to Recent O_3 Concentrations Above Background for the Period April – September (Based on Bell et al., 2004) – Total and Contribution of 24-Hour Average O_3 Ranges*

Figure 5-6a. Based on 2004 Air Quality

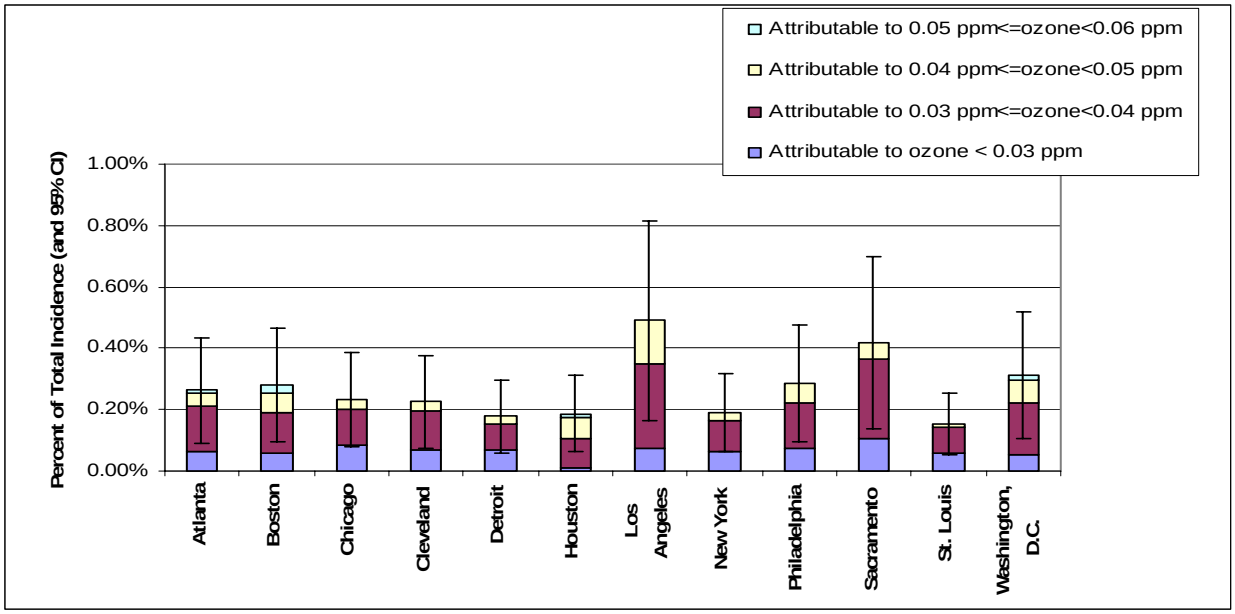
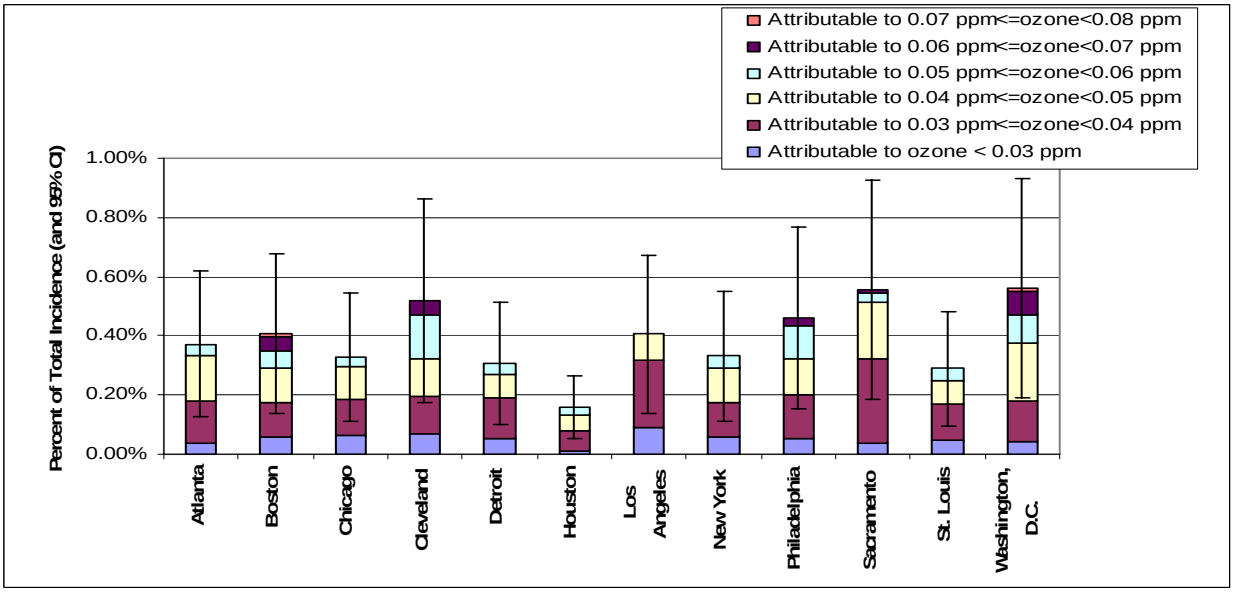


Figure 5-6b. Based on 2002 Air Quality



*Note that as shown in scatter plots in Appendix 5A.2, 8-hr daily maximum concentrations at the highest monitor are roughly twice the level of the average of the 24-hr average O_3 concentrations over all monitors within an urban area which are used in this figure, this ratio varies across areas.

5.4.2 Just Meeting Current and Alternative Ozone Standards

As described in Chapter 4 and briefly in section 5.3.2.2, the risk estimates described in this section represent the risks for two separate O_3 seasons based on adjusting the O_3 levels observed in 2004 or 2002 to simulate O_3 levels associated with just meeting the current 0.08 ppm standard and several potential alternative 8-hr standards, using the 3-year design value from the 2002-2004 time period. To facilitate comparison of risk estimates across the urban areas, figures used in this section present summaries of the risk estimates for the current and potential alternative 8-hr daily maximum standards. Most of the figures and tables in this section examine the risks associated with alternative standards using the average 4th-highest daily maximum 8-hr average form of the current standard. We present only limited results for several additional alternative standards in this section. Risk estimates for three additional alternative 8-hr standards (0.084 and 0.074 ppm, using an average of the annual 3rd-highest daily maximum 8-hr concentrations averaged over the three year period, and 0.074 ppm using an average of the annual 5th-highest daily maximum 8-hr averages over the three year period are more fully presented in tables in the Risk Assessment TSD. Because we had to simulate the profiles of O_3 concentrations that just meet the current and alternative 8-hour daily maximum O_3 standards in each location, there is additional uncertainty surrounding estimates of the reduced incidence associated with O_3 concentrations that just meet these O_3 standards.

This section first discusses the risk estimates for lung function responses in all and asthmatic school age children, which are based on exposure-response relationships derived from controlled human exposure studies, and then risk estimates are explored for respiratory symptoms in asthmatic children, respiratory-related hospital admissions, and premature mortality which are based on concentration-response relationships obtained from epidemiological studies

The risk estimates for lung function responses are for the O_3 season, which is all year in 3 of the study areas (Houston, Los Angeles, and Sacramento) and which is generally 6-7 months long in the other 9 urban study areas (e.g., March or April to September or October). The risk estimates for lung function responses in all school age children (ages 5 to 18) for just meeting the current 8-hr standard for 12 urban areas are summarized in Tables 5-6 and 5-7 presented in the previous section. Similarly, risk estimates for lung function responses in asthmatic school age children (ages 5 to 18) for just meeting the current 8-hr standard for 5 urban areas are summarized in Tables 5-8 and 5-9 in the previous section. Additional risk estimates for all and asthmatic school age children are presented in the Risk Assessment TSD and Post (2007), including estimates based on adjusting 2003 air quality to just meet the current and several alternative standards.

Figure 5-7 shows the median estimates of the percent of all school age children estimated to experience at least one FEV_1 decrement $\geq$ 15% during the O_3 season across the 12 urban areas for recent air quality (2002) and upon just meeting the current and several alternative 8-hr standards. Figure 5C-1 in Appendix 5C of this Staff Paper shows a similar figure based on 2004 air quality data. For just meeting the current 8-hr standard the ranges of median estimates across the 12 urban areas are 0.9 - 5.4% based on adjusting 2002 air quality data and 0.8 - 2.2% based on adjusting 2004 air quality data. In terms of total occurrences of FEV_1 decrement $\geq$ 15% during the O_3 season, Table 5-9 shows a range of median estimates from about 72,000 to over 750,000 responses during the O_3 season for all school age children based on adjusting 2002 air quality data to just meeting the current 8-hour standard and from about 40,000 to about 370,000 responses across the 12 urban areas associated with adjusting 2004 O_3 concentrations to just meeting the current 8-hour standard.

As an illustration of the changes in the number of school age children estimated to experience FEV_1 decrements $\geq$ 15% across the range of alternative standards, under the current standard the median estimates range from 10,000 to about 142,000 children per urban area across the 12 urban areas and this would be reduced to a range of 1,000 to 20,000 children under the most stringent alternative standard examined (i.e., 0.064 ppm, 4th-highest daily 8-hr maximum). Somewhat lower estimates are observed based on adjusting 2004 air quality to just meet the current and alternative 8-hr standards, with a range from 4,000 to about 43,000 children for just meeting the current standard which is reduced to a range from 1,000 to 14,000 children under the 0.064 ppm, 4th-highest daily 8-hr maximum standard. By comparing the estimated number of occurrences shown in Tables 5C-5 with the number of children estimated to experience 1 or more responses shown in Tables 5C-1, one can get an estimate of the average number of occurrences of a given response in an O_3 season. For example, for Atlanta it is estimated that 36,000 children would have an FEV_1 decrement $\geq$ 15% and that there would be 196,000 occurrences of this response in this same population when 2002 air quality is adjusted to just meet the current 8-hr standard. Thus, on average it is estimated that there would be about 5 occurrences per O_3 season per responding child for air quality just meeting the current 8-hr standard in this urban area. We recognize that some children in the population might have only 1 or 2 occurrences while others likely have 6 or more occurrences per O_3 season.

Figure 5-8 shows the 95% confidence intervals for the lung function risk estimates for each of the 12 urban areas using the FEV_1 decrement $\geq$ 15% health response for recent O_3 levels (2002) and for 2002 air quality adjusted to just meet the current and alternative 8-hr average n^{th} daily maximum standards. A comparable figure (Figure 5C-2) using 2004 air quality and adjusting 2004 air quality to just meet the current and alternative 8-hr standards is included in Appendix 5C.

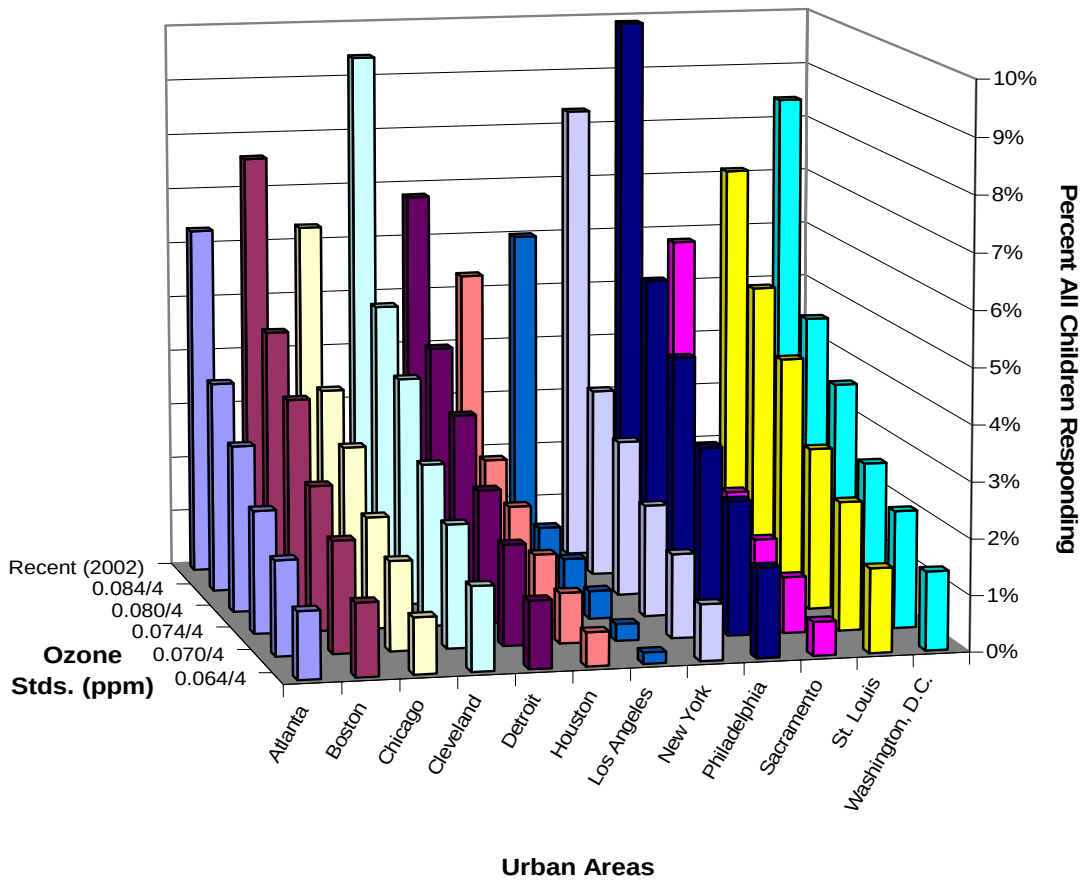
Figures 5-9 shows the median estimates of the percent of asthmatic school age children estimated to experience at least one FEV_1 decrement $\geq$ 10% during the O_3 season across the five urban areas for recent air quality (2002) and upon just meeting the current and several alternative 8-hr average 4th-highest daily maximum standards. Figure 5C-3 in Appendix 5C of this Staff Paper shows a similar figure based on 2004 air quality data. For just meeting the current 8-hr standard the ranges of median estimates across the 5 urban areas are 3.2 - 6.9% based on adjusting 2004 air quality data and 3.4 – 10.9% based on adjusting 2002 air quality data. In terms of total occurrences of FEV_1 decrement $\geq$ 15% during the O_3 season, Table 5-9 shows a range of median estimates from about 61,000 to 240,000 responses during the O_3 season for asthmatic school age children based on adjusting 2004 air quality data to just meeting the current 8-hour standard and from about 52,000 to nearly 510,000 responses across the 5 urban areas associated with adjusting 2002 O_3 concentrations to just meeting the current 8-hour standard.

Figure 5-10 shows the 95% confidence intervals for the lung function risk estimates for asthmatic school age children in each of the 5 urban areas using the FEV_1 decrement $\geq$ 10% health response for recent O_3 levels (2002) and for 2002 air quality adjusted to just meet the current and alternative 8-hr average 4th-highest daily maximum standards. A comparable figure (Figure 5C-4) using 2004 air quality and adjusting 2004 air quality to just meet the current and alternative 8-hr standards is included in Appendix 5C.

Figure 5-11 summarizes respiratory symptom response risk estimates associated with O_3 exposures during the April to September period for moderate/severe asthmatic children ages 0 to 12 in the Boston urban area based on the concentration-response relationships reported in Gent et al. (2003) for 2002 air quality and the current and alternative 8-hr standards based on adjusting 2004 air quality data. Figure 5C-5 (Appendix 5C) presents comparable estimates associated with 2004 air quality and just meeting the current and alternative 8-hr standards based on adjusting 2004 air quality data. These figures include risk estimates for chest tightness based on single pollutant models and models that included $PM_{2.5}$. Two additional symptom endpoints, shortness of breath and wheeze are included in the tables in the Risk Assessment TSD and show similar patterns as the risk estimates for chest tightness.

The median estimated number of days involving chest tightness (using the concentration-response relationship with only O_3 in the model) ranges from 4,500 (based on adjusting 2004 air quality) to 6,100 (based on adjusting 2002 air quality) upon meeting the current 8-hr standard and these are reduced to 3,100 (based on adjusting 2004 air quality) to 4,600 days upon meeting the most stringent alternative examined (0.064 ppm, 4th-highest daily maximum 8-hr average). These same ranges correspond to 8 - 11% of total incidence of chest tightness upon meeting the

Figure 5-7. Percent of All Children (Ages 5-18) Engaged in Moderate Exertion Estimated to Experience At Least One Lung Function Response (Decrement in $FEV_1 \geq$ 15%) Associated with Exposure to O_3 Concentrations That Just Meet the Current and Alternative Average 4th-highest Daily Maximum 8-Hour Standards, for Location-Specific O_3 Seasons (Based on Adjusting 2002 Air Quality)*



*An 8-hr average standard, denoted m/n is characterized by a concentration m ppm and an nth daily maximum. So for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. The 4th-highest daily maximum standards, denoted m/4, require that the average of the 3 annual nth-highest daily maxima over a three year period be at or below the specified level (m ppm). 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient are presented in Table 5C-2 in Appendix 5C.

Figure 5-8. Percent of All Children (Ages 5-18) Engaged in Moderate Exertion Estimated to Experience At Least One Lung Function Response (Decrement in $FEV_1 \geq 15$ %) Associated with Recent Air Quality (2002) and Just Meeting the Current and Alternative Average nth Daily Maximum 8-Hour Standards, for Location-Specific O_3 Seasons (Based on Adjusting 2002 Air Quality)*

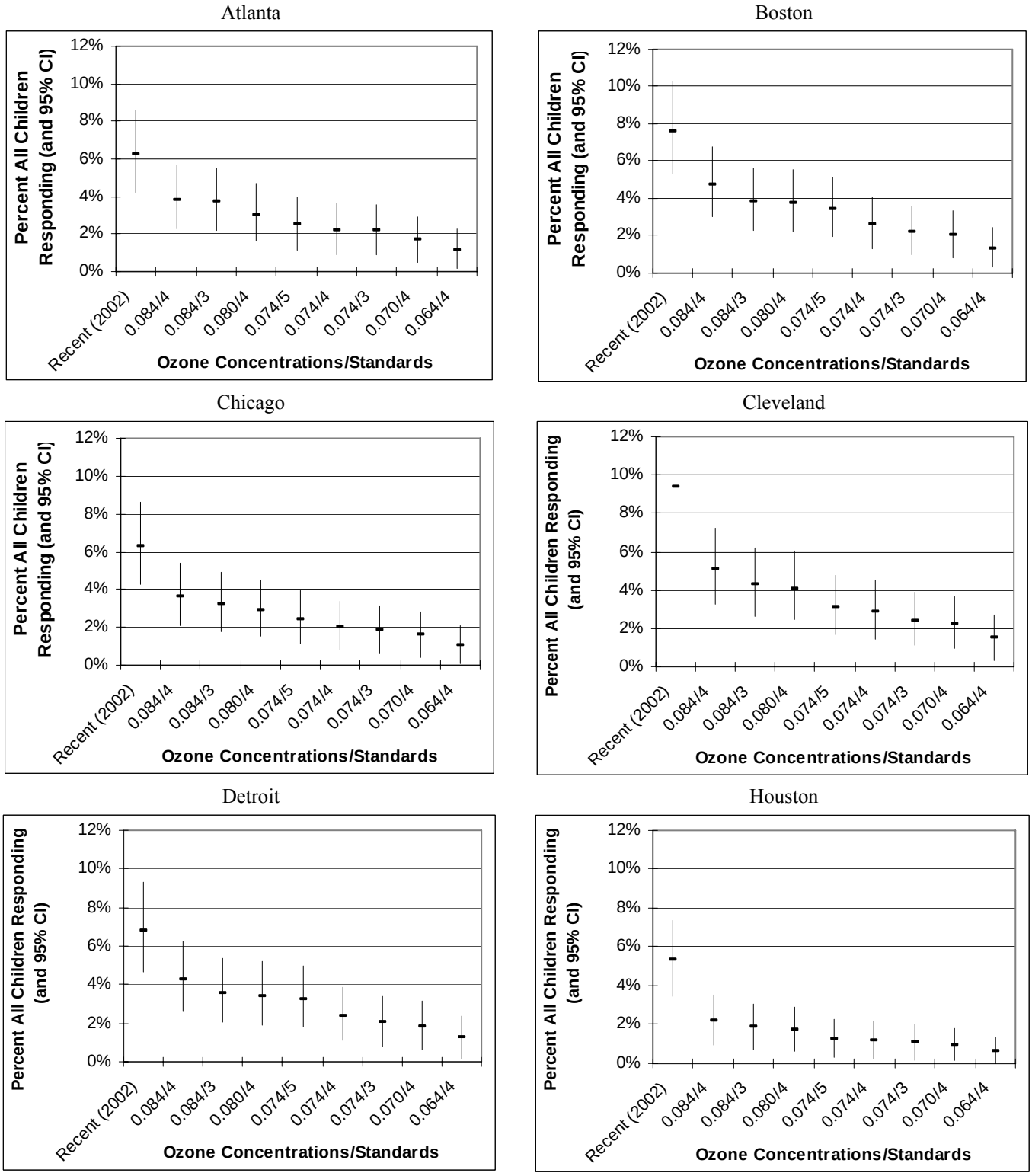
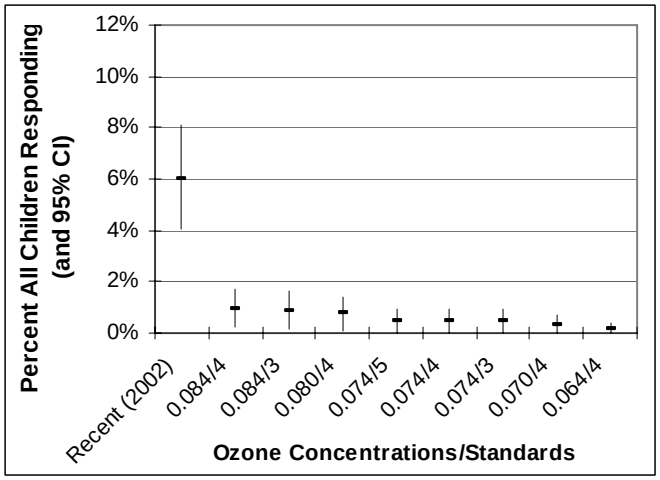
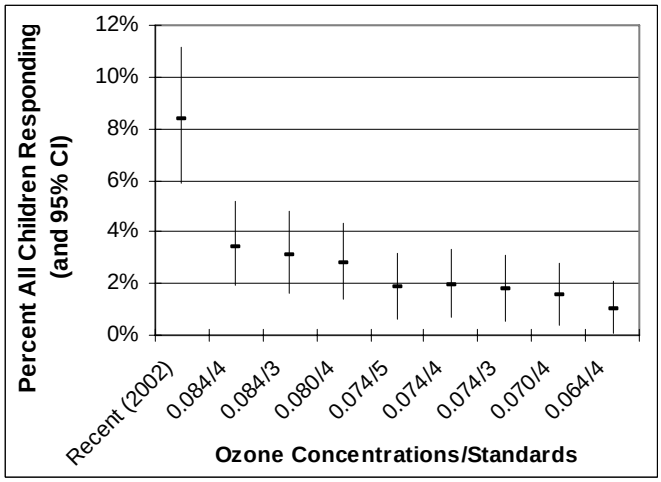


Figure 5-8. (Continued)

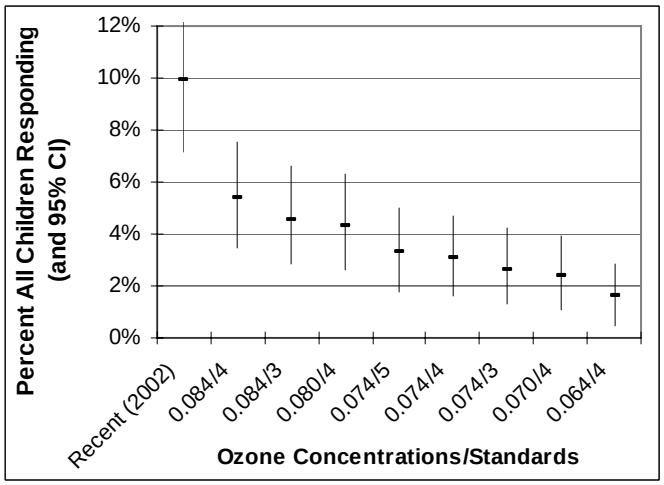
Los Angeles



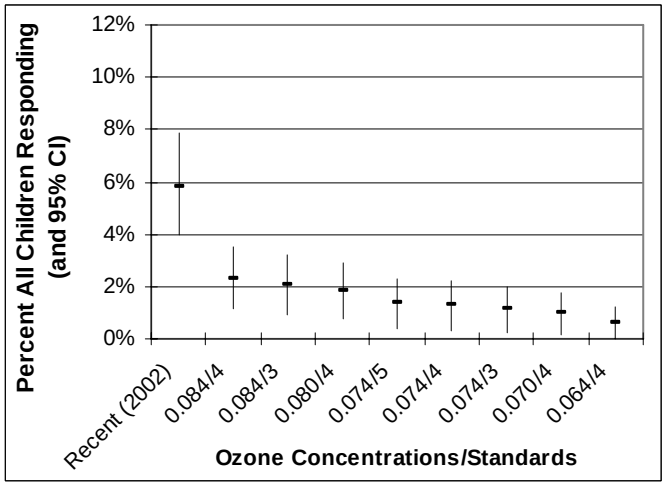
New York



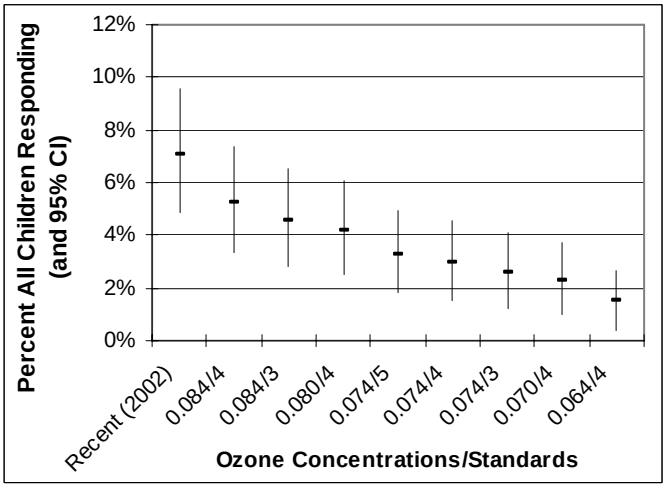
Philadelphia



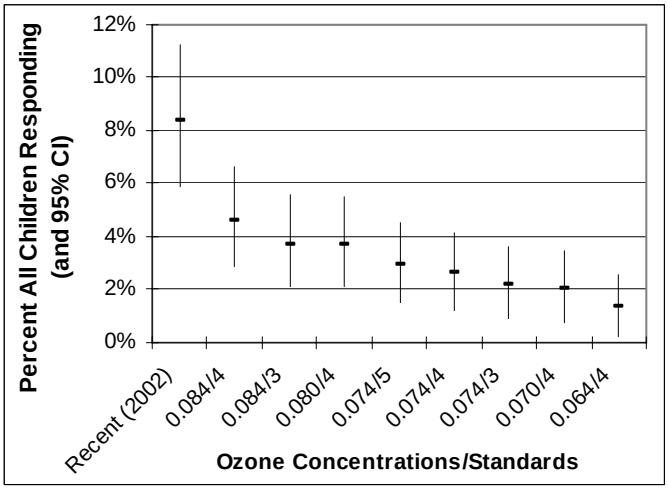
Sacramento



St. Louis

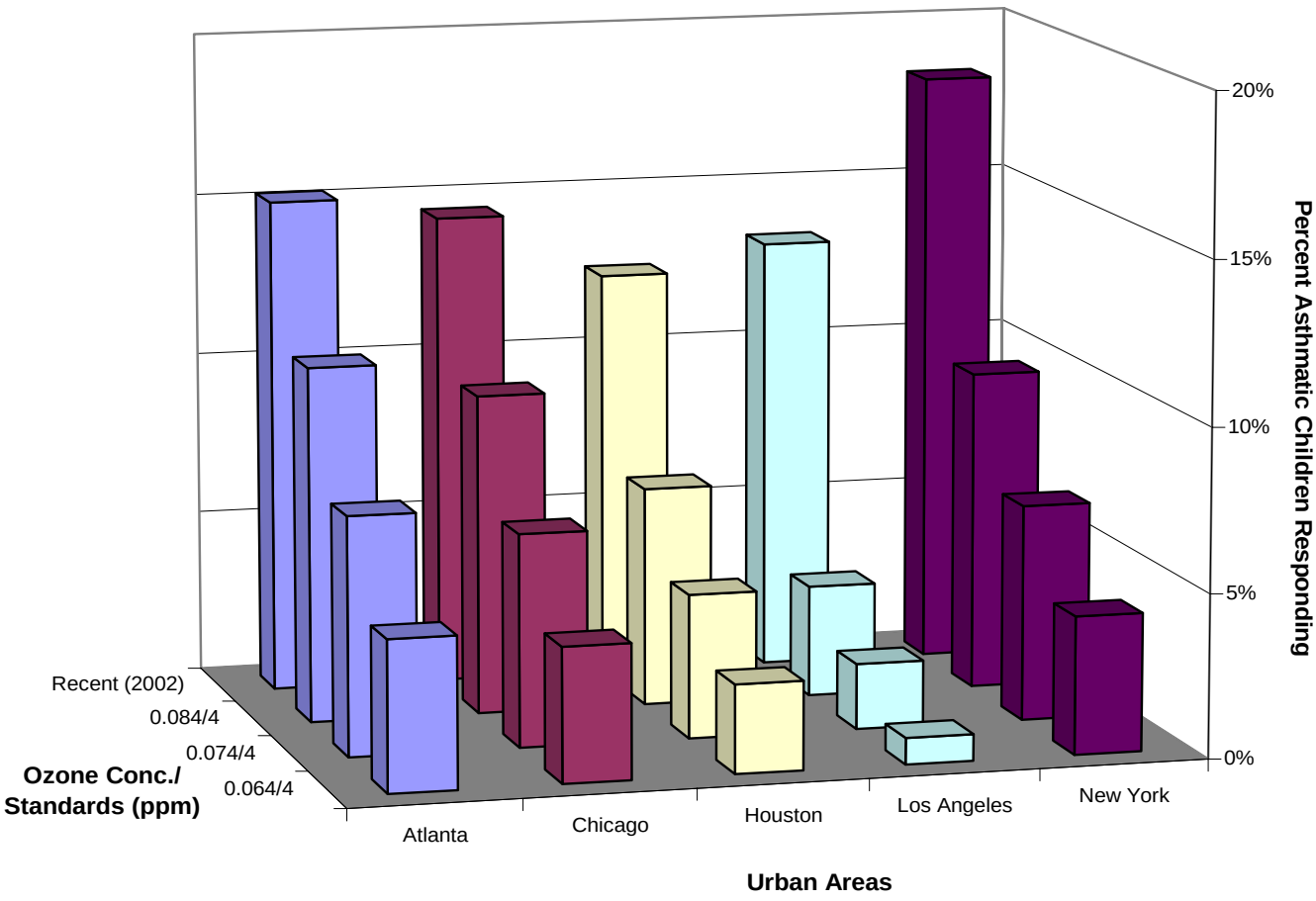


Washington, D.C.



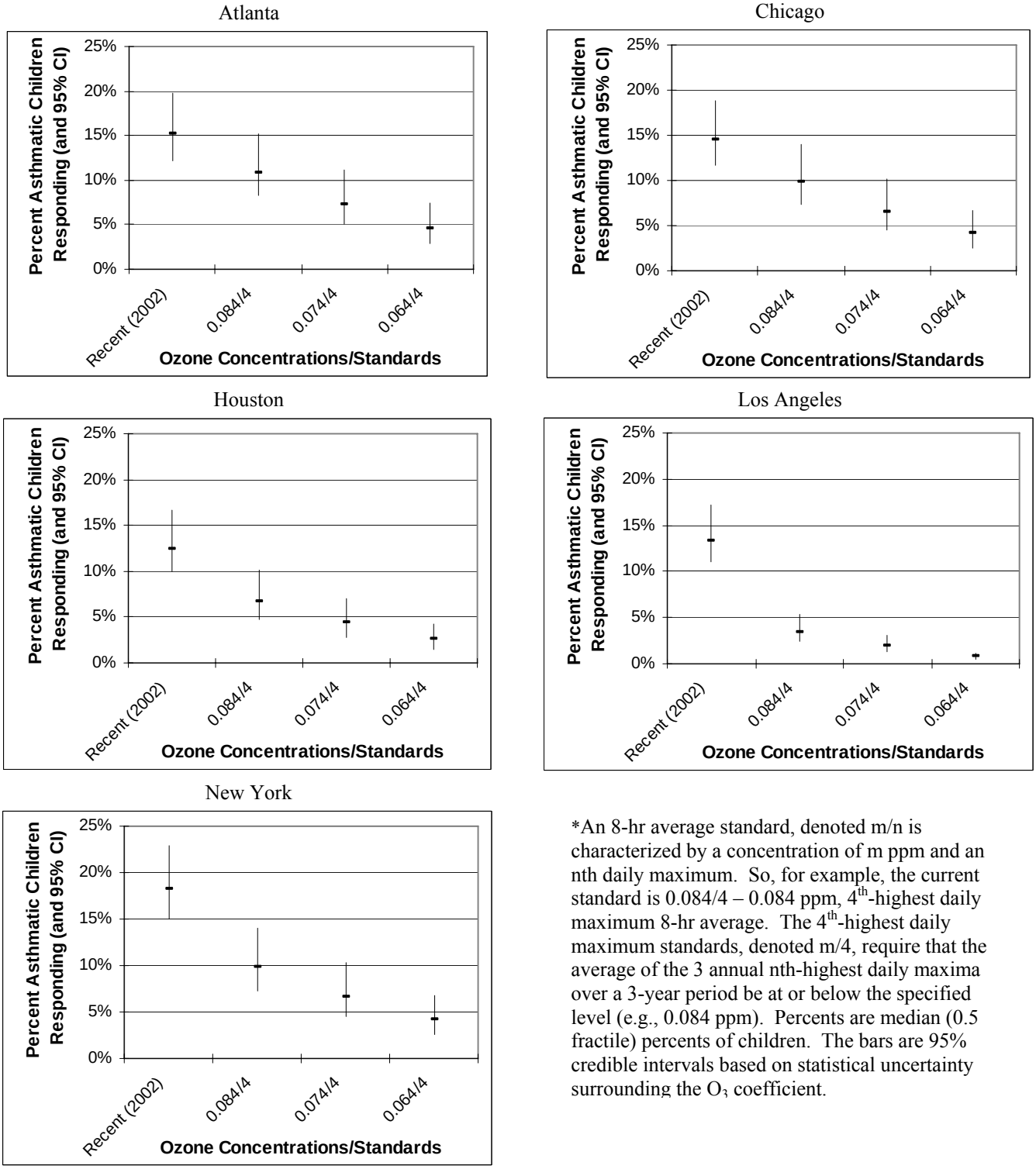
*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth-highest daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. Percents are median (0.5 fractile) percents of children. The bars are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

Figure 5-9. Percent of Asthmatic Children (Ages 5-18) Engaged in Moderate Exertion Estimated to Experience At Least One Lung Function Response (Decrement in $FEV_1 \geq 10\%$) Associated with Exposure to O_3 Concentrations That Just Meet the Current and Alternative Average 4th-highest Daily Maximum 8-Hour Standards, for Location-Specific O_3 Seasons (Based on Adjusting 2002 Air Quality)



*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth-highest daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. The 4th-highest daily maximum standards, denoted m/4, require that the average of the 3 annual nth-highest daily maxima over a 3-year period be at or below the specified level. 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient are presented in Table 5C-5 in Appendix 5C.

Figure 5-10. Percent of Asthmatic Children (Ages 5-18) Engaged in Moderate Exertion Estimated to Experience At Least One Lung Function Response (Decrement in $FEV_1 \geq 10$ %) Associated with Recent Air Quality (2002) and Exposure to O_3 Concentrations That Just Meet the Current and Alternative 8-Hour Standards, for Location-Specific O_3 Seasons: Based on Adjusting 2002 O_3 Concentrations*



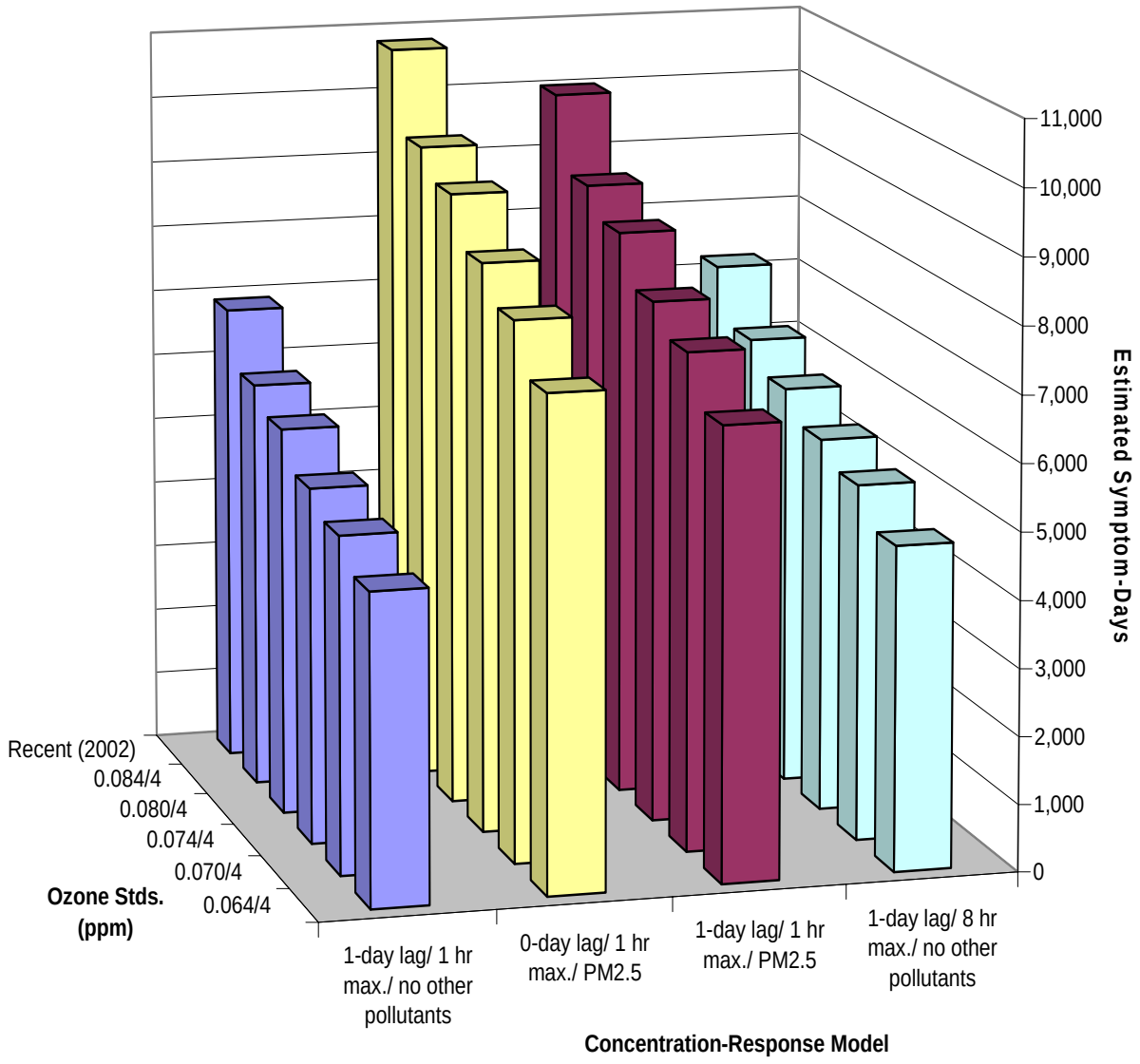
*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. The 4th-highest daily maximum standards, denoted m/4, require that the average of the 3 annual nth-highest daily maxima over a 3-year period be at or below the specified level (e.g., 0.084 ppm). Percents are median (0.5 fractile) percents of children. The bars are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

current 8-hr standard and to about 5.5 - 8% of total incidence of chest tightness upon meeting a 0.064 ppm, 4th-highest daily maximum 8-hr average standard. As shown in Tables 5C-7 and 5C-9 (Appendix 5C), the symptom with the greatest incidence is wheeze and is based on an O_3 concentration-response relationship that included $PM_{2.5}$ in the model. These median estimates range from about 13,000 days with wheeze (based on adjusting 2004 air quality) to nearly 18,000 days (based on adjusting 2002 air quality) upon meeting the current 8-hr standard and these estimates are reduced to 9,000 (based on adjusting 2004 air quality) to about 13,000 (based on adjusting 2002 air quality) upon meeting a 0.064 ppm, 4th-highest daily maximum 8-hr average standard. Confidence intervals, based on statistical uncertainty reflecting sample size considerations for incidence and percent of total incidence are shown in Tables 5C-7 through 5C-10 (Appendix 5C) based on adjusting 2004 and 2002 air quality.

Figure 5-12 summarizes unscheduled hospital admission risk estimates for respiratory illness and asthma in New York City associated with short-term exposures to O_3 concentrations in excess of background levels from April through September under recent air quality and when the current and alternative 8-hr standards are just met based on adjusting 2004 and 2002 air quality data, respectively. For total respiratory illness, Figure 5-12 shows about 6.4 cases per 100,000 relevant population, which represents 1.5% of total incidence or 513 cases when 2002 O_3 levels are adjusted to just meet the current 8-hr standard. For asthma-related hospital admissions, which are a subset of total respiratory illness admissions, the estimates are about 5.5 cases per 100,000 relevant population, which represents about 3.3% of total incidence or 438 cases for this same air quality scenario. For increasingly more stringent alternative 8-hr standards, Figure 5-12 shows a gradual reduction in respiratory illness cases per 100,000 relevant population from 6.4 cases per 100,000 upon just meeting the current 8-hr standard to 4.6 cases per 100,000 under the most stringent 8-hr standard (i.e., 0.064 ppm, average 4th-highest daily maximum) analyzed. The comparable estimates based on adjusting 2004 air quality are shown in Figure 5C-6 (Appendix 5C) and are somewhat higher, but show a similar pattern of gradual reduction. Confidence intervals, based on statistical uncertainty reflecting sample size considerations for incidence, incidence per 100,000 relevant population, and percent of total incidence are shown in Tables 5C-11 and 5C-12 (Appendix 5C) based on adjusting 2004 and 2002 air quality data to just meet the current and potential alternative standards.

Additional respiratory-related hospital admission estimates for three other locations are provided in the Risk Assessment TSD. We note that the concentration-response functions for each of these locations examined different outcomes in different age groups (e.g., > age 30 in Los Angeles, >age 64 in Cleveland and Detroit, vs. all ages in New York City), making comparison of the risk estimates across the areas very difficult. For hospital admissions in Detroit, none of the estimates were statistically significant and the median estimates were

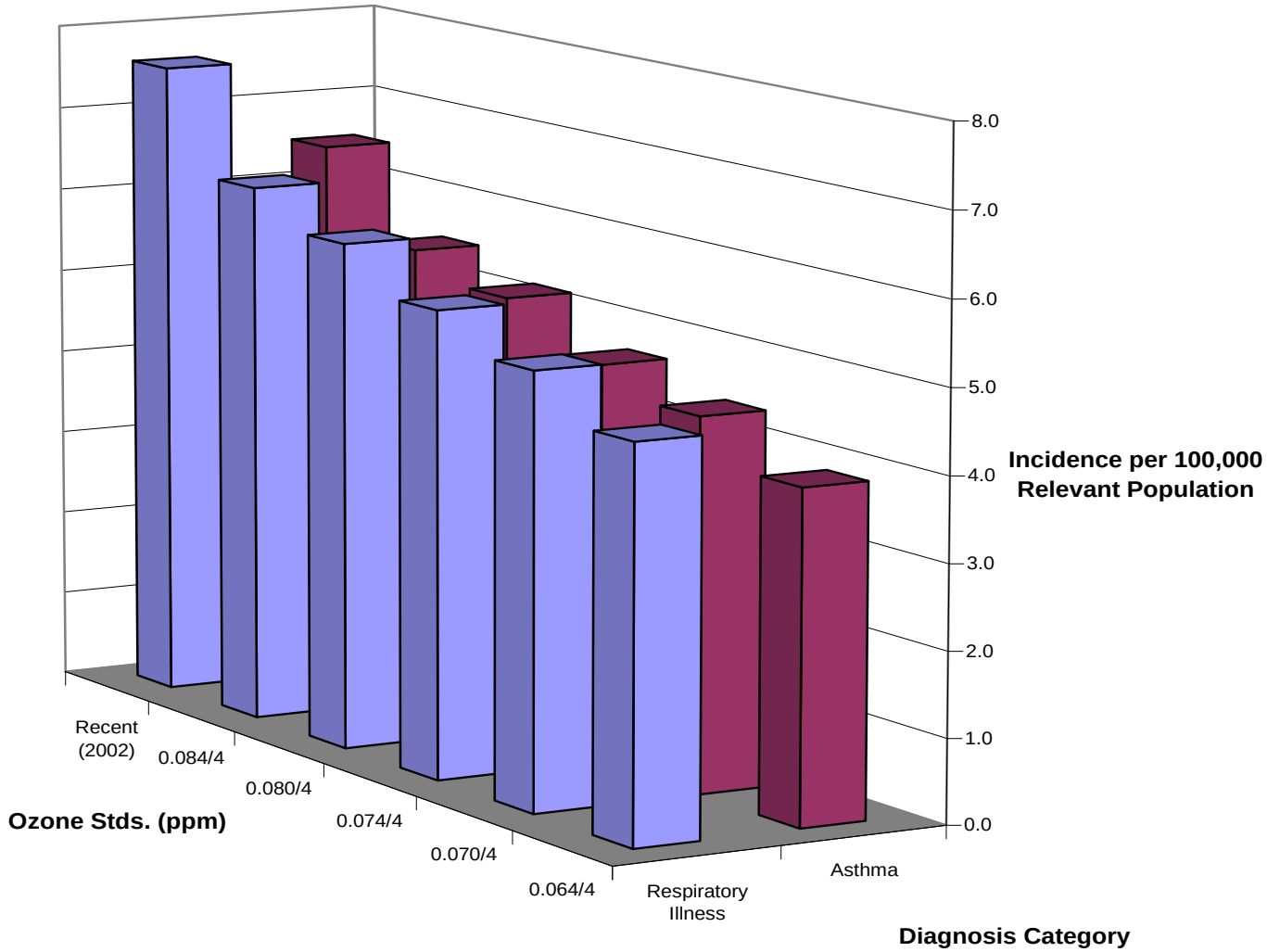
Figure 5-11. Estimated Symptom-Days for Chest Tightness Among Moderate/Severe Asthmatic Children (Ages 0 – 12) in Boston Associated with Recent (April-September 2002) O_3 Levels and with Levels Just Meeting Alternative Average 4th-Highest Daily Maximum 8-Hour Ozone Standards* (Based on Gent et al., 2003)



*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. The 4th-highest daily maximum standards, denoted m/4, require that the average of the 3 annual nth daily maxima over a 3-year period be at or below the specified level. 95% confidence intervals associated with these risk estimates are presented in Table 5C-9 in Appendix 5C.

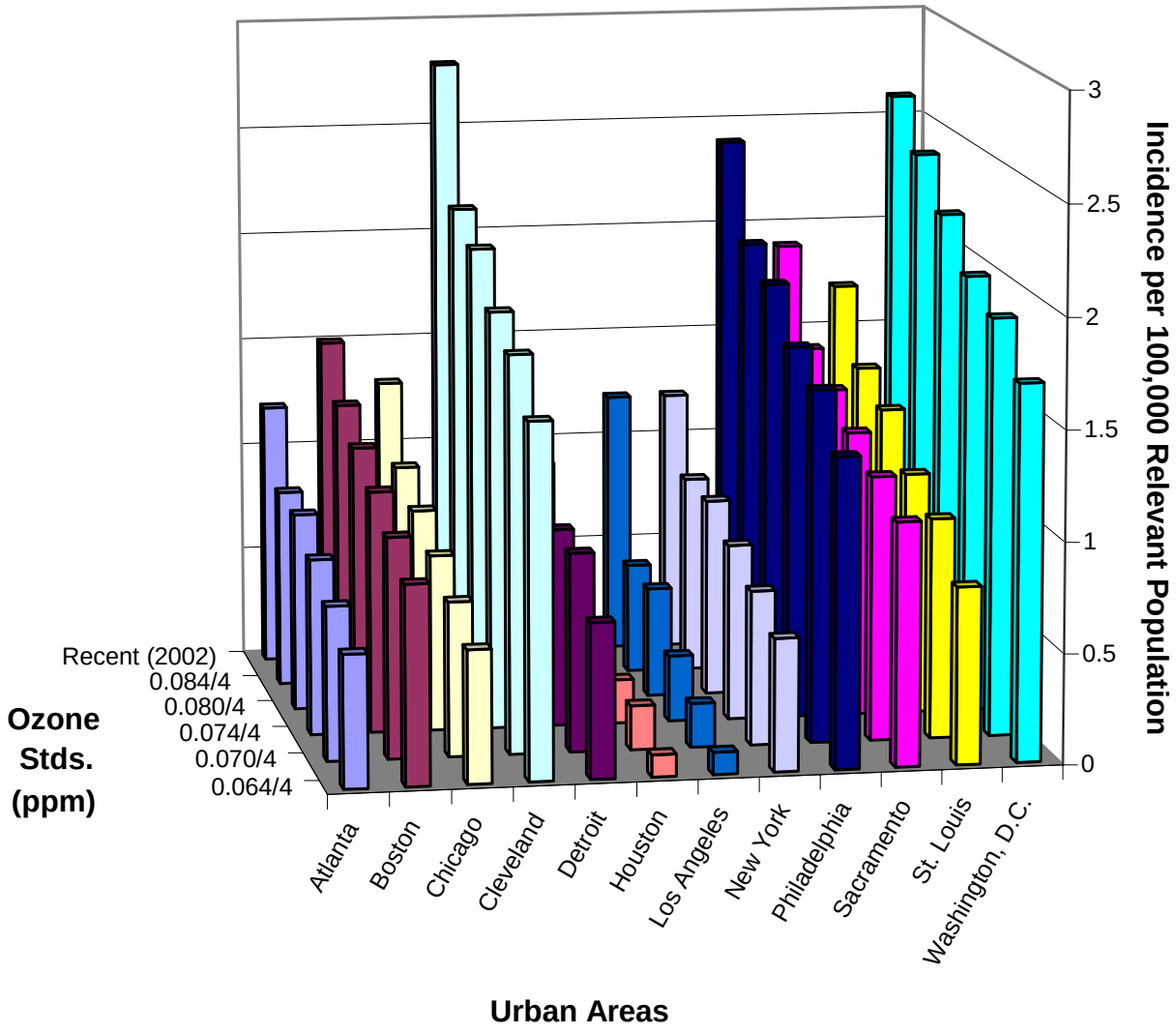
Figure 5-12. Estimated Incidence of (Unscheduled) Respiratory Hospital Admissions per 100,000 Relevant Population in New York Associated with Recent (April – September, 2002) O_3 Levels and with O_3 Levels Just Meeting Alternative Average 4th-Highest Daily Maximum 8-Hour Standards

(based on Thurston et al., 1992)



*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. The 4th-highest daily maximum standards, denoted m/4, require that the average of the 3 annual nth daily maxima over a 3-year period be at or below the specified level. 95% confidence intervals associated with these risk estimates are provided in Table 5C-12 in Appendix 5C.

Figure 5-13. Estimated Incidence of Non-Accidental Mortality per 100,000 Relevant Population Associated with Recent Air Quality (2002) and with Just Meeting Alternative Average 4th-Highest Daily Maximum 8-Hour Ozone Standards (Using Bell et al., 2004 – 95 U.S. Cities Function), Based on 2002 Ozone Concentrations



*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. The 4th-highest daily maximum standards, denoted m/4, require that the average of the 3 annual nth daily maxima over a 3-year period be at or below the specified level. 95% confidence intervals associated with these risk estimates are provided in Figure 5-14.

negative for 0- and 1-day lags and small but positive for 2- and 3-day lags for COPD-related and pneumonia hospital admissions.

Figure 5-13 summarizes the results of the assessment of the reduced non-accidental mortality risks associated with O_3 concentrations above background that just meet the current and several potential alternative 8-hr daily maximum standards across the 12 urban areas for air quality adjusted based on 2002 air quality data. This figure shows the annual median risk estimates for recent air quality and for just meeting alternative 8-hr standards based on the O_3 coefficients estimated in the studies based on adjusting 2002 air quality data. Ranges reflecting the statistical uncertainty, taking into account sample size considerations, based either on the 95 percent confidence intervals around those estimates (if the coefficients were estimated using classical statistical techniques) or on the 95 percent credible intervals (if the coefficients were estimated using Bayesian statistical techniques) are presented in Tables 5C-13 through 5C-16 (Appendix 5C) and in the Risk Assessment TSD. The risk estimates in this figure are based on the 95-city function reported in Bell et al. (2004) for non-accidental mortality. Additional risk estimates for cardiorespiratory mortality are included in the Risk Assessment TSD for 8 of the 12 urban areas. Also, Figure 5C-7 (Appendix 5C) shows comparable risk estimates based on adjusting 2004 air quality data.

Figure 5-14 shows the median estimates and 95% credible intervals for each of the 12 urban areas for non-accidental mortality based on the 95-cities concentration-response function in Bell et al. (2004) for 2002 air quality data and just meeting alternative standards based on adjusting 2002 air quality data. Figure 5C-8 (Appendix 5C) presents the comparable figure for 2004 air quality and just meeting alternative standards based on adjusting 2004 air quality data. For example, Figure 5-14 shows a median risk estimate associated with just meeting the current 8-hr standard for non-accidental mortality in Atlanta is around 0.9% of total incidence and the 95% credible interval is about 0.3% to about 1.5% of total incidence. While the 95% credible intervals get progressively smaller as one considers more stringent standards, as discussed previously these credible intervals do not consider overall model uncertainty (e.g., whether or not the shape of the concentration-response relationship is best represented by a log linear relationship versus a more sigmoidal shape, particularly at lower O_3 concentration levels).

The results in this portion of the risk assessment across the 12 urban areas follow the same patterns as the results discussed in section 5.4.1 for risks associated with recent year O_3 concentrations, because they are largely driven by the same concentration-response function coefficient estimates and confidence or credible intervals. While there is a noticeable reduction in the median risk estimates in some of the urban areas between that associated with a recent year of air quality and just meeting the current 8-hr standard, the reductions associated with progressively more stringent alternative 8-hr standards are more modest. Based on adjusting

Figure 5-14. Annual Warm Season (April to September) Estimated O_3-Related Non-Accidental Mortality Associated with Recent (2002) O_3 Levels and Levels Just Meeting Alternative 8-hr O_3 Standards (Using Bell et al., 2004 – 95 U.S. Cities Function)

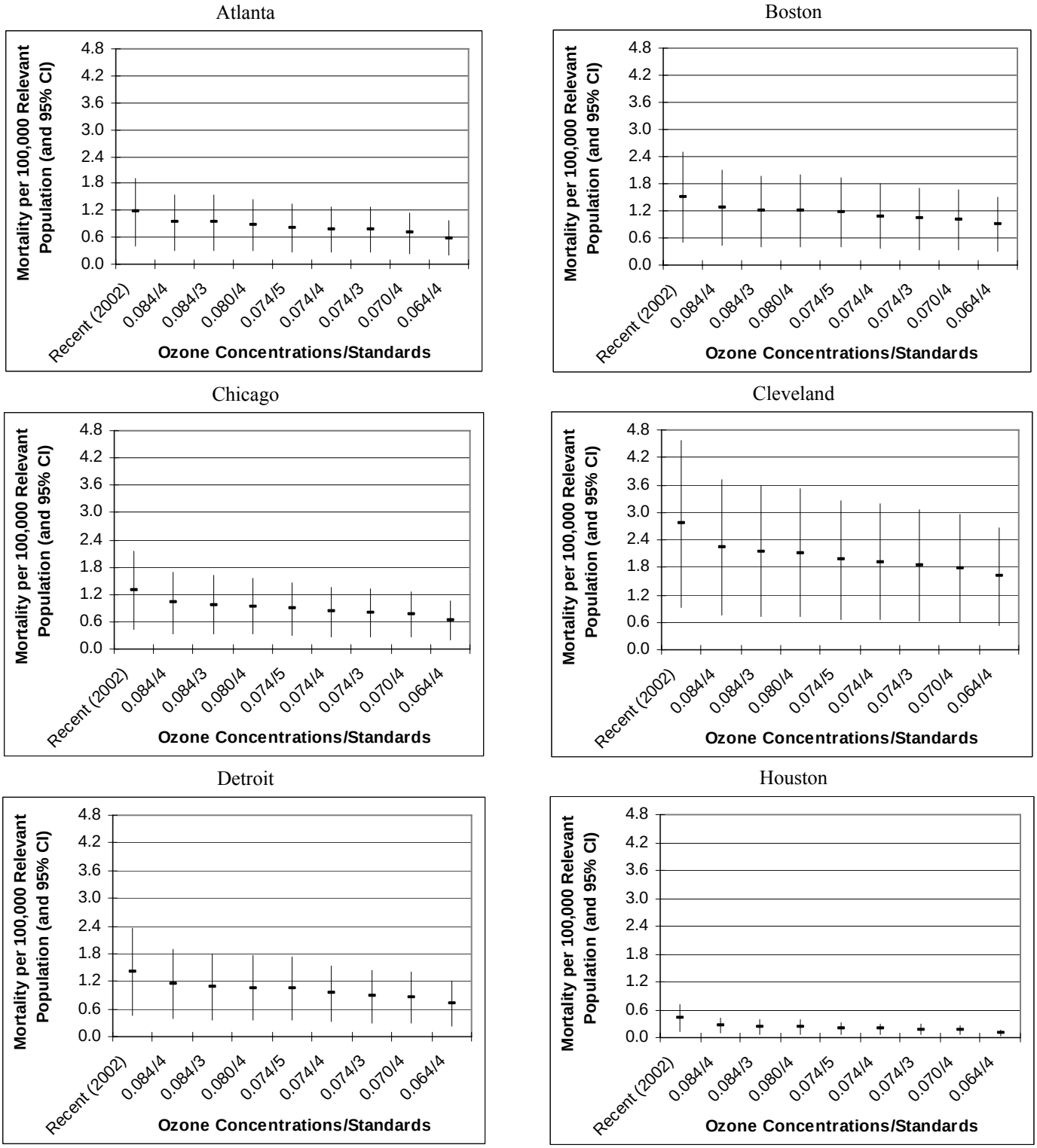
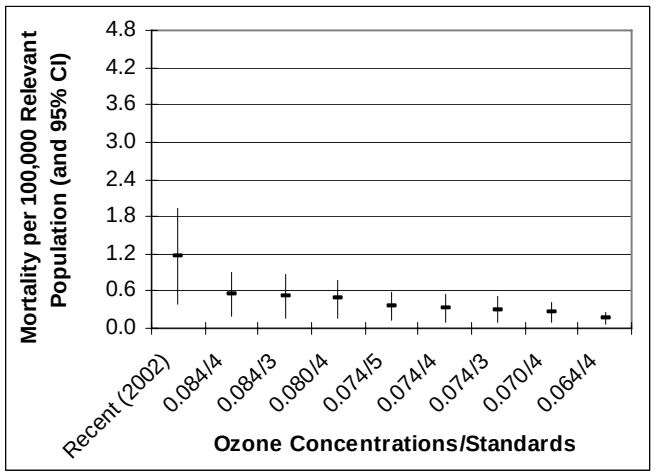
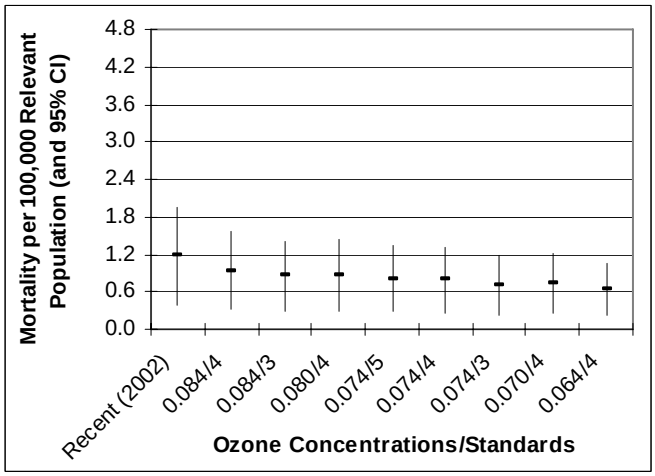


Figure 5-14 (continued)

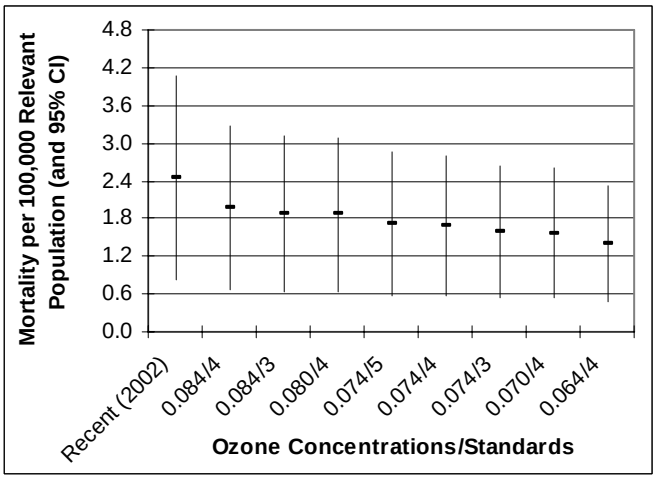
Los Angeles



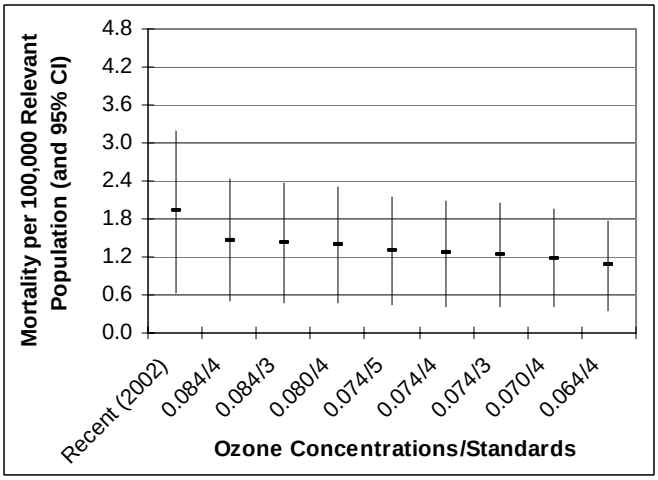
New York



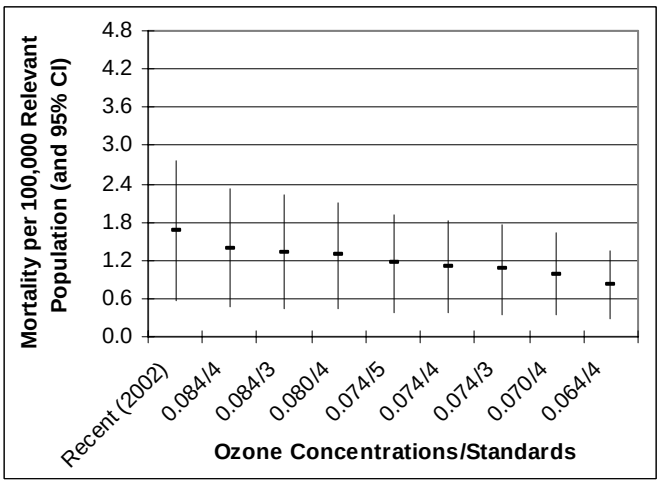
Philadelphia



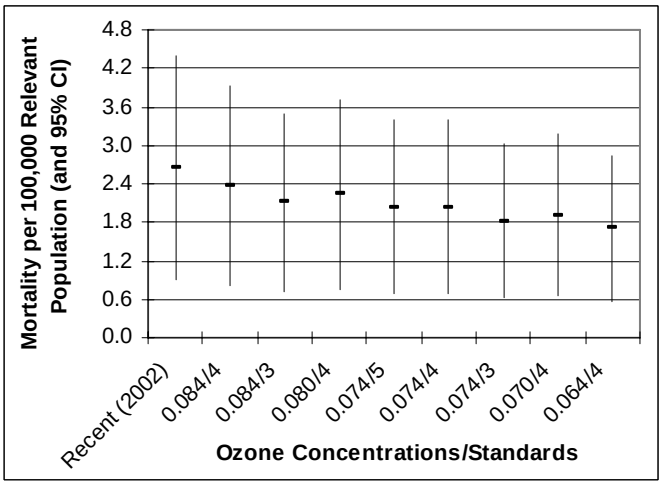
Sacramento



St. Louis



Washington, D.C.



*An 8-hr average standard, denoted m/n is characterized by a concentration of m ppm and an nth daily maximum. So, for example, the current standard is 0.084/4 – 0.084 ppm, 4th-highest daily maximum 8-hr average. Percents are median (0.5 fractile) percents of children. The bars are 95% credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

2002 air quality data, the range of median estimates associated with O_3 upon just meeting the current standard is about 0.3 to 2.4 cases per hundred thousand relevant population across the 12 urban areas and this range is reduced to about 0.1 to 1.7 cases per 100,000 relevant population upon just meeting the most stringent alternative standard analyzed (0.064 ppm, average 4th-highest daily maximum 8-hr average) We also note that the risk estimates expressed in terms of incidence per 100,000 population are noticeably smaller for Houston based on both 2002 and 2004 air quality data and for Los Angeles based on 2002 air quality, especially upon just meeting the current or alternative 8-hr standards than the other urban areas. The risk estimates are notably higher in most of the urban areas for 2002 air quality data and air quality data simulated to just meet the current and alternative standards based on adjusting 2002 data.

As shown in Table 5C-13 through 5C-16 in Appendix 5C of this chapter, estimated O_3-related (non-accidental) mortality reported by Schwartz (2004) for Chicago, Detroit, and Houston, based on both the single-city and the multi-city concentration-response functions, tend to be higher than the Bell et al. (2004) estimates in those locations in large part because Schwartz used the 1-hr maximum O_3 concentration, rather than the 24-hr average, as the exposure metric. The changes from 1-hr maximum O_3 concentrations that just meet the current 8-hr O_3 standard to background 1-hr maximum O_3 concentrations were generally larger in these assessment locations than the corresponding changes using the 24-hr average metric.

Figure 5-15a and b shows the estimated annual percent of non-accidental mortality mortality associated with short-term exposure to O_3 concentrations that just meet the current 8-hour daily maximum standard that fall within specified ranges. The pattern of results is similar to the pattern seen for recent year O_3 concentrations discussed in section 5.4.1. Using simulated O_3 concentrations that just meet the current 8-hour standard based on 2004 air quality data, all O_3-related non-accidental mortality was associated with O_3 concentrations less than 0.06 ppm, 24-hr average and most of that was associated with O_3 concentrations less than 0.04 ppm, 24-hr average. Using simulated O_3 concentrations that just meet the current 8-hour standard based on 2002 air quality data, all O_3-related non-accidental mortality was associated with O_3 concentrations less than 0.08 ppm, 24-hr average and the great majority was associated with O_3 concentrations less than 0.05 ppm, 24-hr average. The results for cardiorespiratory mortality follow a similar pattern. As discussed in section 5.4.1, scatter plots comparing 8-hr daily maximum concentrations at the highest monitor with the average of the 24-hr average over all monitors within an urban area were developed and are included in Appendix 5A.2 to provide some perspective on the 24-hr intervals shown. These scatter plots show that 8-hr daily maximum concentrations on average are roughly twice the observed 24-hr average levels, although there is considerable variability in this relationship from day-to-day within an urban area. There also is some variability in this relationship between 8-hr daily maximum and 24-hr average levels across the 12 urban areas.

Figure 5-15. Estimated Annual Percent of Non-Accidental Mortality Associated with Short-Term Exposure to O_3 Above Policy Relevant Background for the Period April – September When the Current 8-Hour Standard is Just Met (Based on Bell et al., 2004) – Total and Contribution of 24-Hour Average O_3 Ranges

Figure 5-15a. Based on Adjusting 2004 Air Quality Data

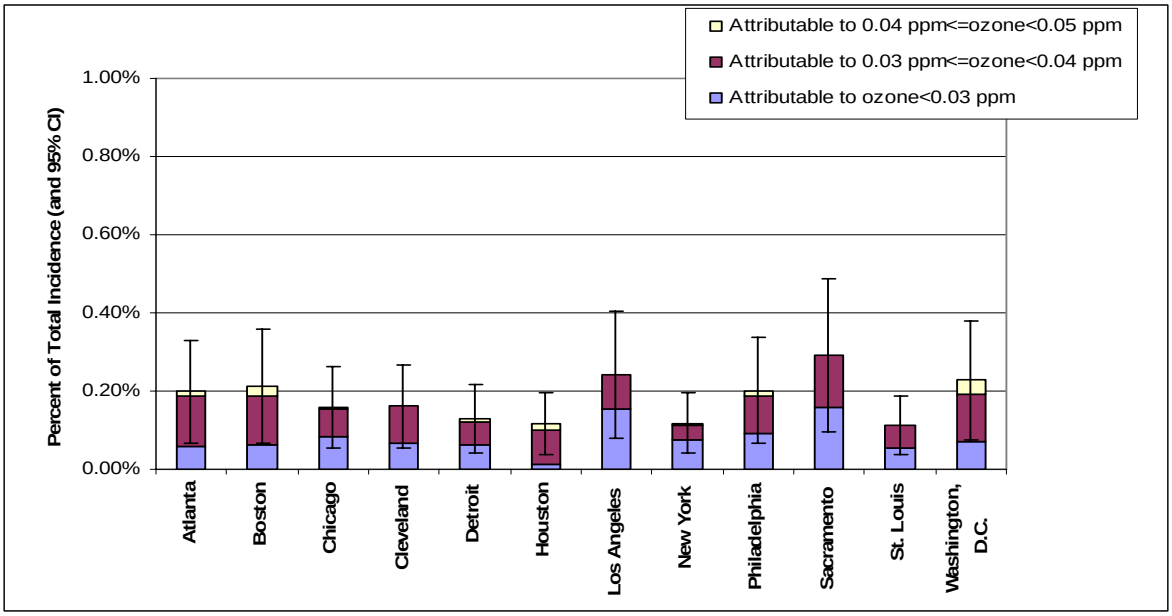
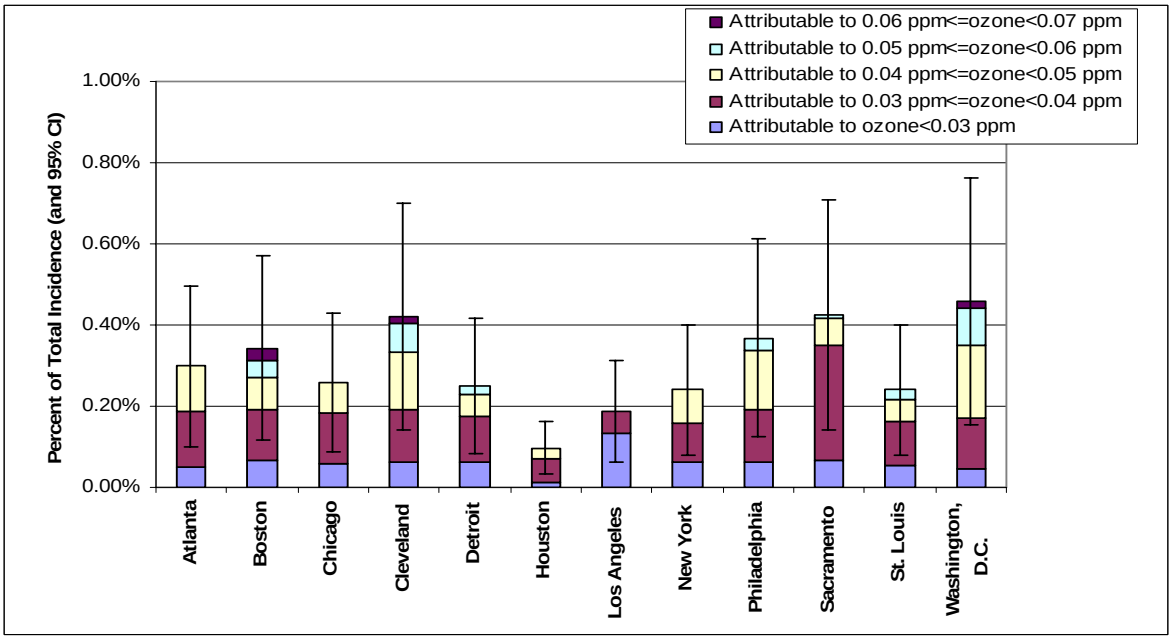


Figure 5-15b. Based on Adjusting 2002 Air Quality Data



*Note that as shown in scatter plots in Appendix 5A.2, 8-hr daily maximum concentrations at the highest monitor are roughly twice the level of the average of the 24-hr average O_3 concentrations over all monitors within an urban area which are used in this figure, although this ratio varies across areas.

5.4.3 Sensitivity Analyses

We have conducted sensitivity analyses examining the influence of alternative assumptions about background (i.e., PRB) levels on the lung function and mortality risk estimates and the impact of alternative assumptions about the shape of the lung function exposure-response relationship on the lung function health risk estimates. These sensitivity analyses were motivated by the discussion at the August 24-25, 2006 CASAC meeting and are presented below and in sections 3.3 and 4.3 of the Risk Assessment TSD.

Reflecting the discussion at the August CASAC meeting, the CASAC panel suggested (Henderson, 2006c) that one approach to deal with the uncertainties surrounding estimation of PRB levels would be to assess the change in total risk (i.e., all O_3-related risks above 0 ppm) associated with alternative standards relative to the total risks associated with just meeting the current standard, without subtracting estimated risks associated with PRB levels from either estimate. As shown in Tables 4-48 and 4-49 of the Risk Assessment TSD, where non-accidental mortality risks associated with O_3 were estimated for all days above 0 ppm for two recent years of air quality (2002 and 2004), the largest part of the total risk was related to levels between 0 and our baseline estimated PRB levels. Adopting the approach suggested by CASAC would place emphasis on the region of the concentration-response relationship (i.e., at levels below 0.035 ppm down to 0 ppm) where there is the greatest uncertainty about whether effects occur. In addition, the approach suggested by CASAC only addresses risks relative to the current standard and does not address the risk remaining upon meeting the current and alternative standards, which is an important consideration in setting a NAAQS. For assessing risks remaining upon just meeting a standard, EPA has decided as a matter of policy that only risks in excess of PRB are relevant to the decision, and thus staff judges it is still appropriate to estimate risks in excess of estimated PRB levels.

As discussed below, we have examined the change in lung function risk estimates associated with alternative standards relative to the current standard and have found that these estimates are generally insensitive to alternative assumptions about PRB. Similarly, changes in non-accidental mortality risk estimates for just meeting alternative standards relative to the current standard also are less sensitive to assumptions about the levels used to represent PRB. We recognize that the lung function and non-accidental mortality risk estimates remaining upon just meeting the current and alternative standards are impacted to varying degrees by the assumptions about PRB levels depending on area, year of air quality, and health endpoint.

5.4.3.1 Impact of Alternative Assumptions About Background

Risk estimates associated with O_3 concentrations discussed in this chapter and in the Risk Assessment TSD have been developed– either based on O_3 concentrations from a recent year of

air quality or O_3 concentrations "rolled back" to just meet a standard – above PRB. We selected three locations – Atlanta, Los Angeles, and New York – for a sensitivity analysis for lung function responses, and calculated lung function responses using (1) the original PRB estimates, (2) lower PRB estimates for each location, and (3) higher PRB estimates for each location. We also conducted a sensitivity analysis for non-accidental mortality associated with O_3 exposure for all 12 urban areas. For all of the urban areas, except Atlanta, the lower PRB estimates were calculated by subtracting 5 ppb from the original PRB estimates; for Atlanta, the lower PRB estimates were calculated by subtracting 10 ppb from the original PRB estimates. In all locations, the higher PRB estimates were calculated by adding 5 ppb to the original PRB estimates.[16]

The lung function sensitivity analyses for alternative estimates of PRB were run for all school age children, with response defined as a decrement in $FEV_1 \geq 15\%$, and for asthmatic school age children, with lung function response defined as a decrement in $FEV_1 \geq 10\%$. Table 5C-17 shows the results of this sensitivity analysis for all school age children in terms of the number of children estimated to experience a lung function response of concern (i.e., $FEV_1 \geq 15\%$) based on adjusting 2002 and 2004 air quality to just meet the current and two alternative 8-hr standards. Additional tables showing the sensitivity analysis results for total occurrences of this lung function response for all children are included in the Risk Assessment TSD (see section 3.3.1). The sensitivity analysis results for lung function responses in asthmatic children are similar in pattern and also are presented in the Risk Assessment TSD (see section 3.3.1).

The impact of alternative lower and higher assumed PRB levels on lung function responses in all school age children (with responses defined as a decrement in $FEV_1 \geq 15\%$,) was relatively small, generally much less than +/- 3%. Assuming lower PRB levels increased the estimated number of children with a response, while assuming higher PRB levels decreased the estimated number of children with a response. In terms of total occurrences of moderate lung function responses, different assumptions about PRB had a somewhat larger impact, but the impact was still generally less than about +/- 10% relative to our base case assumption for PRB.

Figure 5-16 shows the impact of lower and higher PRB assumed levels in terms of the relative percent changes in non-accidental mortality risk from the current 8-hr standard based on adjusting 2002 air quality data in each of the 12 urban areas. One observes that for most, but not all of the locations, the general pattern is not significantly impacted by the choice of PRB assumptions. The

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[16]Summarizing its assessment of the validity of the GEOS-CHEM model, the O_3 CD states, "in conclusion, we estimate that the PRB ozone values reported by Fiore et al. (2003) for afternoon surface air over the United States are likely 10 ppbv too high in the southeast in summer, and accurate within 5 ppbv in other regions and seasons." These error estimates are based on comparison of model output with observations for conditions that most nearly reflect those given in the PRB definition, i.e., at the lower end of the probability distribution.

choice of PRB has a somewhat greater impact on the non-accidental mortality based on adjusting 2004 air quality data (see Figure 4-19 in the Risk Assessment TSD), which is likely due to the significantly lower O_3 levels associated with just meeting the alternative standards based on this year of air quality data.

Results of the PRB sensitivity analysis for non-accidental mortality associated with O_3 exposures expressed in terms of absolute estimates are presented in Table 5C-18 for 2002 air quality adjusted to just meet the current standard. Additional sensitivity analysis results for recent air quality (both 2002 and 2004 air quality) and for 2004 and 2002 air quality adjusted to just meet the current and two alternative standards are included in the Risk Assessment TSD (see section 4.3 and Appendix I). Table 5C-18 illustrates the impact of alternative assumed PRB levels on incidence of O_3-related non-accidental mortality per 100,000 population. Lower and higher assumed PRB levels generally resulted in increased and decreased, respectively, estimates in the incidence of mortality per 100,000 population. As shown in this table, estimates assuming lower PRB levels results in increased estimates of non-accidental mortality incidence per 100,000 that are often 50 to 100% greater than the base case estimates. Similarly, estimates assuming higher PRB levels results in decreased estimates of non-accidental mortality incidence per 100,000 that are 50% or greater less than the base case estimates.

As discussed in section 4.3 of the Risk Assessment TSD, because O_3 concentrations just meeting the current standard are substantially lower than O_3 concentrations observed for the recent years of air quality (for most of the urban areas), the change in the assumed PRB levels had a greater impact on the estimates of mortality associated with levels just meeting the current standard, in terms of percent change in the estimate. Similarly, changing the estimates of PRB tended to have progressively greater impacts on the estimates of mortality risk as progressively more stringent standards were considered. Not surprisingly, assumptions about PRB have a greater impact on risk estimates associated with the most stringent standard examined, since a greater percentage of days is impacted, in terms of being classified as above or below PRB, by the assumptions concerning PRB levels.

5.4.3.2 Impact of Alternative Assumptions About the Shape of Exposure-Response Relationships for Lung Function Decrements

As described in section 5.3.1.3, the exposure-response functions used in the primary analyses are based on the assumption that the relationship between exposure and response has a

Figure 5-16. Sensitivity Analysis of Estimated Percent Change in O_3-related Non-Accidental Mortality (Using Bell et al., 2004 – 95 Cities) From the Current Standard to Alternative 8-hr Standards and a Recent Year of Air Quality, Using Base Case, Higher, and Lower PRB Estimates*

Figure 5-16a. Based on Adjusting 2004 O_3 Concentrations

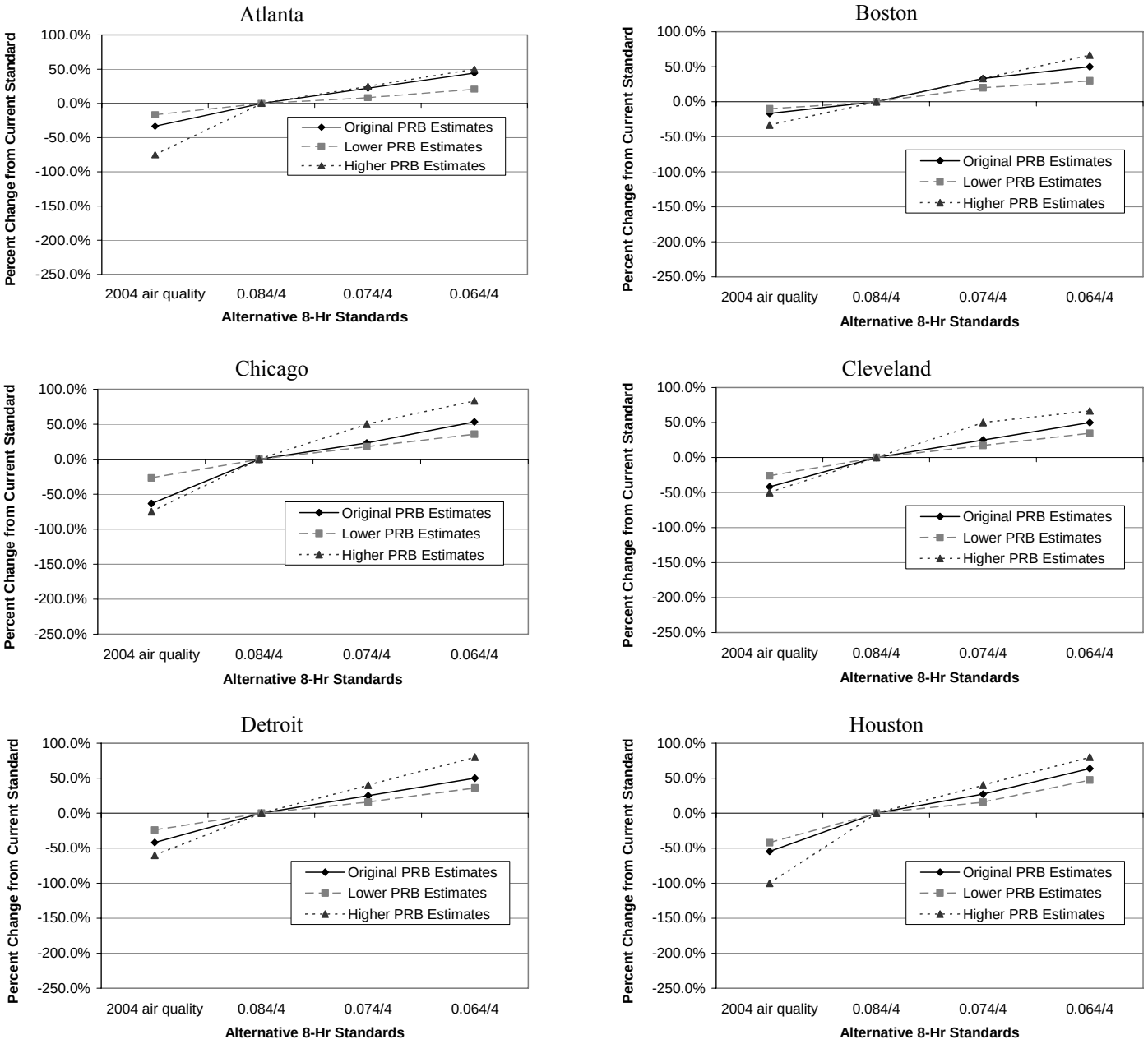
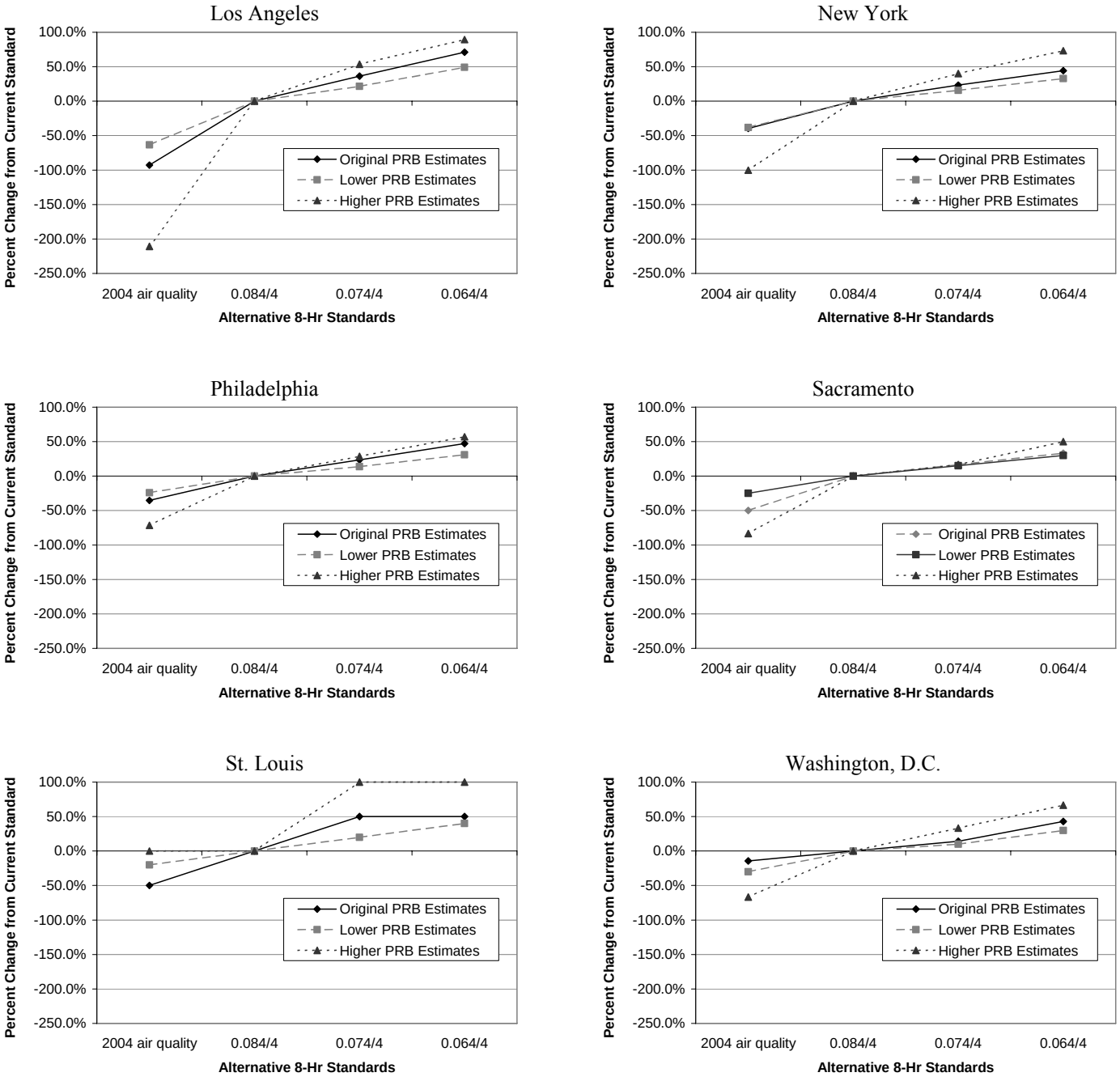


Figure 5-16a ccntinued



*The 8-hr average standards shown in these figures, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum. So, for example, the current standard is 0.084/4 -- 0.084 ppm, 4th-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality.

Figure 5-16b. Based on Adjusting 2002 O_3 Concentrations

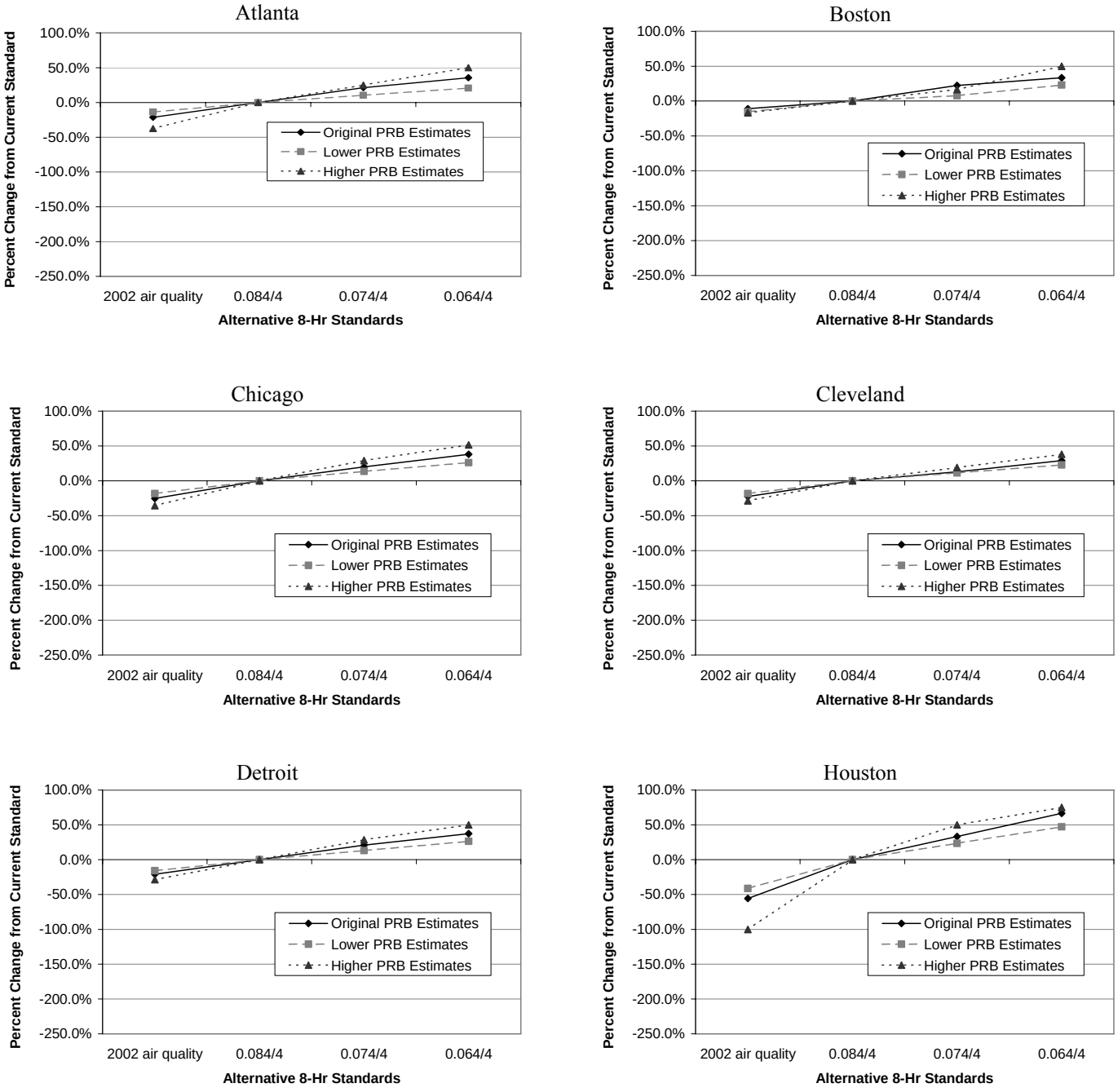
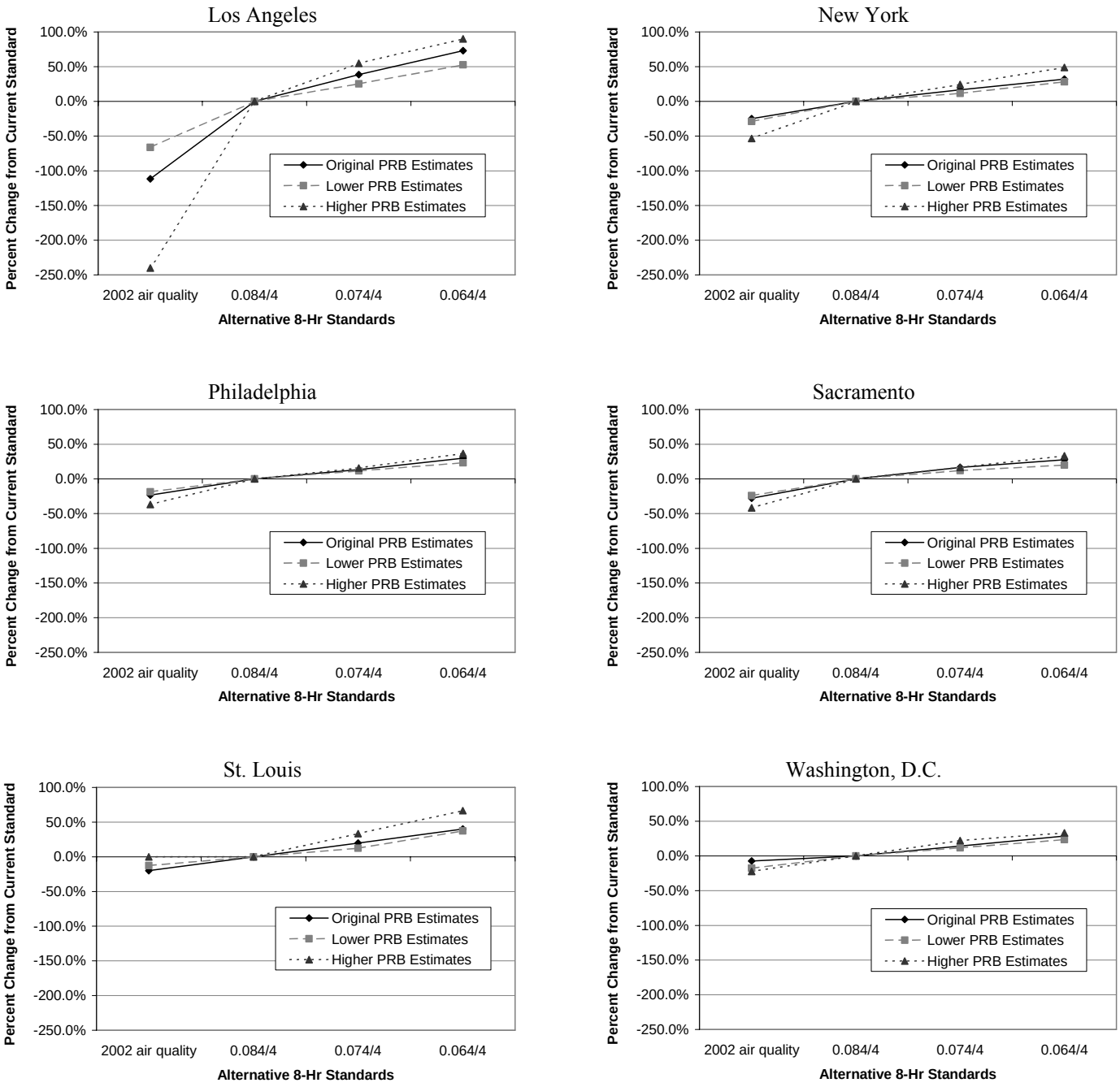


Figure 5-16b continued



*The 8-hr average standards shown in these figures, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum. So, for example, the current standard is 0.084/4 -- 0.084 ppm, 4th-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality.

logistic form with 90 percent probability and a linear (hockeystick) form with 10 percent probability. We have conducted sensitivity analyses which examine the influence of two alternative assumptions about the exposure-response functions for lung function decrements in all and asthmatic school age children. These alternative exposure-response functions are based on an 80 percent logistic/20 percent linear split and a 50 percent logistic/50 percent linear split, in five locations – Atlanta, Chicago, Houston, Los Angeles, and New York. Appendix C of the Risk Assessment TSD presents tables showing the results of the sensitivity analysis for a change in $FEV_1 \geq 15\%$ for all school age children for a recent year of air quality as well as when O_3 concentrations just meet each of three 4th-highest daily maximum standards – 0.084/4, 0.074/4 and 0.064/4, based on adjusting 2004 and 2002 data, respectively. Appendix C also shows the corresponding impacts on the estimated number of asthmatic school age children experiencing at least one lung function response, defined as a change in $FEV_1 \geq 10\%$.

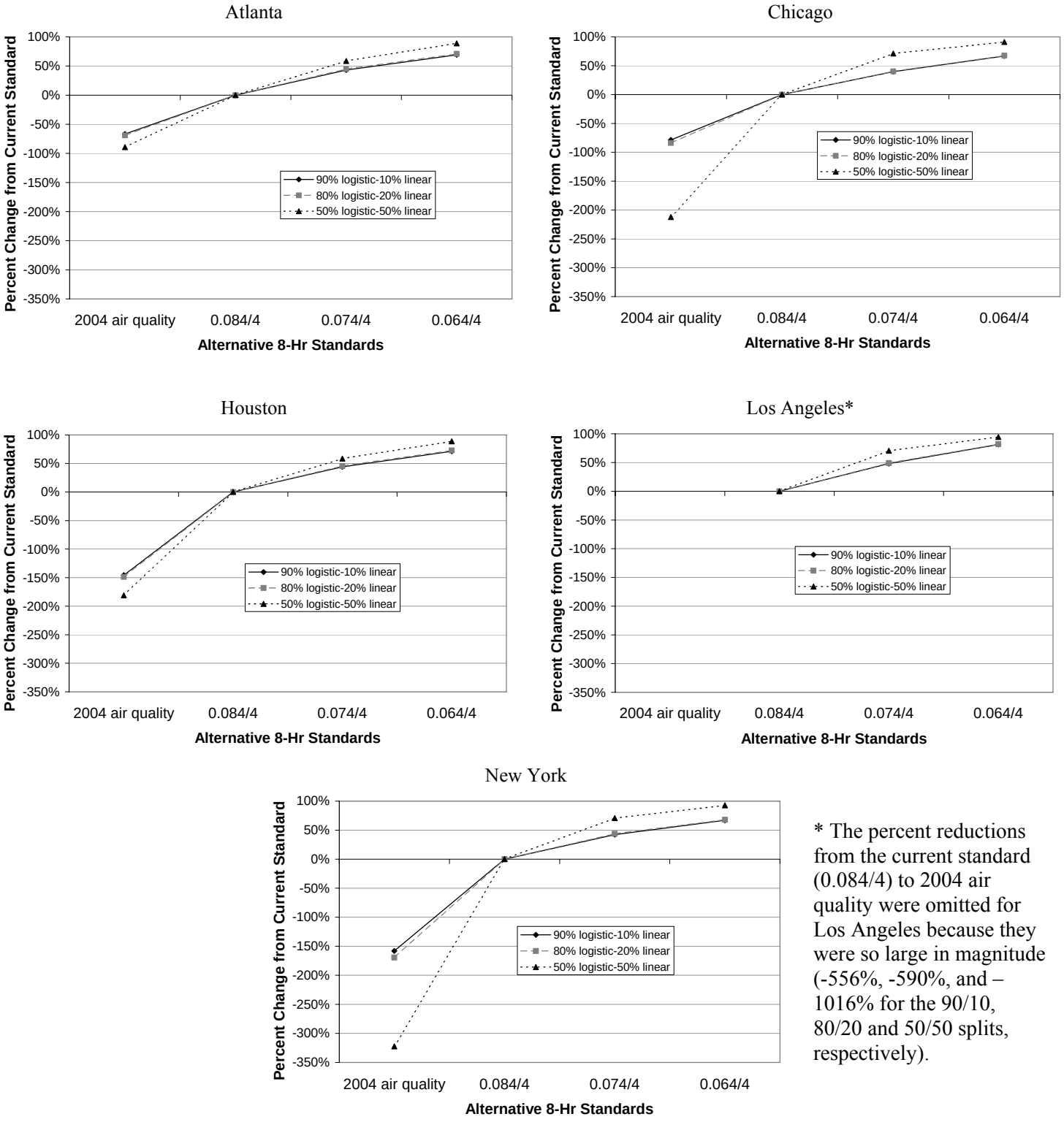
Figures 5-17a and b and 5-18a and b show the impacts of alternative estimates of exposure-response functions on estimated percent changes in response among all school age children and asthmatic school age children, respectively, when O_3 concentrations are changed from those just meeting the current standard to a recent year of air quality (results are shown based on both 2004 and 2002 air quality) and to those just meeting each of the two alternative standards given above. A positive percent change reflects a reduction in risk relative to just meeting the current 8-hr standard. Since the comparisons are with respect to just meeting the current standard, the percent changes for the recent year of air quality are negative (i.e., going from just meeting the current standard to 2002 or 2004 air quality represents an increase in risk). The impacts of changing the functional form of the exposure-response relationship varied substantially. Changing from the 90 percent logistic/10 percent linear base case to the 80%/20% split generally had only a small impact for all and asthmatic school age children, with most risk estimates being within 5% of the base case estimates. The impacts of changing from the base case to the 50% logistic/50% linear case were generally (although not always) larger. We observed greater changes for all and asthmatic school age children between the 50/50 split and the base case in terms of percent change in risk for the two more stringent alternative standards relative to the current standard

5.4.4 Comparison with Risk Estimates from Prior Review

As noted in section 5.1.1, EPA conducted a health risk assessment during the prior O_3 NAAQS review. We recognize that two of the health endpoints, lung function (FEV_1) decrements for children and respiratory-related and asthma hospital admissions were included in

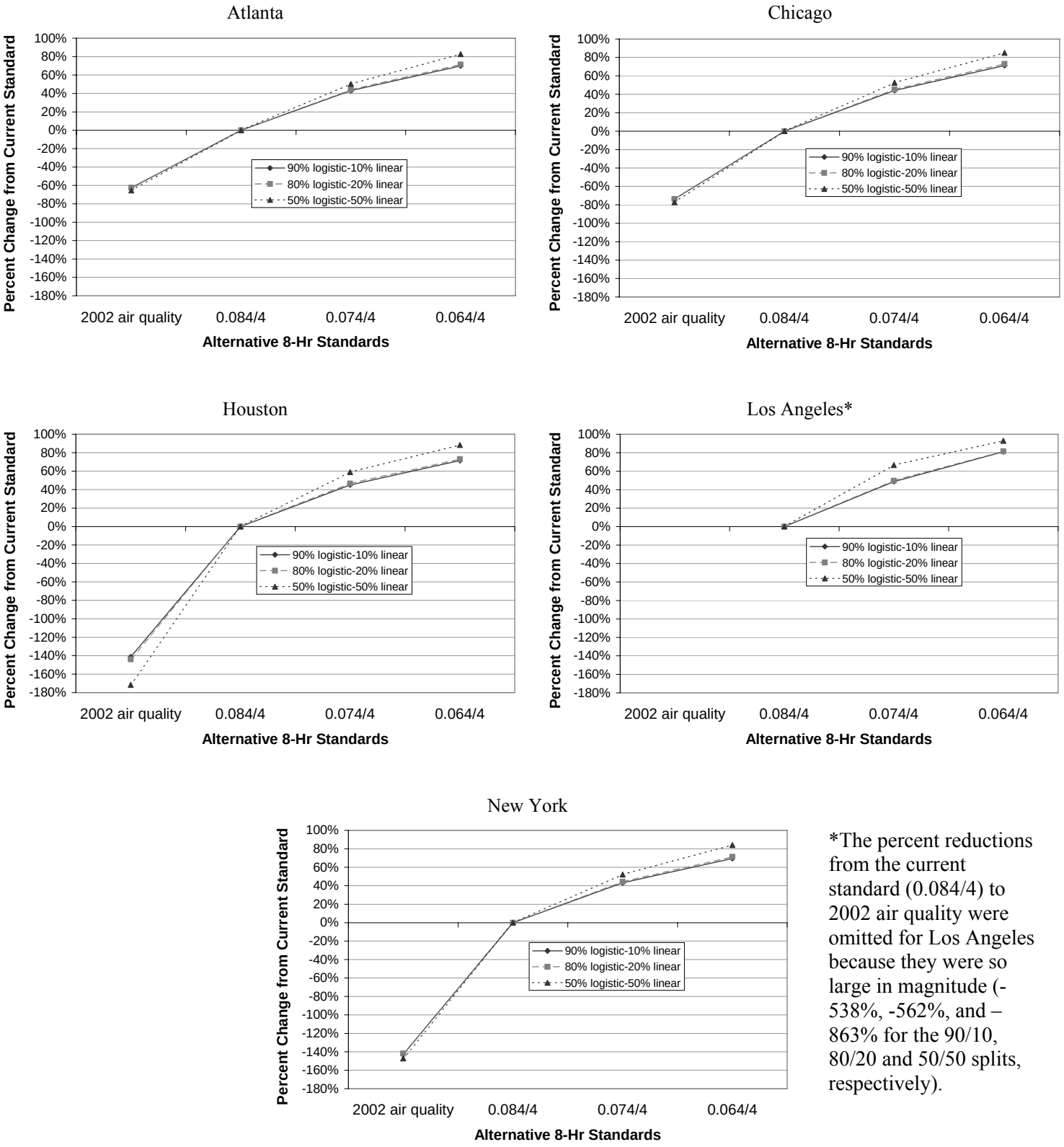
Figure 5-17. Sensitivity Analysis: Impact of Alternative Estimates of Exposure-Response Function on Estimated Percent Changes From the Current Standard in Numbers of All Children (Ages 5-18) Engaged in Moderate Exertion Experiencing at Least One Decrement in FEV_1 ≥15%

Figure 5-17a. Based on Adjusting 2004 O_3 Concentrations



* The percent reductions from the current standard (0.084/4) to 2004 air quality were omitted for Los Angeles because they were so large in magnitude (-556%, -590%, and –1016% for the 90/10, 80/20 and 50/50 splits, respectively).

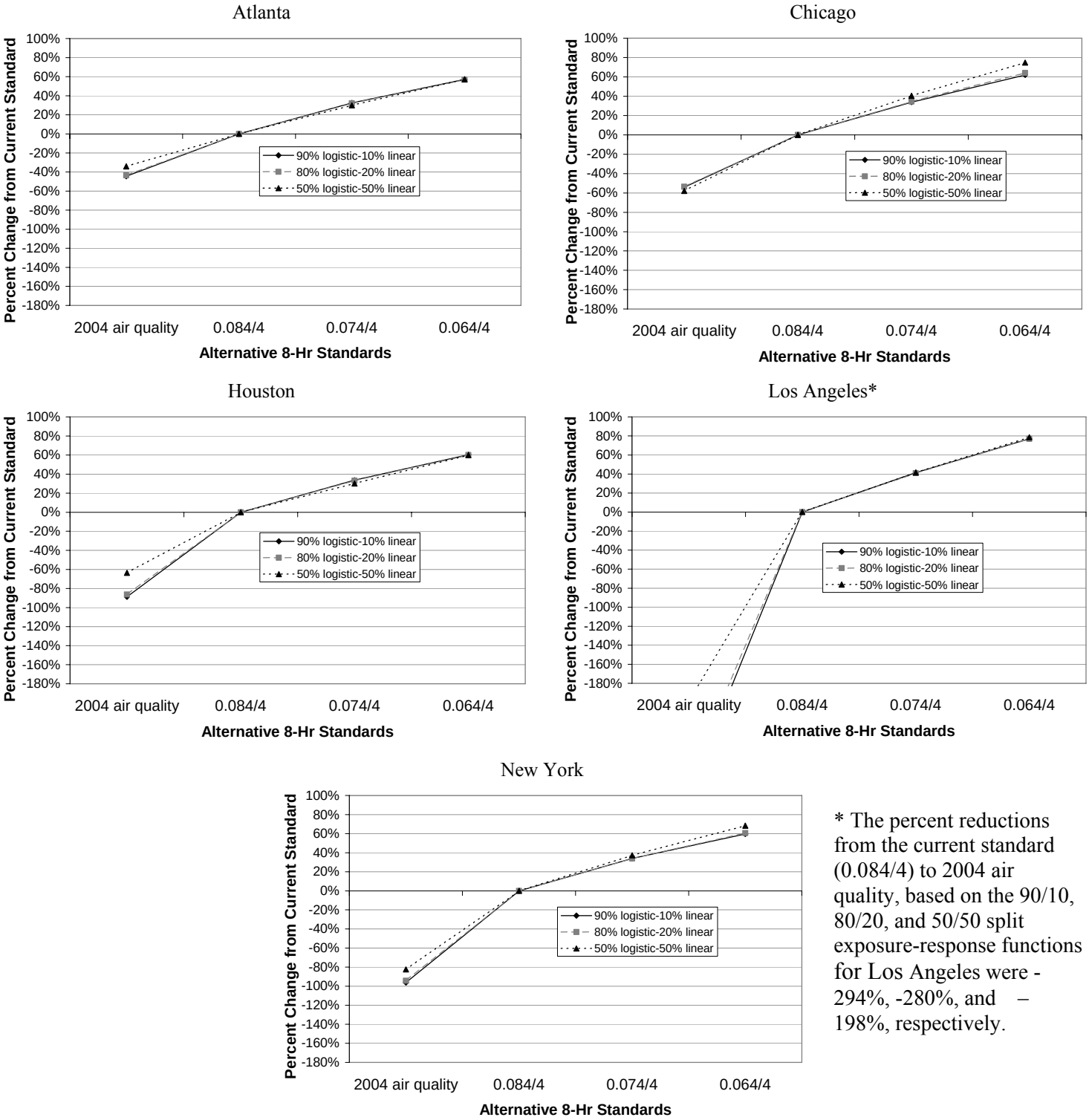
Figure 5-17b. Based on Adjusting 2002 O_3 Concentrations



*The percent reductions from the current standard (0.084/4) to 2002 air quality were omitted for Los Angeles because they were so large in magnitude (-538%, -562%, and –863% for the 90/10, 80/20 and 50/50 splits, respectively).

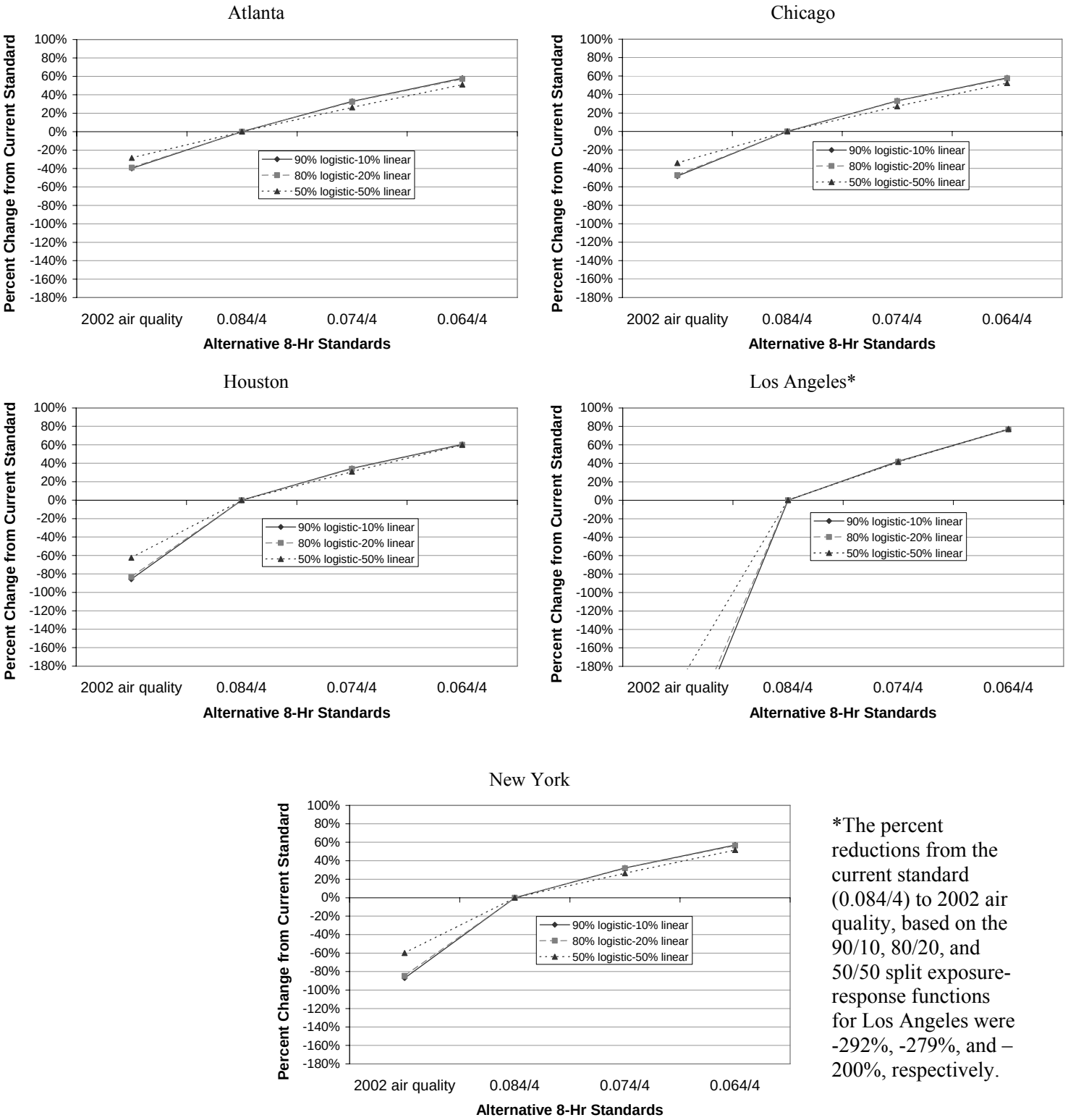
Figure 5-18. Sensitivity Analysis: Impact of Alternative Estimates of Exposure-Response Function on Estimated Percent Changes From the Current Standard in Numbers of Asthmatic Children (Ages 5-18) Engaged in Moderate Exertion Experiencing at Least One Decrement in $FEV_1 \geq 10\%$

Figure 5-18a. Based on Adjusting 2004 O_3 Concentrations



* The percent reductions from the current standard (0.084/4) to 2004 air quality, based on the 90/10, 80/20, and 50/50 split exposure-response functions for Los Angeles were -294%, -280%, and –198%, respectively.

Figure 5-18b. Based on Adjusting 2002 O_3 Concentrations



*The percent reductions from the current standard (0.084/4) to 2002 air quality, based on the 90/10, 80/20, and 50/50 split exposure-response functions for Los Angeles were -292%, -279%, and –200%, respectively.

the current and prior reviews. The other health endpoints included in the current risk assessment, lung function decrements in asthmatic children, respiratory symptoms in moderate/severe asthmatic children, and non-accidental and cardiorespiratory mortality are based on more recent scientific studies and, were not included in the prior review.

The lung function risk estimates developed for the current and prior review are based on exposure distributions generated by running O_3 exposure models and exposure-response relationships developed using the available controlled human exposure studies data. There have been significant changes in the exposure model between the prior and current review. As discussed in Chapter 4, no direct comparison is being made of the differences in exposure estimates for children engaged in moderate exertion associated with just meeting the current 8-hr standard between the past and current reviews. This is due to differences in the exposure model, differences in the population coverage within the urban areas included in the analyses, and differences in the population definitions included in the past and current exposure analyses. Due to the differences in the exposure analyses, as well as changes in the exposure-response relationships (e.g., change in the shape of the exposure-response relationship from a linear relationship to one that is more sigmoidal or s-shaped), and changes in estimates for PRB used in the assessments, no direct comparison is being made of the differences in lung function risk estimates for children between the current and prior review.

We note that the current estimates for O_3-related hospital admissions for respiratory illness and asthma for New York City are higher than the estimates in the risk assessment conducted during the prior O_3 NAAQS review. Both the prior and current assessments used the same concentration-response functions for these health outcomes. The main reason for higher estimates in the current assessment is the use of a single value of 0.04 ppm for background in the prior review which is higher than the current modeled values for background in the current assessment which are in the range of about 0.015 to 0.035 ppm. Thus, under the current risk assessment O_3 levels above background but below 0.04 ppm are contributing additional estimated cases that were not included in the assessment for the prior O_3 NAAQS review.

5.4.5 Key Observations

In considering the quantitative estimates from the risk assessment the limitations and uncertainties associated with the risk estimates discussed in section 5.3.1.4 for lung function decrements and section 5.3.2.5 for respiratory symptoms, hospital admissions, and pre-mature mortality should be kept in mind. It is also important to consider the degree of confidence about the extent to which O_3 is causally related to each of the effects for which risk estimates were produced (see section 3.7.5). For example, there is clear and convincing evidence of causality for lung function decrements in healthy children under moderate exertion for 8-hr average O_3

exposures. We also judge that there is strong evidence for a causal relationship between respiratory symptoms in asthmatic children and O_3 exposures and between hospital admissions for respiratory causes and ambient O_3 exposures. There is greater uncertainty and somewhat less confidence about the relationship between O_3 and non-accidental and cardiorespiratory mortality, although the CD's overall evaluation is that it is highly suggestive that this relationship exists.

Recent O_3 Air Quality Levels

Section 5.4.1 has presented risk estimates associated with two recent years of air quality as represented by 2002 and 2004 monitoring data. Presented below are key observations resulting from this part of the risk assessment.

- The ranges in median estimates of the number of school age children (ages 5-18) estimated to experience at least one FEV_1 decrement $\geq$ 15% due to 8-hr O_3 exposures during the O_3 season across the 12 urban areas are 10,000 to nearly 220,000 (based on 2004 air quality) and 24,000 to about 345,000 (based on 2002 air quality). In terms of percent of this population the ranges in median estimates are 1.4 to 6.0% (based on 2004 air quality) and 5.3 to 9.9% (based on 2002 air quality). In terms of estimated occurrences of this same response the ranges in median estimates are 69,000 to nearly 1.5 million (based on 2004 air quality) and about 145,000 to over 1.5 million (based on 2002 air quality). The average number of occurrences per child in an O_3 season ranged from about 4 to 8 depending on urban area and year.
- The ranges in median estimates of the number of asthmatic school age children (ages 5-18) estimated to experience at least one FEV1 decrement $\geq$10% due to 8-hr O_3 exposures during the O_3 season across the five urban areas are 12,000 to 62,000 (based on 2004 air quality) and 17,000 to about 118,000 (based on 2002 air quality). In terms of the percent of this population the ranges in median estimates are 4.9 to 13.6% (based on 2004 air quality) and 12.5 to 18.3% (based on 2002). In terms of total occurrences of this same response the ranges in median estimates are 109,000 to about 660,000 responses (based on 2004 air quality) and from about 96,000 to 834,000 responses (based on 2002 air quality). Dividing the estimated total number of occurrences by the number of asthmatic children estimated to experience this lung function response, results in each child being estimated to have on average between roughly 5 and 10 occurrences depending on urban area and year.
- Estimates for increased respiratory symptoms (i.e., chest tightness, shortness of breath, and wheeze) in asthmatic children (ages 0-12) who used maintenance medications were only developed for the Boston urban area. The ranges in median estimates of symptom days for these three health outcomes are about 5,000 to 15,000 (based on 2004 air quality) and about 7,000 to 20,000 (based on 2002 air quality). In terms of percent of total incidence for these three health outcomes the ranges in median estimates are about 8 to 14% (based on 2004 air quality) and about 11 to 20% (based on 2002 air quality).
- Estimates for respiratory-related hospital admissions (e.g,, asthma-related) were developed for three urban areas (New York, Los Angeles, and Detroit). The median

estimates for New York are about 380 (based on 2004 air quality) and about 520 (based on 2002 air quality) O_3-related excess hospital admissions for asthma. For 2004 and 2002 air quality, these estimates represent about 3 and 4%, respectively, of total incidence.

- The risk assessment included a variety of estimates based on single- and multi-city studies for non-accidental and cardiorespiratory mortality. Since the median estimates from single-city and multi-city studies and models were generally of similar magnitude, with a few notable exceptions, we have focused on the estimates based on the multi-city studies to compare risk estimates across the 12 urban areas. The median estimates for incidence for non-accidental mortality (based on Bell et al., 2004 – 95 cities concentration-response function) range from about 3 to 130 (based on 2004 air quality) which is about 0.2 to 0.4% of total incidence. These same estimates based on 2002 air quality range from about 10 to 110 which is about 0.2 to 0.6% of total incidence. Estimates of O_3-related non-accidental mortality reported by Schwartz (2004) for Chicago, Detroit, and Houston, based on both single city and multi-city concentration-response functions, are somewhat higher than other estimates for these locations. This is mainly due to the use of the 1-hr maximum O_3 concentration in Schwartz (2004), rather than the 24-hr average, as the exposure metric.

- Examining the contribution of various O_3 ranges to these non-accidental mortality estimates, we found all of the mortality was associated with 24-hr average concentrations less than 0.06 ppm and most of it was associated with concentrations less than 0.04 ppm for 2004 air quality. For 2002, all of the O_3-related non-accidental mortality was associated with 24-hr average concentrations less than 0.08 ppm and the great majority was associated with concentrations less than 0.05 ppm. Based on an examination of O_3 air quality relationships between 24-hr average concentrations averaged over the urban monitors in an urban area on a given day and the highest daily maximum 8-hr average at any of the monitors in the same area on the corresponding day, we note that the 8-hr daily maximum concentrations are on average about twice the 24-hr average level. So, for example, a range of 0.04 to 0.06 ppm, 24-hr average corresponds with roughly daily maximum 8-hr levels in the range 0.08 to 0.12 ppm measured at the highest fixed-site monitor within an urban area.

Meeting the Current and Alternative 8-hr Standards

Section 5.4.2 has presented risk estimates associated with just meeting the current and several potential alternative 8-hr standards based on adjusting 2004 and 2002 monitoring data using design values for the 2002-2004 time period. Presented below are key observations resulting from this part of the risk assessment.

- In comparing risk estimates for alternative standards, uncertainties in quantifying the health risks associated ambient O_3 concentrations would be expected to remain relatively constant in different models. Thus, we have greater confidence in relative comparisons in risk estimates between alternative standards than in the absolute magnitude of risk

estimates associated with any particular standard. As discussed in section 5.4.3.1, differences in risk between alternative standards (e.g., reductions in risk for alternative standards relative to the current standard or recent air quality levels) is in most situations not impacted by assumptions about estimated PRB levels. Reductions in risks for alternative standards relative to the risks associated with just meeting the current standard are presented in section 6.3.4.

- Significant year-to-year variability in O_3 concentrations combined with the use of a 3-year design value to determine the amount of air quality adjustment to be applied to each year analyzed, results in significant year-to-year variability in the annual health risk estimates associated with just meeting the current and potential alternative 8-hr standards.

- The range in median estimates of the number of school age children (ages 5-18) estimated to experience at least one FEV_1 decrement > 15% due to 8-hr O_3 exposures during the O_3 season across the 12 urban areas when the current 8-hr standard is just met (based on adjusting 2002 air quality) are about 10,000 to 12,000. These estimated risks would be reduced to a range of 3,000 to 43,000 children under the most stringent alternative standard examined (i.e., 0.064 ppm, 4th-highest daily 8-hr maximum –i.e., the 0.064/4 alternative) based on adjusting 2002 air quality. In terms of percent of this population the ranges in median estimates are about 1.0 to 5.4% (based on 2002 air quality) for just meeting the current standard and these estimates are reduced to about 0.2 to 1.6% upon just meeting the 0.064/4 alternative standard. In terms of estimated occurrences of this same response the range in median estimates are 72,000 to about 750,000 when the current 8-hr standard is just met (based on 2002 air quality) These estimated risks would be reduced to a range of 26,000 to about 340,000 occurrences upon just meeting the most stringent alternative standard (0.064/4). The average number of occurrences per child in an O_3 season ranged from about 4 to 10 for air quality just meeting the current standard across the 12 urban areas (based on 2002 air quality). The average number of occurrences per child ranged from 4 to 15 for the most stringent alternative standard analyzed (0.064/4). The risk estimates associated with just meeting the current and alternative 8-hr standards based on adjusting 2004 air quality are generally of similar magnitude, although somewhat lower in 10 of the 12 urban areas.

- The ranges in median estimates of the number of asthmatic school age children (ages 5-18) estimated to experience at least one FEV_1 decrement $\geq$ 10% due to 8-hr O_3 exposures during the O_3 season across the five urban areas are about 9,000 to 63,000 or in terms of percentage of this population range from 3.4 to 10.9% associated with just meeting the current standard (based on adjusting 2002 air quality). These estimated risks would be reduced to a range of about 4,000 to 27,000 asthmatic children under the most stringent alternative standard examined (i.e., 0.064 ppm, 4th-highest daily 8-hr maximum –i.e., the 0.064/4 alternative) based on adjusting 2002 air quality. In terms of estimated occurrences of this same response the range of median estimates are 52,000 to nearly 510,000 responses associated with just meeting the current standard (based on adjusting 2002 air quality). These estimated risks would be reduced to a range of 14,000 to about 275,000 occurrences upon just meeting the most stringent alternative standard (0.064/4). The average number of occurrences per asthmatic child in an O_3 season ranged from

about 6 to 11 associated with just meeting the current standard (based on 2002 air quality). The average number of occurrences per asthmatic child ranged from about 4 to 12 upon meeting the most stringent alternative examined (0.064 ppm, 4th-highest daily maximum 8-hr average). The risk estimates associated with just meeting the current and alternative 8-hr standards based on adjusting 2004 air quality are generally of similar magnitude, although somewhat lower in 3 of the 5 areas analyzed.

- Estimates for increased respiratory symptoms (i.e., chest tightness, shortness of breath, and wheeze) in moderate/severe asthmatic children (ages 0-12) were only developed for the Boston urban area. The median estimated number of days involving chest tightness (using the concentration-response relationship with only O_3 in the model) ranges from 4,500 (based on adjusting 2004 air quality) to 6,100 (based on adjusting 2002 air quality) upon meeting the current 8-hr standard and these are reduced to 3,100 (based on adjusting 2004 air quality) to 4,600 days upon meeting the most stringent alternative examined (0.064 ppm, 4th-highest daily maximum 8-hr average). These same ranges correspond to 8 to 11% of total incidence of chest tightness upon meeting the current 8-hr standard and to about 5.5 to 8% of total incidence of chest tightness upon meeting a 0.064 ppm, 4th-highest daily maximum 8-hr average standard. Similar patterns of reduction were observed for each of the reported respiratory symptoms.

- Estimates for respiratory-related hospital admissions (e.g,, respiratory illness, asthma-related) were developed for three urban locations (New York City, Los Angeles, and Detroit). For asthma-related admissions in New York City the estimates are about 3.9 cases per 100,000 relevant population, which represents about 2.4% of total incidence or 313 cases upon just meeting the current standard based on adjusting 2004 air quality data. For increasingly more stringent alternative 8-hr standards, a gradual reduction in the cases per 100,000 relevant population is observed from 3.9 cases per 100,000 upon just meeting the current 8-hr standard to about 2.6 cases per 100,000 under the most stringent 8-hr standard (i.e., 0.064 ppm, average 4th-highest daily maximum) analyzed. Based on adjusting 2002 air quality data, asthma-related admissions in New York City are about 5.5 cases per 100,000 relevant population, which represents about 3.3% of total incidence or about 440 cases upon just meeting the current standard. For increasingly more stringent alternative 8-hr standards, a gradual reduction is observed from 5.5 cases per 100,000 (3.3% of total incidence) upon just meeting the current 8-hr standard to about 3.9 cases per 100,000 (2.4% of total incidence).

- Based on the median estimates for incidence for non-accidental mortality (based on Bell et al., 2004 – 95 cities concentration-response function), meeting the most stringent standard shown (0.064 ppm, 4th-highest daily maximum) is estimated to reduce mortality by 55 percent of what it would be associated with just meeting the current standard (based on adjusting 2004 air quality data). Adjusting 2002 air quality data to just meet the 0.064 ppm, standard results in a 40 percent reduction in non-accidental mortality relative to just meeting the current 8-hr standard . The patterns for cardiorespiratory mortality are similar. The aggregate O_3-related cardiorespiratory mortality at the most stringent standard shown is estimated to be about 57 percent of what it would be at the current standard, using simulated O_3 concentrations that just meet the current and

alternative 8-hour standards based on 2004 air quality data. Using 2002 air quality data, the corresponding result is about 42 percent.

- Much of the contribution to the risk estimates for non-accidental and cardiorespiratory mortality upon just meeting the current 8-hr standard is associated with 24-hr O_3 concentrations between background and 0.04 ppm. Based on examining relationships between 24-hr concentrations averaged across the monitors within an urban area and 8-hr daily maximum concentrations, 8-hr daily maximum levels at the highest monitor in an urban area associated with these averaged 24-hr levels are generally about twice as high.

Uncertainty and Variability

- There is noticeable variability in estimated O_3-related incidence of morbidity and mortality across the 12 urban areas analyzed for both recent years of air quality and for air quality adjusted to simulate just meeting the current and several potential alternative 8-hr standards. This variability is likely due to differences in air quality distributions, differences in exposure related to many factors including varying activity patterns and air exchange rates, differences in baseline incidence rates, and differences in susceptible populations and the age distribution across the 12 urban areas. For the lung function part of the risk assessment, spatial variability in air quality and population exposure inputs has been included in the assessment by use of a location specific exposure analysis and location specific input data to that analysis. For the epidemiology-based health endpoints, spatial variability in key inputs has been embedded in the analysis by use of location specific inputs (e.g., air quality, population data, baseline incidence data, concentration-response relationships).

- An important uncertainty to consider is the extent to which the associations between O_3 and the health endpoints included in the assessment actually reflect causal relationships. For lung function decrements, respiratory symptoms in moderate to severe asthmatic children, and respiratory-related hospital admissions there is clear and very strong evidence supporting the judgment that the relationships are causal. With respect to non-accidental and cardiorespiratory mortality, there is greater uncertainty, with the CD concluding that the overall body of evidence is highly suggestive that O_3 directly or indirectly contributes to nonaccidental and cardiopulmonary-related mortality (CD, p. 8-78). Given the overall evidence, including the strong evidence from the time-series studies reporting associations with non-accidental and cardiorespiratory mortality, along with information about potential mechanisms and a range of health effects observed in controlled human exposure studies (i.e., increased lung inflammation, impacts on host defense, and increased airway responsiveness), and strong evidence showing respiratory-related hospital admissions and emergency department visits for asthmatics, the staff judges that there is a likely causal relationship between O_3 exposures and non-accidental and cardiorespiratory mortality.

- Statistical uncertainty in the exposure-response relationships associated with sampling error has been characterized in the lung function part of the risk assessment. Other

important uncertainties in the exposure-response relationship for the lung function health outcomes include:

- uncertainty associated with the shape of the exposure-response relationship, which also has been considered using a Bayesian Markov Chain Monte Carlo approach recommended by members of the CASAC panel and extrapolation of the relationship to levels below 0.04 ppm, the lowest tested level in controlled human exposure studies;
- uncertainty due to use of 6.6-hr data for subjects engaged in moderate exertion to estimate response associated with 8-hr exposures under moderate or greater exertion;
- uncertainty about whether O_3-induced responses are reproducible, although this is generally supported by other controlled human exposure studies showing significant reproducibility of response;
- uncertainty introduced by use of exposure-response relationships based on 18 to 35 year old subjects to represent the relationships for all and asthmatic school age children age 5 to 18, although the use of adult data is supported by a study testing 8 to 11 year olds and observations from a number of summer camp field studies of school age children which found comparable responses to those observed in adults;
- uncertainty in the estimated exposure-response relationship due to assumption that response on any given day is independent of previous O_3 exposure; and
- uncertainty in the estimated exposure-response due to assumption that the response would not be affected by the presence of other co-pollutants.

- Uncertainties related to estimating the concentration-response relationships for the epidemiological-based part of the risk assessment include:

  - statistical uncertainty due to sampling error which is characterized in the assessment;
  - model uncertainty (i.e., uncertainty about the shape and magnitude of the concentration-response relationship taking into account lags, other pollutants, etc.); and
  - uncertainty about whether a concentration-response function provides an accurate representation of the relationship in the location of interest because of a) the possible role of associated co-pollutants, b) variations in the relationship of total ambient exposure to ambient monitoring in different location, and c) differences in population characteristics and population behavior patterns across locations or over time in the same location.

- Uncertainties related to the air quality data affect both the controlled human exposure studies-based and epidemiological studies-based parts of the risk assessment and include:
  - uncertainties associated with the air quality adjustment procedure that was used to simulate just meeting the current and alternative 8-hr standards; and

- uncertainties about estimated background concentrations for each location, for which sensitivity analyses were conducted examining the impact of alternative lower and higher assumed background concentrations.

- Based on the sensitivity analyses presented in the prior review, alternative air quality adjustment procedure had only a modest impact on the risk estimates. With respect to the uncertainties about estimated background concentrations, as discussed in section 5.4.3, alternative assumptions about PRB levels had a variable impact depending on the health effect considered and the location and standard analyzed in terms of the absolute magnitude of the risk estimates. In most, but not all cases, there was a relatively small impact on either absolute or relative changes in lung function risk estimates due to alternative assumptions about PRB levels. With respect to O_3-related non-accidental mortality, alternative assumptions about background levels had a greater impact. Estimates of risk remaining upon just meeting the current or alternative standards were most affected, with differences of +/-50% or larger observed in many of the areas. Alternative assumptions about PRB levels had a greater impact on the non-accidental mortality risk estimates associated with more stringent 8-hr standards. While noteable differences were observed for non-accidental mortality in some areas, particularly for more stringent standards, the overall pattern of reductions, expressed in terms of percentage reduction relative to the current standard, was significantly less impacted by alternative assumptions for PRB than the absolute magnitude of the risks.

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6. STAFF CONCLUSIONS AND RECOMMENDATIONS ON THE PRIMARY O_3 NAAQS

6.1 INTRODUCTION

This chapter presents staff conclusions and recommendations for the Administrator to consider in deciding whether the existing primary O_3 standard should be revised and, if so, what revised standard is appropriate. Our conclusions and recommendations are based on the assessment and integrative synthesis of information presented in the CD, staff analyses and evaluations presented in Chapters 2 through 5 herein, and the comments and advice of CASAC and interested parties who commented on earlier drafts of this document and related technical support documents.

In recommending policy options for the Administrator's consideration, we note that the final decision on retaining or revising the current O_3 standard is largely a public health policy judgment. A final decision should draw upon scientific information and analyses about health effects, population exposure and risks, as well as judgments about the appropriate response to the range of uncertainties that are inherent in the scientific evidence and analyses. Our approach to informing these judgments, discussed more fully below, is based on a recognition that the available health effects evidence generally 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.

This approach is consistent with the requirements of the NAAQS provisions of the Act and with how EPA and the courts have historically interpreted the Act. These provisions require the Administrator to establish primary standards that, in the Administrator's judgment, are requisite to protect public health with an adequate margin of safety. In so doing, the Administrator seeks to establish standards that are neither more nor less stringent than necessary for this purpose. The Act does not require that primary standards be set at a zero-risk level but rather at a level that avoids unacceptable risks to public health, including the health of sensitive groups.

6.2 APPROACH

To evaluate whether it is appropriate to consider retaining the current primary O_3 standard, or whether consideration of revisions is appropriate, we adopted an approach in this review that builds upon the general approach used in the last review and reflects the broader body of evidence now available. The 1997 final decision notice (62 FR 38856) outlined the key factors considered in selecting the elements of a standard for O_3 (judged to be the most

appropriate indicator for photochemical oxidants): the averaging time; the level (i.e., an ambient O_3 concentration); and the form (i.e., the air quality statistic used as a basis for determining compliance with the standard). Decisions on these elements were based on an integration of information on acute and chronic health effects associated with exposure to ambient O_3; expert judgments on the adversity of such effects on individuals; and policy judgments as to when the standard is requisite to protect public health with an adequate margin of safety, which were informed by air quality analysis and quantitative exposure and risk assessments when possible, as well as qualitative assessment of public health impacts that could not be quantified.

As in the last review, in developing conclusions and recommendations on the primary O_3 standard in this review, staff has taken into account both evidence-based and quantitative exposure- and risk-based considerations. Evidence-based considerations include the assessment of evidence from controlled human exposure, animal toxicological, field, and epidemiological studies for a variety of health endpoints. For those endpoints based on epidemiological studies, we have placed greater weight on associations with health endpoints that the CD has judged to be likely causal based on an integrative synthesis of the entire body of evidence, including not only all available epidemiological evidence but also evidence from animal toxicological and controlled human exposure studies. Less weight has been placed on evidence of associations that were judged to be only suggestive of possible causal relationships. For the purpose of evaluating the level of the O_3 standard in this review, we have placed greater weight on U.S. and Canadian studies, taking into account the extent to which such studies have reported statistically significant associations. This is because findings of U.S. and Canadian studies are more directly applicable for quantitative considerations in this review as studies conducted in other countries may well reflect quite different populations, exposure characteristics, and air pollution mixtures.

Staff's consideration of quantitative exposure- and risk-based information draws from the results of the exposure and risk assessments conducted for as many as twelve urban areas in the U.S. (discussed in Chapters 4 and 5). More specifically, we have considered estimates of the magnitude of O_3-related exposures and risks associated with recent air quality levels, as well as the exposure and risk reductions likely to be associated with just meeting the current 8-hr primary O_3 NAAQS and potential alternative 8-hr standards. We recognize the uncertainties inherent in such estimates, which are discussed in Chapters 4 and 5, in part by providing where possible some sense of the direction and/or magnitude of the uncertainties which should be taken into account as one considers these estimates.

In this review, a series of general questions frames our approach to reaching conclusions and recommendations in section 6.3 below, based on the available evidence and information, as to whether consideration should be given to retaining or revising the current primary O_3

standard. Our review of the adequacy of the current standard in section 6.3.1 begins by addressing questions such as the following:

- To what extent does newly available information reinforce or call into question evidence of associations of O_3 exposures with effects identified in the last review?
- To what extent has evidence of new effects and/or sensitive populations become available since the last review?
- To what extent have important uncertainties identified in the last review been reduced and have new uncertainties emerged?
- To what extent does newly available information reinforce or call into question any of the basic elements of the current standards?

To the extent that the available information suggests that revision of the current standard may be appropriate to consider, we also explore whether the currently available information supports consideration of a standard that is either more or less protective by addressing the following questions:

- Is there evidence that associations, especially likely causal associations, extend to ambient O_3 concentration levels that are as low as or lower than had previously been observed, and what are the important uncertainties associated with that evidence?
- Are exposures of concern and health risks estimated to occur in areas upon meeting the current standard; are they important from a public health perspective; and what are the important uncertainties associated with the estimated risks?

To the extent that there is support for consideration of a revised standard, we then consider the specific elements of the standard (in terms of an indicator for photochemical oxidants, averaging time, level, and form in sections 6.3.2 through 6.3.5, respectively) and identify policy options that we conclude would be appropriate for the Administrator to consider in making public health policy judgments, based on the currently available information, as to the degree of protection that is requisite to protect public health with an adequate margin of safety. In so doing, we address the following questions:

- Does the evidence provide support for considering a different O_3 indicator?
- Does the evidence provide support for considering different averaging times?
- What ranges of levels and forms of alternative standards are supported by the evidence, and what are the uncertainties and limitations in that evidence?
- To what extent do specific levels and forms of alternative standards reduce the estimated exposures of concern and risks attributable to O_3 and other photochemical

oxidants, and what are the uncertainties associated with the estimated exposure and risk reductions?

A summary of staff conclusions and recommendations on policy options for the Administrator's consideration concerning whether and, if so, how to revise the current primary O_3 standard is presented in section 6.3.6. This chapter concludes with a discussion of key uncertainties and recommendations for additional research related to setting a primary O_3 standard in section 6.4.

6.3 PRIMARY O_3 STANDARD

The current primary O_3 standard is an 8-hr standard set at a level of 0.08 ppm, with a form of the annual fourth-highest daily maximum 8-hr average concentration, averaged over three years, to provide protection to the public, especially children and other at-risk populations, against a wide range of O_3-induced effects. As an introduction to the discussion in this section of the adequacy of the current O_3 standard and potential options for alternative standards, it is useful to summarize the key factors that formed the basis of the decision in the last review to revise the averaging time, level, and form of the then current 1-hr standard.

In the last review, the key factor in deciding to revise the averaging time of the primary standard was evidence from controlled human exposure studies of healthy young adult subjects exposed for 1 to 8 hr to O_3. The best documented health endpoints in these studies were decrements in indices of lung function, such as forced expiratory volume in 1 second (FEV_1), and respiratory symptoms, such as cough and chest pain on deep inspiration. For short-term exposures of 1 to 3 hr, group mean FEV_1 decrements were statistically significant for O_3 concentrations only at and above 0.12 ppm, and only when subjects engaged in very heavy exertion. By contrast, prolonged exposures of 6 to 8 hr produced statistically significant group mean FEV_1 decrements at the lowest O_3 concentrations evaluated in those studies, 0.08 ppm, even when experimental subjects were engaged in more realistic intermittent moderate exertion levels. The health significance of this newer evidence led to the conclusion in the 1997 final decision that the 8-hr averaging time is more directly associated with health effects of concern at lower O_3 concentrations than is the 1-hr averaging time.

Based on the available evidence of O_3-related health effects, the following factors were of particular importance in the last review in informing the selection of the level and form of a new 8-hr standard: (1) quantitative estimates of O_3-related risks to active children, who were judged to be a sensitive subgroup of concern, in terms of transient and reversible respiratory effects judged to be adverse, including moderate to large decreases in lung function and moderate to severe pain on deep inspiration, and the uncertainty and variability in such estimates; (2) consideration of both the estimated percentages, total numbers of children, and

number of times they were likely to experience such effects; (3) epidemiological evidence of associations between ambient O_3 and increased respiratory hospital admissions, and quantitative estimates of percentages and total numbers of asthma-related admissions in one example urban area that were judged to be indicative of a pyramid of much larger effects, including respiratory-related hospital admissions, emergency department (ED) visits, doctor visits, and asthma attacks and related increased medication use; (4) quantitative estimates of the number of "exposures of concern" (defined as exposures $\geq$ 0.08 ppm for 6 to 8 hr) that active children are likely to experience, and the uncertainty and variability in such estimates; (5) the judgment that such exposures are an important indicator of public health impacts of O_3-related effects for which information is too limited to develop quantitative risk estimates, including increased nonspecific bronchial responsiveness (e.g., related to aggravation of asthma), decreased pulmonary defense mechanisms (suggestive of increased susceptibility to respiratory infection), and indicators of pulmonary inflammation (related to potential aggravation of chronic bronchitis or long-term damage to the lungs); (6) the broader public health perspective of the number of people living in areas that would breathe cleaner air as a result of the revised standard; (7) consideration of the relative seriousness of various health effects and the relative degree of certainty in both the likelihood that people will experience various health effects and their medical significance; (8) the relationship of a standard level to estimated "background" levels associated with nonanthropogenic sources of O_3; and (9) CASAC advice and recommendations. Additional factors that were considered in selecting the form of the standard included the public health implications of the expected number of times in an O_3 season that the standard level might be exceeded in an area that is in attainment with the standard and the year-to-year stability of the air quality statistic, so as to avoid disruptions to ongoing control programs which could interrupt public health protections.

In reaching a final decision in the last review, the Administrator was mindful that O_3 exhibits a continuum of effects, such that there is no discernible threshold above which public health protection requires that no exposures be allowed or below which all risks to public health can be avoided. The final decision reflected a recognition that important uncertainties remained, for example with regard to interpreting the role of other pollutants co-occurring with O_3 in observed associations, understanding biological mechanisms of O_3-related health effects, and estimating human exposures and quantitative risks to at-risk populations for these health effects.

6.3.1 Adequacy of Current O_3 Standard

Overall, the new evidence available in this review generally supports and builds further upon key health-related conclusions drawn in the previous review. New human clinical studies provide information about lung function and respiratory symptom responses to prolonged

exposures at O_3 levels at and below 0.08 ppm. There is an expanded body of evidence about the mechanisms of respiratory effects, including important new evidence about increased susceptibility of people with asthma and limited new evidence about plausible mechanisms by which O_3 exposure could induce effects on the cardiovascular system. In this review, there is additional epidemiological evidence supporting associations between O_3 exposure and respiratory symptoms in asthmatic children, ED visits and hospital admissions for respiratory causes, and new evidence that links O_3 exposure to premature mortality.

As discussed in Chapter 3, the CD concludes that, based on the extensive body of human clinical, toxicological, and epidemiological evidence, there is a causal relationship between short-term O_3 exposure and a range of respiratory morbidity effects, including lung function decrements, increased respiratory symptoms, airway inflammation, and increased airway responsiveness. Aggregate population time-series studies provide evidence that ambient O_3 concentrations are positively and robustly associated with respiratory-related hospitalizations and ED visits during the warm season. The CD concludes that the overall body of evidence supports a causal relationship between acute ambient O_3 exposures and these respiratory morbidity outcomes (CD, p. 8-77). Based on the evidence from animal toxicology, human clinical, and epidemiological studies, the CD concludes that a generally limited body of evidence provides considerable plausibility for cardiovascular mechanisms and is highly suggestive that O_3 can directly or indirectly contribute to cardiovascular-related morbidity, but that much needs to be done to more fully substantiate links between ambient O_3 exposures and adverse cardiovascular outcomes (CD, p. 8-77-78). The CD also finds that, consistent with observed O_3-related increases in respiratory- and cardiovascular-related morbidity, several newer U.S. multi-city time-series studies, single-city studies, and several meta-analyses of these studies, provide relatively strong epidemiological evidence for associations between short-term O_3 exposure and all-cause (non-accidental) mortality, even after adjustment for the influence of season and PM (CD, p. 8-78).

In considering this evidence as a basis for evaluating the adequacy of the current O_3 standard, we recognize that important uncertainties remain. For example, as discussed above in section 3.4, we note that inherent limitations in time-series epidemiological studies raise questions about the utility of such evidence to inform judgments about a NAAQS for an individual pollutant such as O_3 within a mix of highly correlated pollutants, such as the mix of photochemical oxidants, especially at ambient O_3 concentrations below levels at which O_3-related effects have been observed in controlled human exposure studies. We also recognize that the available epidemiological evidence neither supports nor refutes the existence of thresholds at the population level for effects such as increased hospital admissions and premature mortality. There are limitations in epidemiological studies that make discerning thresholds in populations

difficult, including low data density in the lower concentration ranges, the possible influence of exposure measurement error, and interindividual differences in susceptibility to O_3-related effects in populations.

While there clearly are limitations in interpreting the findings from the epidemiological studies, as noted above, we conclude for the following reasons that if a population threshold level does exist, it would likely be well below the level of the current O_3 standard and possibly within the range of background levels. This conclusion is supported by the discussions in Chapter 3 above (section 3.4.5) and more fully in the CD (Chapter 7, section 7.6.5) of the several epidemiological studies that have explored the question of potential thresholds and of the studies that included analyses excluding days over 0.08 ppm or even lower O_3 levels. We note that seasonal epidemiological studies show no consistent O_3-related effects during the cold season when O_3 concentrations are generally low, in contrast to a pattern of generally positive and statistically significant O_3-related effects in the warm season when O_3 concentrations are generally appreciably higher. In addition to direct consideration of the epidemiological studies, the findings from controlled human exposure studies strongly suggest that both lung function decrements and respiratory symptoms occur in healthy adult subjects at levels down to at least 0.060 ppm (+/- 0.003 ppm) with some subjects experiencing notable effects (e.g., >10% FEV_1 decrement, pain on deep inspiration). Controlled human exposure studies also found significant responses in indicators of lung inflammation and cell injury at 0.080 ppm (+/- 0.004 ppm) in healthy adult subjects. These effects were observed in healthy young adult subjects and it is likely that greater responses and responses at lower levels would occur in people with asthma and other respiratory diseases. As discussed in Chapter 3, the physiological effects observed in controlled human exposure studies have been linked to aggravation of asthma and increased susceptibility to respiratory infection, potentially leading to increased medication use, increased school and work absences, increased visits to doctors' offices and emergency departments (EDs), and increased hospital admissions. These observations provide additional support for the conclusion that the associations observed in the epidemiological studies, particularly for respiratory-related effects and potentially for cardiovascular effects, extend down to ozone levels well below 0.084 ppm.

Based on the above considerations and findings from the CD, while being mindful of important remaining uncertainties, staff concludes that the newly available information generally reinforces our judgments about causal relationships between O_3 exposure and respiratory effects observed in the last review and broadens the evidence of O_3-related associations to include additional respiratory-related endpoints, newly identified cardiovascular-related health endpoints, and mortality. Newly available evidence also has identified increased susceptibility in people with asthma. While recognizing that important uncertainties and research questions

remain, we also conclude that progress has been made since the last review in advancing the understanding of potential mechanisms by which ambient O_3, alone and in combination with other pollutants, is causally linked to a range of respiratory- and cardiovascular-related health endpoints. Thus, we generally find support in the available evidence, including the direction of the evidence developed since the last review, for consideration of an O_3 standard that is at least as protective as the current standard and do not find support for consideration of an O_3 standard that is less protective than the current standard. This general conclusion is consistent with the advice and recommendations of CASAC and with the views expressed by all interested parties who provided comments on the previous draft of this document. While CASAC and some commenters supported revising the current standard to provide increased public health protection, and other commenters supported retaining the current standard, no one who provided comments supported a standard that would be less protective than the current standard.

Having reached this general conclusion, we discuss in greater detail below the available evidence (section 6.3.1.1) and exposure- and risk-based considerations (section 6.3.1.2) to more fully inform consideration of the adequacy of the current standard. We also take into account the views expressed by CASAC and public commenters (section 6.3.1.3) in reaching staff conclusions on the adequacy of the current standard (section 6.3.1.4).

6.3.1.1 Evidence-based Considerations

In looking more specifically at the controlled human exposure and epidemiological evidence summarized in Chapter 3 and Appendix 3B, staff first notes that controlled human exposure studies provide the clearest and most compelling evidence for an array of human health effects that are directly attributable to acute exposures to O_3 per se (CD, p. 8-73). We also note that evidence from such human studies, together with animal toxicological studies, help to provide biological plausibility for health effects observed in epidemiological studies. In considering the available evidence, we have focused on studies that examined health effects that have been demonstrated to be caused by exposure to O_3 or for which the CD judges associations with O_3 to be causal or likely causal. In considering the epidemiological evidence as a basis for reaching conclusions about the adequacy of the current standard, we have focused on studies reporting effects in the warm season, for which the effect estimates are more consistently positive and statistically significant than those from all-year studies. We have considered the extent to which such studies provide evidence of associations that extend down to ambient O_3 concentrations below the level of the current standard, which would call into question the adequacy of the current standard. In so doing, we note, as discussed above, that if a population threshold level does exist for an effect observed in such studies, it would likely be at a level well below the level of the current standard. We have also sought to characterize whether the area in

which a study was conducted likely would or would not have met the current standard during the time of the study, although we recognize that the confidence that would appropriately be placed on the associations observed in any given study, or on the extent to which the association would likely extend down to relatively low O_3 concentrations, is not dependent on this distinction. Further, we have considered studies that examined subsets of data that include only days with ambient O_3 concentrations below the level of the current O_3 standard, or below even lower O_3 concentrations, and continue to report statistically significant associations. We judge that such studies are directly relevant to considering the adequacy of the current standard, particularly in light of reported responses to O_3 at levels below the current standard found in controlled human exposure studies.

In examining air quality information from the epidemiological studies for the purpose of determining whether they were conducted in areas that likely would or would not have met the current standard, we note that it is difficult to consistently characterize relevant air quality statistics.[1] These difficulties arise in particular in panel studies of lung function or respiratory symptoms in which the study periods were often shorter than a complete O_3 season; Appendix 3B includes 98^{th} and 99^{th} percentile values as a way to approximate the fourth-highest value for studies with differing study periods. Difficulties also arise in all studies in which the air quality data were averaged across multiple monitors in a study area (as are reported in Appendix 3B), since an area's attainment status is determined by the monitor measuring the highest O_3 concentrations in an area, not averaged across monitors. For studies with relatively low air quality values that are based on averaging across multiple monitors, we have further explored the available air quality data so as to help inform a comparison with the level of the current standard, as discussed below.

Lung Function, Respiratory Symptoms, and Other Respiratory Effects

Health effects for which the CD continues to find clear evidence of causal associations with short-term O_3 exposures include lung function decrements, respiratory symptoms, pulmonary inflammation, and increased airway responsiveness. In the last review, these O_3-induced effects were demonstrated with statistical significance down to the lowest level tested in controlled human exposure studies at that time (i.e., 0.08 ppm). As discussed in Chapter 3 (section 3.3.1.1.1), in new controlled human exposure studies, healthy adult volunteers were exposed to 6.6-hr average O_3 levels down to lower levels (i.e., 0.04 and 0.06 ppm) while engaged in moderate exertion. These studies did not report statistically significant changes in the group mean FEV_1 decrements between lung function decrements associated with the 0.06 ppm or 0.04

[1]Determining attainment with the current standard is based on the 3-year average of the annual (over an O_3 season) fourth-highest daily maximum 8-hr average O_3 concentration at each monitor in an area.

ppm levels versus the filtered air (i.e., essentially 0 ppm O_3) exposure when lung function changes were analyzed for each hourly interval (i.e., after 1, 2, 3, 4.6, 5.6, and 6.6 hours of exposure). However, as discussed in Section 3.3.1.1.1, we note that the pre- and post-exposure data presented in the Adams (2006) study show a small (< 3%) group mean FEV_1 decrement following the 6.6-hr exposure at 0.06 ppm, which may be statistically significantly different from filtered air responses. Notably, total respiratory symptoms (which includes pain on deep inspiration, shortness of breath, and cough) following 5.6 and 6.6 h exposures at 0.06 ppm (during a triangular exposure pattern, that is more representative of those encountered in summer air pollution episodes than a square-wave exposure pattern) reached statistical significance. In addition to information about group mean decrements, this study also reports that a small percentage (7%) of healthy adult subjects experienced moderate lung function decrements (≥ 10% FEV_1) with exposure to 0.06 and 0.04 ppm O_3 when corrected for the effects of exercise in clean air. The distribution of individual responses related to lung function decrements (≥ 10, 15, and 20% FEV_1) found in these new studies are considered as part of the quantitative risk assessment for lung function responses in children discussed below.

Newer information indicates that people with moderate-to-severe asthma have somewhat larger decreases in lung function in response to O_3 relative to healthy individuals and that lung function responses in people with asthma appear to be affected by baseline lung function (i.e., responses increase with increasing disease severity, CD, p. 8-80). As discussed in the CD (Chapter 6, sections 6.8 and 6.9; Chapter 8, sections 8.7 and 8.8) this newer information expands our understanding of the physiological basis for increased sensitivity in people with asthma and other airway diseases, recognizing that people with asthma present a different response profile for cellular, molecular, and biochemical responses than people who do not have asthma. New evidence indicates that some people with asthma have increased occurrence and duration of nonspecific airway responsiveness, which is an increased bronchoconstrictive response to airway irritants. Controlled-human exposure studies also indicate that some people with allergic asthma and rhinitis have increased airway responsiveness to allergens following O_3 exposure. Exposures to O_3 exacerbated lung function decrements in people with pre-existing allergic airway disease, with and without asthma. Ozone-induced exacerbation of airway responsiveness persists longer and attenuates more slowly than O_3-induced lung function decrements and respiratory symptom responses and can have important clinical implications for asthmatics.

Newly available human exposure studies suggest that some people with asthma also have increased inflammatory responses, relative to non-asthmatic subjects, and that this inflammation may take longer to resolve. The new data on airway responsiveness, inflammation, and various molecular markers of inflammation and bronchoconstriction indicate that people with asthma and allergic rhinitis (with or without asthma) comprise susceptible groups for O_3-induced

adverse effects. This body of evidence qualitatively informs our evaluation of the adequacy of the current O_3 standard in that it indicates that human clinical and epidemiological panel studies of lung function decrements and respiratory symptoms that evaluate only healthy, non-asthmatic subjects likely underestimate the effects of O_3 exposure on asthmatics and other susceptible populations.

In addition to the experimental evidence of lung function decrements, respiratory symptoms, and other respiratory effects in healthy and asthmatic populations discussed above, epidemiological studies have reported associations of lung function decrements and respiratory symptoms in several locations (Appendix 3B; also Figure 3-4 for respiratory symptoms). As discussed in Chapter 3, two large U.S. studies (Mortimer et al., 2002 (the National Cooperative Inner-City Asthma Study (NCICAS)), Gent et al., 2003), as well as several smaller U.S. and international studies, have reported fairly robust associations between ambient O_3 concentrations and measures of lung function and daily symptoms (e.g., chest tightness, wheeze, shortness of breath) in children with moderate to severe asthma and between O_3 and increased asthma medication use. The NCICAS reported statistically significant increases in incidence of $\geq$ 10% declines in morning lung function and respiratory symptoms in asthmatic children for multi-day lags in 8-hr average O_3 concentrations in single pollutant models. For various co-pollutant models, the O_3 effect was attenuated, but there was still a positive association. Gent et al. (2003) included asthmatic children in the area of southern New England and reported associations between various respiratory symptoms and both daily 1-hr maximum and 8-hr maximum O_3 levels for asthmatics who used maintenance medications and would be considered moderate to severe asthmatics, while not finding an association for mild asthmatics, defined as not using maintenance medication. In this study, effects of O_3, but not $PM_{2.5}$, remained statistically significant and even increased in magnitude in two-pollutant models (CD, p.7-53). The CD concludes that overall the multi-city NCICAS, Gent et al. (2003) and several other single-city studies indicate a fairly robust positive association between ambient O_3 concentrations and increased respiratory symptoms in asthmatics. We recognize, however, that uncertainties remain with regard to the relative contribution of O_3 and other co-pollutants, some of which show moderate correlations during the summer time, for the effects observed in asthmatic individuals.

In considering the large number of single-city epidemiological studies reporting lung function or respiratory symptoms in healthy or asthmatic populations (Appendix 3B), we note that most such studies that reported positive and often statistically significant associations in the warm season were conducted in areas with relevant air quality statistics that are indicative of areas that likely would not have met the current standard (e.g., Gent et al., 2003; Ostro et al., 2001; Neas et al., 1995; Delfino et al, 1998; Linn et al., 1996; Korrick et al., 1998). In considering the large multi-city NCICAS (Mortimer et al., 2002), we note that the 98th percentile

8-hr daily maximum O_3 concentrations at the monitor reporting the highest O_3 concentrations in each of the study areas ranged from 0.084 ppm to > 0.10 ppm. The authors indicate that less than 5 percent of the days in the eight areas had 8-hr daily O_3 concentrations exceeding 0.080 ppm. The authors observed that when days with 8-hr average O_3 levels greater than 0.080 ppm were excluded, similar effect estimates were seen.

There also are other studies in which the relevant air quality statistics provide some indication that lung function and respiratory symptom effects may be occurring in areas that likely would have met the current standard (e.g., Naeher et al., 1999; Ross et al., 2002; Brauer et al., 1996). We note that this last group of studies reported associations that were often but not always statistically significant, and that Brauer et al. (1996) was an outdoor worker study of berry pickers with exposure patterns that would not be typical of the general population.

Respiratory Hospital Admissions and Emergency Department Visits

As discussed in Chapter 3 (section 3.3.1.1.6), at the time of the last review, many time-series studies indicated positive associations between ambient O_3 and increased respiratory hospital admissions and emergency room visits, providing strong evidence for a relationship between O_3 exposure and increased exacerbations of preexisting lung disease at O_3 levels below the level of the then current 1-hr standard. Analyses of data from studies conducted in the northeastern U.S. indicated that O_3 air pollution was consistently and strongly associated with summertime respiratory hospital admissions (CD, section 8.4.4).

Since the last review, new epidemiological studies have evaluated the association between short-term exposures to O_3 and unscheduled hospital admissions for respiratory causes (CD, section 7.3.3). Large multi-city studies, as well as many studies from individual cities have reported positive and often statistically significant O_3 associations with total respiratory hospitalizations as well as asthma- and COPD-related hospitalizations, especially in studies analyzing the O_3 effect during the summer or warm season. Analyses using multipollutant regression models suggest that copollutants generally do not confound the association between O_3 and respiratory hospitalizations, and that the O_3 effect estimates were generally robust to PM adjustment in all-year and warm-season only data (CD, p. 7-79; Figure 7-12). The CD concludes that the evidence supports a causal relationship between acute O_3 exposures and increased respiratory-related hospitalizations during the warm season (CD, p. 8-77).

In looking specifically at U.S. and Canadian respiratory hospitalization studies that reported positive and often statistically significant associations (and that either did not use GAM or were reanalyzed to address GAM-related problems), we note that many such studies were conducted in areas that likely would not have met the current O_3 standard, with many providing only all-year effect estimates, and with some reporting a statistically significant association in the warm season (e.g., Schwartz (1996) – Cleveland). Of the studies that provide some indication

that O_3-related respiratory hospitalizations may be occurring in areas that likely would have met the current standard, we note that some are all-year studies (e.g., Sheppard et al., 2003; Yang et al., 2003), whereas others reported statistically significant warm-season associations (e.g., Burnett et al., 1997a, in 16 Canadian cities; and Burnett et al., 1997b, 2001, in Toronto). In further examining the relevant air quality statistics in the 16 Canadian cities study (Burnett et al., 1997a), we observe that while the aggregated 98th percentile O_3 concentration was calculated as 47 ppb (Appendix 3B), the fourth-highest values at the monitors reporting the highest O_3 concentrations in each of the cities ranged from approximately 35 to 110 ppb, making it difficult to determine the extent to which the reported association can be attributed to effects occurring in areas that likely would have met the current U.S. standard. We also further examined the relevant air quality statistics in the Burnett et al. (1997b, 2001) studies in Toronto. We observed that in one of those studies (Burnett et al., 2001) the fourth-highest values at the highest monitor ranged from approximately 80 to 150 ppb across the years of the study (from 1980 to 1994). In the other study (Burnett et al., 1997b) the calculated 98th percentile values averaged across the several monitors used in the study ranged from 62 to 64 ppb in each of the three years of the study (Appendix 3B), but individual monitor data were not available for further examination. Based on these observations, we find it difficult to judge the extent to which these studies provide evidence of an association with respiratory-related hospitalizations in areas that likely would have met the current standard. Nonetheless, as discussed above, we recognize that these studies do provide evidence of associations that likely extend down to relatively low ambient O_3 concentrations, well below the level of the current standard.

Emergency department visits for respiratory causes have been the focus of a number of new studies that have examined visits related to asthma, COPD, bronchitis, pneumonia, and other upper and lower respiratory infections, such as influenza, with asthma visits typically dominating the daily incidence counts (CD, section 7.3.2). Among studies with adequate controls for seasonal patterns, many reported at least one significant positive association involving O_3. However, inconsistencies were observed which were at least partially attributable to differences in model specifications and analysis approach among various studies. In general, O_3 effect estimates from summer-only analyses tended to be positive and larger compared to results from cool season or all-year analyses. Almost all of the studies that reported statistically significant effect estimates had calculated 98th percentile O_3 concentrations (Appendix 3B), averaged across monitors, that are indicative of areas that likely would not have met the current standard. The notable exception were two studies in Montreal (Delfino et al., 1997, 1998) that reported statistically significant warm-season associations with O_3 and ED visits in a population of older adults with a calculated 98th percentile value, averaged across several monitors, of approximately 60 ppb (and for which individual monitor data were not available for further

examination), although the CD raises questions about the plausibility of this result due to the low O_3 concentrations and inconsistent results across years and age groups. The CD concluded that analyses stratified by season generally supported a positive association between O_3 concentrations and ED visits for asthma in the warm season. These studies provide evidence of effects in areas that likely would not have met the current standard, and evidence of associations that likely extend down to relatively low ambient O_3 concentrations.

Mortality

The 1996 CD concluded that an association between daily mortality and O_3 concentrations for areas with high O_3 levels (e.g., Los Angeles) was suggested. However, due to a very limited number of studies available at that time, there was insufficient evidence to conclude that the observed association was likely causal, and thus the possibility that O_3 exposure may be associated with mortality was not relied upon in the 1997 decision on the O_3 primary standard.

Since the last review, as described above, the body of evidence with regard to O_3-related health effects has been expanded by animal, human clinical, and epidemiological studies, and now includes biologically plausible mechanisms by which O_3 may affect the cardiovascular system. In addition, there is stronger information linking ozone to serious morbidity outcomes, such as hospitalization, that are associated with increased mortality. Thus, there is now a coherent body of evidence that describes a range of health outcomes from pulmonary function decrements to hospitalization and premature mortality.

Newly available large multi-city studies designed specifically to examine the effect of O_3 and other pollutants on mortality have provided much more robust and credible information. The extended NMMAPS analysis included data from 95 U.S. cities and included an additional 6 years of data, from 1987-2000 (Bell et al., 2004), and significant associations were reported between O_3 and mortality that were robust to adjustment for PM (CD, p. 7-100). Using a subset of the NMMAPS data set, Huang et al. (2005) focused on associations between cardiopulmonary mortality and O_3 exposure (24-hr average) during the summer season only. The authors report the increase in mortality per 20 ppb change in O_3 concentration using a 7-day distributed lag model was greater than the increase in mortality measured on the same day (CD, p. 7-92), suggesting that the effect of O_3 on mortality is immediate but also persists for several days. Using a case-crossover study design, Schwartz (2005) assessed associations between daily maximum concentrations and mortality in 14 cities, matching case and control periods by temperature, and using data only from the warm season. The reported effect estimate was similar to time-series analysis results with adjustment for temperature, suggesting that associations between O_3 and mortality were robust to the different adjustment methods for temperature (CD, p. 7-93). Two of the recent multi-city mortality studies (Bell et al., 2004;

Gryparis et al., 2004) have also reported associations for multiple averaging times (CD, p. 8-38). Bell and colleagues (2004) reported an effect estimate for the association with 1-hr O_3 concentrations that was slightly larger than that reported for 8-hr O_3 concentrations, and both were slightly larger than the association with 24-hr average O_3, but the effect estimates did not differ statistically.

One recent multi-city study (Bell et al., 2006) examined the shape of the concentration-response function for the O_3-mortality relationship in 98 U.S. urban communities for the period 1987 to 2000 specifically to evaluate whether a "safe" threshold level exists. Results from various analytic methods all indicated that any threshold, if it exists, would likely occur at very low concentrations, far below the level of the current O_3 NAAQS and other, lower international O_3 standards,[2] and nearing background levels. Notably, in a subset analysis using only days that were below the level of the current O_3 NAAQS, the O_3-mortality association remained statistically significant with only a small change in the size of the effect estimate. Further, in a subset analysis based on 24-hr average O_3 concentrations, the effect estimates declined and lost statistical significance only when the maximum daily average concentration included was $\leq$ 10 ppb (Bell et al., 2006, p. 14 and Figure 2), which corresponds to daily maximum 8-hr average concentrations in U.S. cities that are within the range of background concentrations. The authors conclude that "interventions to further reduce ozone pollution would benefit public health, even in regions that meet current regulatory standards and guidelines" (Bell et al., 2006, p. 3).

New data are also available from several single-city studies conducted world-wide, as well as from several meta-analyses that have combined information from multiple studies. The majority of these studies suggest that there is an elevated risk of total non-accidental mortality associated with acute exposure to O_3, especially in the summer or warm season when O_3 levels are typically high, with somewhat larger effect estimate sizes for associations with cardiovascular mortality (CD, p. 7-175). As shown in Figure 7-21 of the CD, the results of recent publications show a pattern of positive, often statistically significant associations between short-term O_3 exposure and mortality during the warm season (CD, p. 7-97). For example, statistically significant associations were reported in southern California (Ostro, 1995), Philadelphia (Moolgavkar et al., 1995), Dallas (Gamble et al., 1998), and Vancouver (Vedal et al., 2003), as well as numerous studies conducted in other countries. In contrast, no evidence of an association was seen in a study conducted in Pittsburgh (Chock et al., 2000). In considering results from year-round analyses, there remains a pattern of positive results but the findings are

[2] Other international 8-hr O_3 standards considered by Bell et al. (2006, Table 1) include the California standard of 0.070 ppm, the Canadian standard of 0.065 ppm, and the World Health Organization guideline and European Commission target value of approximately 0.061 ppm.

less consistent. For example, statistically significant associations were reported in Philadelphia (Moolgavkar et al., 1995) and Dallas (Gamble et al., 1998), while positive but not statistically significant associations were reported in Detroit (Lippmann et al., 2000, reanalyzed in Ito, 2003), San Jose (Fairley, 1999, reanalyzed Fairley, 2003), and Atlanta (Klemm et al., 2004). No evidence for associations was reported in Los Angeles (Kinney et al., 1995), Coachella Valley (Ostro et al., 2003), and St. Louis and Eastern Tennessee (Dockery et al., 1992). In most single-city analyses, effect estimates were not substantially changed with adjustment for PM (CD Figure 7-22, p. 7-101).

Almost all single-city studies that show statistically significant associations with mortality had calculated 98th percentile O_3 concentrations (Appendix 3B) that are indicative of O_3 levels that likely would not have met the current standard. The notable exception was a study in Vancouver (Vedal et al., 2003) that reported a statistically significant warm-season association with O_3 and total non-accidental mortality that was robust in two-pollutant models, with a calculated 98th percentile value, averaged across many monitors, of approximately 53 ppb. Upon further examination, the relevant air quality statistics for each individual monitor in this study ranged from 57 to 59 ppb. This study provides evidence of an O_3-related mortality association in the warm season in an area with O_3 levels that were well below those that would have met the current standard. However, the authors questioned whether O_3, other gaseous pollutants, and PM may be acting as surrogate markers of pollutant mixes that contain more toxic compounds, since the low measured concentrations were unlikely, in their opinion, to cause the observed effects (CD, p.7-155). Another study done in Vancouver over a much longer time period (Villeneuve et al., 2003) did not provide evidence of O_3-related mortality associations, but only all-year results were presented which may be more likely confounded by other pollutants than the warm-season results in Vedal et al. (2003).

Three recent meta-analyses evaluated potential sources of heterogeneity in O_3-mortality associations (Bell et al., 2005; Ito et al., 2005; Levy et al., 2005). The CD (p. 7-96) observes common findings across all three analyses, in that all reported that effect estimates were larger in warm season analyses, reanalysis of results using default GAM criteria did not change the effect estimates, and there was no strong evidence of confounding by PM (CD, p. 7-97). Bell et al. (2005) and Ito et al. (2005) both provided suggestive evidence of publication bias, but O_3-mortality associations remained after accounting for that potential bias. The CD (7-97) concludes that the "positive O_3 effects estimates, along with the sensitivity analyses in these three meta-analyses, provide evidence of a robust association between ambient O_3 and mortality."

The CD finds that the results from U.S. multi-city time-series studies, along with the meta-analyses, provide relatively strong evidence for associations between short-term O_3 exposure and all-cause mortality even after adjustment for the influence of season and PM (CD,

p. 8-78). The results of these analyses indicate that copollutants generally do not appear to substantially confound the association between O_3 and mortality (CD, p. 7-103; Figures 7-22 and 7-23). In addition, several single-city studies observed positive associations of ambient O_3 concentrations with total nonaccidental and cardiopulmonary mortality. Finally, from those studies that included assessment of associations with specific causes of death, it appears that effect estimates for associations with cardiovascular mortality are larger than those for total mortality; effect estimates for respiratory mortality are less consistent in size, possibly due to reduced statistical power in this subcategory of mortality (CD, p. 7-108). For cardiovascular mortality, the CD (Figure 7-25, p. 7-106) suggests that effect estimates are consistently positive and more likely to be larger and statistically significant in warm season analyses. The CD (p. 8-78) concludes that these findings are highly suggestive that short-term O_3 exposure directly or indirectly contributes to non-accidental and cardiopulmonary-related mortality, but additional research is needed to more fully establish underlying mechanisms by which such effects occur.

6.3.1.2 Exposure- and Risk-based Considerations

In addition to the evidence-based considerations, staff has also considered exposures and health risks estimated to occur upon meeting the current O_3 standard to help inform judgments about the extent to which exposure and risk estimates may be judged to be important from a public health perspective, taking into account key uncertainties associated with the estimated exposures and risks. For this review, exposures have been estimated for people of all ages, school age children (ages 5-18), and asthmatic school age children in 12 urban areas across the U.S.[3] In this discussion we focus particularly on the exposure estimates for all and asthmatic school age children while at moderate or greater exertion levels since these groups are particularly at risk for experiencing O_3-related health effects due to the greater amount of time spent outdoors during the O_3 season engaged in relatively high levels of physical activity. With regard to the quantitative risk assessment, we have estimated the occurrences of moderate or greater lung function decrements in all and asthmatic school age children, respiratory symptoms in children with moderate to severe asthma, respiratory-related hospital admissions, and non-accidental and cardiorespiratory mortality. We have selected these particular health endpoints because they can be considered to be adverse to the health of individuals, either in single or with repeated occurrences, and the requisite data for the assessment are available.

In making judgments as to when various O_3-related effects become regarded as adverse to the health of individuals, staff has relied upon the guidelines published by the American

[3]As discussed in Chapter 4, since the proportion of children classified as "active" in the exposure analysis has been overestimated, in part due to uncertainty in the CHAD MET values, we have not discussed this population group in this chapter.

Thoracic Society (ATS) and the advice of CASAC. As discussed in Chapter 3, the ATS (1985) has defined adverse respiratory health effects as medically significant physiologic changes, which include acute changes such as those that interfere with the normal activity or acute respiratory illness, and longer-term changes such as progressive respiratory dysfunction, permanent respiratory injury, or incapacitating illness. Of the morbidity effects estimated in the risk assessment, hospital admissions are clearly adverse to the health of the individual. Recognizing that respiratory symptoms in moderate to severe asthmatic children are likely to cause increased medication use, interference with normal activities, or increased absences from school, occurrences of such symptoms are considered to be adverse. For active healthy people, moderate lung function decrements would likely interfere with normal activity for relatively few individuals within this group who are particularly responsive to O_3 exposures; whereas large functional responses would likely interfere with normal activities for a greater proportion of individuals within this group, such that even single occurrences of large functional responses would be considered adverse under ATS guidelines. In judging the extent to which moderate lung function decrements are adverse, especially in healthy people, an additional factor that has been considered is whether these occur on a single occasion or repeatedly over the course of an O_3 season. It has been judged that repeated occurrences of moderate responses, even in otherwise healthy individuals, may be considered to be adverse since they could well set the stage for more serious illness. For people with lung disease, even moderate lung function decrements would likely interfere with normal activity for many individuals within this group, and would likely result in additional and more frequent use of medication; large functional responses would likely interfere with normal activity for most individuals in this group and would increase the likelihood that these individuals would seek medical treatment. Thus, occurrences of either moderate or large functional responses in people with lung disease would be considered to be adverse to the health of individuals experiencing these effects.

Beyond the health effects discussed above that are included in the risk assessment, Chapter 3 discusses a broader array of O_3-related health endpoints that are representative of a "pyramid of effects" that include various indicators of morbidity that could not be included in the risk assessment (e.g., school absences, increased medication use, ED visits). Ozone-related effects that are judged to be important indicators of this broader array of health endpoints, and are thus potentially adverse to the individuals experiencing such effects, include: (1) increased nonspecific airway responsiveness which is related, for example, to aggravation of asthma, potentially leading to increased medication use, increased school and work absences, increased visits to doctors' offices and EDs, and increased admissions to hospitals; (2) decreased pulmonary defense mechanisms which are suggestive of increased susceptibility to respiratory infection, potentially leading to increased school and work absences, increased visits to doctors'

offices and EDs, and increased admissions to hospitals; and (3) indicators of pulmonary inflammation which are related to aggravation of asthma, potentially leading to increased medication use, increased school and work absences, increased visits to doctors' offices and EDs, and increased admissions to hospitals; increased cellular permeability associated with inflammation may be a mechanism by which O_3 exposure can lead to cardiovascular system effects; and potential chronic effects such as chronic bronchitis or long-term damage to the lungs that can lead to reduced quality of life. Some perspective on the O_3-related public health impacts of these types of effects are characterized based on the results of the exposure analysis in terms of estimates of the number of occurrences of "exposures of concern," as discussed below.

We estimated exposures and risks for the three most recent years (2002 - 2004) for which data were available at the time of the analyses, as discussed in Chapters 4 and 5. Within this 3-year period, 2002 was a year with generally poorer air quality in most, but not all, areas and provides a generally more upper-end estimate of exposures and risks, while 2004 was a year with generally better air quality in most, but not all, areas and provides a generally more lower-end estimate of exposures and risks. Exposure and risk estimates for the year 2003 generally fall between the estimates for 2002 and 2004. In presenting these results, we note that the exposure analysis and risk assessment discussed in Chapters 4 and 5, respectively, identify a number of uncertainties, as highlighted below.

With respect to the exposure analysis, the exposure modeling approach accounts for variability in ambient O_3 levels, demographic characteristics, physiological attributes, activity patterns, and factors affecting microenvironmental concentrations. In our judgment the most important uncertainties affecting the exposure estimates are related to the modeling of activity patterns over an O_3 season, modeling micro-scale variations in ambient concentrations (e.g., near roadways), and modeling air exchange rates in microenvironments. Another important uncertainty that does not directly affect estimates of exposure, but affects the characterization of how many exposures are associated with moderate or greater exertion, is the characterization of energy expenditure (i.e., measured in terms of METS - metabolic equivalents of work) for children engaged in various activities. As discussed in section 4.3.4.7, the uncertainty in METS values carries over to the uncertainty of the modeled ventilation rates, which are important since they are used to classify exposures of potentially greater risk.

A comprehensive picture of the uncertainty of the exposure model estimations has been developed through sets of complementary analyses addressing these different aspects of the overall uncertainty. An analysis was performed which accounts for the uncertainties associated with the microenvironment models and the ambient air quality data. Analyses have been conducted to address the remaining significant sources of uncertainty: near-road exposures and the activity data. The uncertainty of the model structure (as distinct from uncertainty driven by

uncertain model inputs) is judged to be less important than the uncertainties of the model inputs and parameters. Based on these analyses, relatively small uncertainties are associated with the estimation of ambient concentrations and microenvironment model parameters. Relatively larger uncertainties are associated with the representativeness of the CHAD activity diaries with respect to the specific cities and time periods modeled. The APEX model significantly underestimates the frequency of occurrences of individuals experiencing repeated 8-hour average exposures greater than 0.06, 0.07, and 0.08 ppm. While the frequency of repeated occurrences is significantly underestimated, we have more confidence that the estimates for total number of occurrences (i.e., person days with exposures greater than 0.06, 0.07, and 0.08 ppm) are not biased. Section 4.6 provides a summary of the exposure modeling uncertainties; the details of this uncertainty analysis are described in Langstaff (2007). It is important to note that there have been significant improvements in several components of the exposure model and in the inputs to the model (e.g., better characterization of the ambient air quality surface across each area, more complete data on air exchange rates, much larger human activity database) relative to the exposure analysis conducted for the 1997 review. Thus, while we recognize and have considered the kind and degree of uncertainties associated with the exposure estimates, we believe, consistent with CASAC's views (Henderson, 2006c), that the exposure analysis represents a state-of-the-art modeling approach and the quality of the estimates is such that they are suitable to be used as an input to the decisions on the O_3 standard.

As discussed in Chapter 5, uncertainties related to the air quality data affect both the controlled human exposure studies-based and the epidemiological studies-based parts of the risk assessment. These include uncertainties associated with the air quality adjustment procedure that was used to simulate just meeting the current and alternative 8-hr standards, and the uncertainties associated with estimating background O_3 concentrations for each location. Based on sensitivity analyses conducted in the prior review, alternative air quality adjustment procedures had only a modest impact on the risk estimates. With respect to uncertainties about estimated background concentrations, as discussed in section 5.4.3, alternative assumptions about PRB levels have a variable impact depending on the location and standard analyzed in terms of the absolute magnitude of the risk estimates. Alternative assumptions about PRB levels have a greater impact on risk estimates associated with more stringent 8-hr standards. However, the overall pattern of reductions, expressed in terms of percentage reduction relative to the current standard, is generally unaffected by alternative assumptions for PRB for most of the standards analyzed.

With respect to the lung function part of the health risk assessment, key uncertainties include uncertainties in the exposure estimates for children engaged in moderate or greater exertion (noted above) and uncertainties associated with the shape of the exposure-response relationship, especially at levels below 0.08 ppm, 8-hr average, where only limited data are

available down to 0.04 ppm and there is an absence of data below 0.04 ppm. The uncertainty associated with the shape of the exposure-response relationship was considered using a Bayesian Markov Chain Monte Carlo approach as recommended by members of the CASAC panel (Henderson, 2006c). As discussed in section 5.3.2.5, the impacts of changing the functional form of the exposure-response relationship varied substantially. Changing from the 90% logistic/10% linear base case to the 80%/20% split generally had only a small impact for all school age children, with most risk estimates being within 5% of the base case estimates. The impacts of changing from the base case to the 50% logistic/50% linear case were generally (although not always) larger. We observed greater changes for all school age children between the 50/50 split and the base case in terms of percent change in risk for two more stringent alternative standards relative to the current standard. With respect to the lung function risk estimates for asthmatic children, there were relatively small changes observed between the 50/50 split and the base case in the percent changes associated with two more stringent alternative standards relative to the current standard.

With respect to the part of the health risk assessment based on effects reported in epidemiological studies, an important uncertainty for the mortality risk estimates is the extent to which the associations reported between O_3 and non-accidental and cardiorespiratory mortality actually reflect causal relationships. Other important uncertainties for this part of the risk assessment include uncertainties (1) surrounding estimates of O_3 coefficients in concentration-response functions used in the assessment, (2) concerning the specification of the concentration-response model (including the shape of the relationship) and whether or not a population threshold or non-linear relationship exists within the range of concentrations examined in the studies, (3) related to the extent to which concentration-response relationships derived from studies in a given location and time when O_3 levels were higher or behavior and/or housing conditions were different provide accurate representations of the relationships for the same locations with lower air quality distributions and different behavior and/or housing conditions, and (4) concerning the possible role of co-pollutants which also may have varied between the time of the studies and the current assessment period. For both parts of the risk assessment, statistical uncertainty due to sampling error has been characterized. As discussed in section 5.4.5, there are additional unquantified uncertainties including model uncertainty noted above.

While we and CASAC have recognized the various uncertainties that are inherent in conducting such risk assessments, CASAC found the health risk assessment to be "well done, balanced, and reasonably communicated" (Henderson, 2006c). We have considered the kind and extent of uncertainties in the health risk estimates, but judge that these estimates, discussed in Chapter 5 and in this chapter, are appropriate for consideration as an input to the decisions on the O_3 standard.

The exposure and risk assessments estimated occurrences of exposures of concern and lung function decrements, respectively, in school-age children ages 5 to 18. Due to the increased amount of time spent outdoors engaged in relatively high levels of physical activity, school-age children, as a group, are particularly at risk for experiencing O_3-related health effects. We report results for school-age children down to age five, but there is a trend for younger children to attend school. We are not taking these younger children into account in our analysis due to a lack of information which would let us characterize this group of children. Some states allow 4-year-olds to attend kindergarten, and more than 40 states have preschool programs for children younger than five (Blank and Mitchell, 2001). In 2000, six percent of U.S. children ages 3 to 19 who attend school were younger than five years old (2000 Census Summary File 3, Table QT-P19: School Enrollment). Clearly the estimates of exposures of concern and lung function decrements in school-age children would be higher had we been able to include this group of children in the exposure and risk assessments.

Exposure Assessment Results

The results of exposure assessments, which provide estimates of the number of people exposed to different levels of ambient O_3 while at prescribed exertion levels, serve two purposes. First, the entire range of modeled personal exposures to ambient O_3 is an essential input to the risk assessments based on exposure-response functions from controlled human exposure studies. Secondly, estimates of personal exposures to ambient O_3 levels at and above specific benchmark concentrations provide some perspective on the public health impacts of health effects that we cannot currently evaluate in quantitative risk assessments that may occur at current air quality levels, and the extent to which such impacts might be reduced by meeting the current and alternative standards. This is especially true when there are exposure levels at which we know or can reasonably infer that specific O_3-related health effects are occurring. We refer to exposures at and above these benchmark concentrations as "exposures of concern."

Estimating exposures of concern is important because it provides some indication of the potential public health impacts of a range of O_3-related health outcomes, such as pulmonary inflammation, increased airway responsiveness, or changes in host defenses.[4] These particular

[4] We note that estimates of the number of people likely to experience exposures of concern can not be directly translated into quantitative estimates of the number of people likely to experience specific health effects, since sufficient information to draw such comparisons is not available -- if such information were available, we would have included these health outcomes in the quantitative risk assessment As discussed in section 3.3.1.1.3, the studies reporting inflammatory responses and markers of lung injury have clearly acknowledged that there is significant variation in responses of subjects exposed, especially to 6.6 hour O_3 exposures at 0.08 and 0.10 ppm. To provide some perspective on the public health impacts of these health effects, we note that one study (Devlin et al., 1991, Figure 5), for example, showed that roughly 10 to 50% of the 18 young healthy adult subjects in the study experienced notable increases (i.e., $\geq$ 2 fold increase) in most of the inflammatory and cellular injury indicators

effects have been demonstrated in controlled human exposure studies of healthy individuals to occur at levels as low as 0.08 ppm O_3, but have not been evaluated at lower levels in controlled human exposure studies. We have not estimated these effects in a quantitative risk assessment due to the lack of adequate information on the overall exposure-response relationships. For the exposure analysis in the second draft Staff Paper, an exposure of concern was defined the same way as it was in the 1997 review, i.e., an 8-hr average exposure $\geq$ 0.08 ppm O_3 while engaged in intermittent moderate or greater exertion levels (62 FR 38860). However, at the August 2006 CASAC meeting, the CASAC O_3 Panel encouraged staff to also consider exposures of concern at levels lower than 0.08 ppm, expressing the view that there is a range of health effects that likely occur below the lowest level tested for these effects (U.S. EPA 2006e, p.104-105). There is no reason to assume that there is a threshold for effects, such as markers of inflammation, at 0.08 ppm O_3. Moreover, Panel members noted there is evidence of adverse effects that are strongly correlated with pulmonary inflammation, such as ED visits observed in epidemiological studies, at levels well below 0.08 ppm.

We concur with these views, and note that exposures of concern, and the health outcomes they represent, likely occur across a range of O_3 exposure levels, such that there is no one exposure level that addresses all relevant public health concerns. Therefore, we have estimated exposures of concern not only at 0.080 ppm O_3, a level at which there are demonstrated effects, but also at 0.070 and 0.060 ppm O_3, recognizing that there is no apparent threshold for O_3 health effects and that potentially adverse lung function decrements have been demonstrated in controlled human exposure studies of healthy individuals at 0.060 $\pm$ 0.003 ppm O_3. Moreover, there will be varying degrees of concern about exposures at each of these levels, based in part on the population subgroups experiencing them. For example, it is reasonable to conclude that a high degree of protection is warranted against effects that have been clearly demonstrated in healthy people, not only for the general public, but especially for members of sensitive subgroups, such as children or people with asthma and other lung diseases. At levels where effects have not been demonstrated in controlled human exposure studies but there is reason to infer that effects likely occur, or where the evidence is less clear, the appropriate level of protection will depend on the strength of the evidence and the adversity of the effect. At comparable levels of uncertainty in the evidence, it is important to provide stronger protection against effects that are more clearly adverse. Given that there is clear evidence of inflammation, increased airway responsiveness, and changes in host defenses in healthy people exposed to 0.080 $\pm$ 0.004 ppm O_3 and reason to infer that such effects will continue at lower exposure

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analyzed associated with 6.6-hour exposures at 0.08 ppm. We also note that susceptible subpopulations such as those with asthma may be even more affected.

levels, but with increasing uncertainty about the extent to which such effects occur at lower O_3 concentrations, in the following discussion we present information on all three levels of exposures of concern, with a focus on exposures of concern at levels ≥ 0.070 and ≥ 0.060 ppm O_3.

Exposure estimates for 6 scenarios (i.e., recent air quality for each of the three years examined, 2002 to 2004, and just meeting the current standard in each of the same three years based on adjusting air quality in this three-year period to determine the amount of adjustment needed) aggregated across 12 urban areas, have been developed for each of these exposure-of-concern levels. These estimates are shown in Table 6-1. The exposure estimates are for the number and percent of people exposed, in each of the population subgroups, and the number of person-days (occurrences) of exposures, with daily 8-hr maximum average exposures at or above 0.080 ppm (Table 6-1a), 0.070 ppm (Table 6-1b), and 0.060 ppm (Table 6-1c), while at intermittent moderate or greater exertion.[5]

As shown in Table 6-1, the patterns of exposures in terms of percentages of the population exceeding a given exposure level are very similar for the general population and for all and asthmatic school age (5-18) children, although children are about twice as likely to be exposed, based on the percent of the population exposed, at any given level. Thus, in the discussion below, we focus on the patterns observed for all school age children, which includes asthmatic children, with the recognition that these exposure patterns apply to this other subpopulation. While Table 6-1 shows aggregate results, it is important to note that there is substantial variability in the percent of the population subgroups estimated to experience exposures of concern across the 12 urban areas. For example, in the case of exposures of concern ≥ 0.070 ppm O_3, for 2002 when the current standard is just met, while the aggregate estimate is 18% (Table 6-1b), the estimates of exposures for all children range from about 1% to more than 35% of the population across the 12 urban areas analyzed (see Figure 4-7). Variability in the degree of health protection offered across urban areas is an important factor to consider in evaluating the adequacy of the current standard.

[5]Information in Table 6-1 is drawn from Appendix 4A.

Table 6-1a. Summary of Estimates of Number of People Exposed and Number of Occurrences1 at Moderate Exertion2 Associated with 8-Hour Daily Maximum Ozone Concentrations > 0.080 ppm for 12 Urban Areas in the U.S.

Air Quality Scenario	General Population (88.5 million people)		All Children (5-18 years old) (18.3 million children)		Asthmatic Children (5-18 years old) (2.6 million children)	
	Persons (% of population)	Person Days [% change from recent air quality]	Persons (% of population)	Person Days [% change from recent air quality]	Persons (% of population)	Person Days [% change from recent air quality]
2002 Air Quality						
Recent Air Quality	10,790,000 (12%)	15,030,000	4,130,000 (23%)	6,030,000	630,000 (24%)	940,000
Just Meeting Current Standard	2,020,000 (2%)	2,270,000 [85% reduction]	700,000 (4%)	800,000 [87% reduction]	110,000 (4%)	130,000 [86% reduction]
2003 Air Quality						
Recent Air Quality	5,970,000 (7%)	8,320,000	2,430,000 (13%)	3,500,000	340,000 (13%)	470,000
Just Meeting Current Standard	470,000 (0.5%)	480,000 [94% reduction]	160,000 (0.9%)	140,000 [95% reduction]	20,000 (1%)	30,000 [95% reduction]
2004 Air Quality						
Recent Air Quality	2,350,000 (3%)	3,000,000	910,000 (5%)	1,180,000	130,000 (5%)	170,000
Just Meeting Current Standard	80,000 (0.1%)	90,000 [97% reduction]	30,000 (0.1%)	30,000 [98% reduction]	3,000 (0.2%)	3,000 [98% reduction]

[1]Estimates for persons and person days are rounded to the nearest 10,000. Percentages greater than or equal to 1 are rounded to the nearest percent and percentages less than one are rounded to the nearest tenth.

[2]Moderate exertion is defined as having an 8-hr average equivalent ventilation rate (EVR) in the range 13-27 l-min/m^2.

Table 6-1b. Summary of Estimates of Number of People Exposed and Number of Occurrences[1] at Moderate Exertion[2] Associated with 8-Hour Daily Maximum Ozone Concentrations ≥ 0.070 ppm for 12 Urban Areas in the U.S.

Air Quality Scenario	General Population (88.5 million people)		All Children (5-18 years old) (18.3 million children)		Asthmatic Children (5-18 years old) (2.6 million children)	
	Persons (% of population)	Person Days [% change from recent air quality]	Persons (% of population)	Person Days [% change from recent air quality]	Persons (% of population)	Person Days [% change from recent air quality]
2002 Air Quality						
Recent Air Quality	22,770,000 (26%)	44,050,000	8,550,000 (47%)	18,500,000	1,280,000 (50%)	2,830,000
Just Meeting Current Standard	8,890,000 (10%)	12,350,000 [72% reduction]	3,340,000 (18%)	4,880,000 [74% reduction]	520,000 (20%)	770,000 [73% reduction]
2003 Air Quality						
Recent Air Quality	14,100,000 (16%)	24,390,000	5,510,000 (30%)	10,380,000	790,000 (31%)	1,430,000
Just Meeting Current Standard	2,480,000 (3%)	2,710,000 [89% reduction]	1,000,000 (5%)	1,090,000 [90% reduction]	150,000 (6%)	160,000 [89% reduction]
2004 Air Quality						
Recent Air Quality	7,660,000 (9%)	12,040,000	3,020,000 (17%)	5,020,000	440,000 (17%)	710,000
Just Meeting Current Standard	720,000 (0.8%)	770,000 [94% reduction]	260,000 (1%)	270,000 [95% reduction]	40,000 (1%)	40,000 [95% reduction]

[1]Estimates for persons and person days are rounded to the nearest 10,000. Percentages greater than or equal to 1 are rounded to the nearest percent and percentages less than one are rounded to the nearest tenth.

[2]Moderate exertion is defined as having an 8-hr average equivalent ventilation rate (EVR) in the range 13-27 l-min/m^2.

Table 6-1c. Summary of Estimates of Number of People Exposed and Number of Occurrences[1] at Moderate Exertion[2] Associated with 8-Hour Daily Maximum Ozone Concentrations ≥ 0.060 ppm for 12 Urban Areas in the U.S.

Air Quality Scenario	General Population (88.5 million people)		All Children (5-18 years old) (18.3 million children)		Asthmatic Children (5-18 years old) (2.6 million children)	
	Persons (% of population)	Person Days [% change from recent air quality]	Persons (% of population)	Person Days [% change from recent air quality]	Persons (% of population)	Person Days [% change from recent air quality]
2002 Air Quality						
Recent Air Quality	36,410,000 (41%)	112,510,000	12,580,000 (69%)	48,000,000	1,860,000 (72%)	7,260,000
Just Meeting Current Standard	21,860,000 (25%)	45,040,000 [60% reduction]	7,970,000 (44%)	18,760,000 [61% reduction]	1,210,000 (47%)	2,910,000 [60% reduction]
2003 Air Quality						
Recent Air Quality	27,660,000 (31%)	68,790,000	10,220,000 (56%)	29,640,000	1,490,000 (58%)	4,170,000
Just Meeting Current Standard	10,410,000 (12%)	14,220,000 [79% reduction]	4,040,000 (22%)	5,750,000 [81% reduction]	610,000 (24%)	860,000 [79% reduction]
2004 Air Quality						
Recent Air Quality	19,820,000 (22%)	44,620,000	7,570,000 (41%)	19,300,000	1,090,000 (42%)	2,690,000
Just Meeting Current Standard	4,720,000 (5%)	5,920,000 [87% reduction]	1,800,000 (10%)	2,300,000 [88% reduction]	270,000 (11%)	350,000 [87% reduction]

[1]Estimates for persons and person days are rounded to the nearest 10,000. Percentages greater than or equal to 1 are rounded to the nearest percent and percentages less than one are rounded to the nearest tenth.

[2]Moderate exertion is defined as having an 8-hr average equivalent ventilation rate (EVR) in the range 13-27 l-min/m^2.

In addition to city-to-city variability, substantial year-to-year variability in exposure estimates is observed. For example, about 3 million children (17% of all school age children for the 12 urban areas) to more than 8.5 million children (47% of all school age children for the 12 urban areas), are estimated to experience one or more exposures of concern (i.e., $\geq$ 0.070 ppm, while engaged in moderate or greater exertion) for 2004 and 2002, respectively.[6] When air quality is adjusted to simulate just meeting the 8-hr standard, the estimated number of children exposed is substantially reduced. Depending on which year is adjusted for just meeting the current standard, approximately 260,000 children (1% of all school age children) (based on 2004 air quality data) to more than 3.3 million (18% of all school age children) (based on 2002 air quality data) are estimated to experience one or more exposures of concern. These results suggest reductions of approximately 91%, based on 2004 air quality, to 61%, based on 2002 air quality, in the number of children estimated to experience one or more 8-hr average exposures above 0.070 ppm when the current 8-hr O_3 standard is just met.

We have also examined the extent to which individuals are likely to experience repeated exposures of concern, which is an important aspect to consider when making judgments about the extent to which these exposures can be considered adverse to individuals. However, based on the analysis described in section 4.6.4, staff concludes that the APEX exposure model significantly underestimates the frequency of occurrences of individuals (adults and children) experiencing repeated 8-hour average exposures at the three levels. As discussed in Chapter 4, this underestimation results from the way that people's activities are modeled in APEX using CHAD, which does not properly account for repeated behavior of adults and children, and may be the greatest source of uncertainty in the exposure estimates. Thus, it is likely that the number of repeated exposures, which are estimated here to be relatively small, is significantly underestimated. As seen in Table 6-1, for air quality just meeting the current standard, in most cases the estimated number of occurrences is only slightly larger than the estimated number of people exposed, indicating that the estimated number of repeated exposures is relatively small. Moreover, due to limitations in the CHAD database, the exposure assessment does not include outdoor workers, some proportion of whom are likely to be exposed repeatedly day after day to elevated ambient O_3 levels while at work.

Risk Assessment Results

Turning to risk-based considerations, as discussed in Chapter 5, risk estimates have been calculated and are discussed below for several important health endpoints, including:

[6]Unless otherwise noted, estimates for 2003 fall between the estimates for 2002 and 2004.

- Lung function decrements (i.e., ≥ 15% and ≥ 20% reductions in FEV_1) in all school age children for 12 urban areas;
- Lung function decrements (i.e., ≥ 10% and ≥ 20% reductions in FEV_1) in asthmatic school age children for 5 urban areas (a subset of the 12 urban areas);
- Respiratory symptoms (i.e., chest tightness, shortness of breath, wheeze) in moderate to severe asthmatic children for the Boston area;
- Respiratory-related hospital admissions for 3 urban areas;
- Non-accidental and cardiorespiratory mortality for 12 urban areas.

In the sections on lung function decrements in children (all and asthmatic children), we describe the scope of the assessments, the number and percent of children experiencing moderate and large lung function decrements, the number of occurrences of moderate and large lung function decrements, and what risk is estimated to remain after the current 8-hr standard is met. For effects such as lung function decrements, which are transient and reversible, aspects such as the likelihood that these effects would interfere with normal activities or occur repeatedly are important to consider in making judgments about adversity to individuals and are described below. There are discussions about year-to-year variability in the results and variability in results across the urban areas, which are important for making judgments about public health impacts. These estimates indicate that there are substantial differences in the natural fluctuation of air quality levels from year to year. This can result in significant variability in the number of children affected by moderate or greater lung function decrements and the number of occurrences they experience during a 3-year period that is adjusted to just meet the current 8-hr standard. For example, the number of children affected, and the number of occurrences, can increase by more than 100% for a year with generally poorer air quality compared to a year with better air quality, within a three year period.

Other aspects of the risk information presented are important to consider. The first is that there is some degree of consistency in the estimated population risk across the 12 urban areas, as indicated by the percent of the population estimated to be affected, which describes the risk normalized across the urban areas with very different population sizes. In Table 6-2, the percent of all children likely to experience one or more moderate or greater lung function responses (≥ 15% reduction in FEV_1) under recent air quality and when air quality just meets the current 8-hr standard are 7% and 3% (based on 2002 air quality), respectively. The range across the 12 urban areas, from Table 5-8 is approximately 5% to 10% under recent (2002) air quality, and about 1% to 5% when air quality is adjusted to just meet the current 8-hr standard based on that year. The pattern across the 12 urban areas is similar for the risk estimates based on 2004 air quality.

The remaining sections on respiratory symptoms in moderate to severe asthmatic children, respiratory related hospital admissions, and non-accidental and cardiorespiratory

mortality include discussions of the scope of the assessments, provide estimates of the incidence, incidence per 100,000 and percent of total incidence of the effects, and discussion of year-to-year variability in the estimates and the risk that remains after the current standard is met.

Table 6-2. Summary of Number and Percent of All School Age Children (5-18) in 12 Urban Areas Estimated to Experience Lung Function Responses and the Number of Occurrences[1] Associated with 8-Hour Ozone Exposures While Engaged in Moderate Exertion[2] for 2002, 2003, and 2004 Air Quality and Just Meeting the Current 8-Hour Standard[3]

Air Quality Scenario	$FEV_1 \geq 15\%$		$FEV_1 \geq 20\%$	
	Children (% of Children, 5-18)	Occurrences [% reduction from recent air quality]	Children (% of Children, 5-18)[1]	Occurrences [% reduction from recent air quality]
Recent Air Quality (2002)	1,330,000 (7%)	6,160,000	530,000 (3%)	1,370,000
Just Meeting Current Standard	610,000 (3%)	3,170,000 [49% reduction]	170,000 (0.9%)	500,000 [64% reduction]
Recent Air Quality (2003)	990,000 (5%)	4,540,000	370,000 (2%)	920,000
Just Meeting Current Standard	350,000 (2%)	1,920,000 [58% reduction]	80,000 (0.4%)	230,000 [75% reduction]
Recent Air Quality (2004)	620,000 (3%)	3,570,000	190,000 (1%)	570,000
Just Meeting Current Standard	230,000 (1%)	1,620,000 [55% reduction]	40,000 (0.2%)	160,000 [71% reduction]

[1]Estimates for persons and person days greater than 10,000 were rounded to the nearest 10,000. Estimates for persons and person days less than 10,000 were rounded to the nearest thousand. Percentages less than 1 are rounded to the nearest tenth, percentages greater than or equal to 1 are rounded to the nearest percent.

[2]Moderate exertion is defined as having an 8-hr average equivalent ventilation rate $\geq$ 13 l-min/m^2.

[3]Estimates are the aggregate results based on 12 urban areas (Atlanta, Boston, Chicago, Cleveland, Detroit, Houston, Los Angeles, New York, Philadelphia, Sacramento, St. Louis, and Washington, D.C.) based on Tables 5-6 and 5-7 for estimates based on 2002 and 2004 air quality and tables provided in Post (2007) for estimates based on 2003 air quality. Estimates are for the ozone season which is all year in Houston, Los Angeles and Sacramento and March or April to September or October for the remaining urban areas.

Lung function decrements in all school-age children. Tables 5-6 and 5-7 in Chapter 5 display risk estimates for all school age children (ages 5-18) for moderate or greater lung function decrement responses for the 12 urban areas. As with the exposure estimates, the risk estimates are associated with three years of recent air quality (i.e., 2002, 2003, 2004) and air quality based on adjusting these same three years to simulate just meeting the current 0.08 ppm, 8-hr O_3 standard based on the three-year design value. All estimates in both tables reflect

responses associated with exposure to O_3 in excess of exposures associated with policy relevant background (PRB) O_3 concentrations.[7] Table 5-6 shows the number and percent of all school age children estimated to have at least one moderate or greater lung function response (i.e., $\geq$ 15% decrement in FEV_1) during the O_3 season. Table 5-7 displays the total number of occurrences for the moderate or greater lung function responses during the O_3 season. Table 6-2 draws upon the risk estimates for all school age children contained in tables in Chapter 5 and the Risk Assessment TSD, and provides the number of children estimated to experience one or more occurrences of moderate or greater (i.e., $\geq$ 15% decrement in FEV_1) and large or greater (i.e., $\geq$ 20% decrement in FEV_1) lung function responses, and the number of total days of occurrences, aggregated across all 12 urban areas.

As shown in Table 6-2, for the three recent years (2002 - 2004), from 620,000 to over 1.3 million school age children (3 to 7% of all school age children) are estimated to experience 1 or more moderate lung function responses (i.e., $\geq$ 15% reduction in FEV_1) in the 12 urban areas combined. Similar to the exposure estimates discussed above, when air quality is adjusted to simulate just meeting the current 8-hr standard, there are significant reductions in estimated health outcomes. Depending on which year is adjusted to just meet the current 8-hr standard, from about 230,000 to about 610,000 children (1 to 3% of all school age children) are estimated to experience moderate (i.e., $\geq$ 15% reduction in FEV_1) lung function responses in these 12 urban areas combined upon just meeting the current standard. Among all school age children, these estimates indicate that the percent of children likely to experience one or more moderate or greater lung function decrements (i.e., $\geq$ 15% FEV_1 decrement) is reduced by about 54 to 65% when the current standard is just met based on adjusting for 2002, 2003, and 2004 air quality. The analogous reduction in the number of occurrences of lung function decrements $\geq$ 15% is 49 to 58%.

It is also important to note that many of these children will experience repeated occurrences of moderate or greater lung function responses. These results indicate that among all school age children, on average an individual is likely to experience 5 to 7 occurrences of moderate or greater lung function responses during an O_3 season.[8] However, based on the distribution of exposures estimated from the 1997 review, it is reasonable to expect that many

[7]With respect to the impact of uncertainties about estimated background concentrations on the risk estimates presented in Tables 5-6 and 5-7, as discussed in section 5.4.3, alternative assumptions about PRB levels had a variable impact depending on the location and standard analyzed in terms of the absolute magnitude of the risk estimates. However, the overall pattern of reductions, expressed in terms of percentage reduction relative to the current standard, is not impacted by alternative assumptions for PRB for most of the alternative standards analyzed.

[8]This number is estimated for example for all children, by dividing the estimated number of children into the estimated number of occurrences, resulting in an average of about 5 to 7 occurrences per child.

children will experience one or just a few moderate or greater lung function responses, while a smaller number of children will experience large numbers of such responses. These ranges of estimated number of occurrences of moderate or greater lung function decrements in an O_3 season are important in considering the implications for the health status of individuals likely to experience these effects.

As discussed in section 3.6.3, for active healthy people, large lung function responses (i.e., $\geq$ 20% decrement in FEV_1), would likely interfere with normal activities in many sensitive individuals, therefore single occurrences would be considered to be adverse under ATS guidelines, and are the appropriate indicator to consider. As shown in Table 6-2, for the three recent years (2002 - 2004), there were estimated to be from about 1,370,000 to about 570,000 occurrences of large lung function decrements (i.e., $\geq$ 20% reduction in FEV_1) in school age children in the 12 urban areas combined. Similar to the exposure estimates discussed above, when air quality is adjusted to simulate just meeting the current 8-hr standard, there are significant reductions in estimated health outcomes. Depending on which year is considered, upon just meeting the current 8-hr standard, there are estimated to be from about 500,000 to about 160,000 occurrences of large lung function decrements (i.e., $\geq$ 20% reduction in FEV_1) in school age children in these 12 urban areas combined. Among all school age children, these estimates indicate that occurrences of large lung function decrements (i.e., $\geq$ 20% FEV_1 decrement) are reduced by about 64 to 75% when the current standard is just met based on adjusting 2002, 2003, and 2004 air quality.

Lung function decrements in asthmatic school age children. As discussed in greater detail in section 3.6.3, FEV_1 decrements $\geq$ 10% but < 20% have been judged to represent moderate levels of functional responses for active healthy people and would likely interfere with normal activity for relatively few sensitive individuals. However, for persons with lung disease, such as asthma, lung function decrements at the lower end of the moderate range (i.e., FEV_1 decrements $\geq$ 10%) would likely interfere with normal activity for many individuals and would likely result in additional and more frequent use of medication. We also note that new evidence described above indicates that children with asthma, particularly those with moderate-to-severe asthma, are more likely to have lung function and symptomatic responses, and have bigger responses, from O_3 exposure than children who do not have asthma. Studies discussed in section 3.3.1.1 that show increased lung function responses, inflammation, and increased airway responsiveness in asthmatics indicate that the risk estimates for lung function decrements derived from controlled exposures of healthy adult volunteers likely underestimate the percent of asthmatic school age children that would experience decrements in FEV_1. In this final Staff Paper, we use a $\geq$ 10% decrease in FEV_1 as a benchmark for moderate functional responses in asthmatic children. The CASAC endorsed this approach. Thus, as discussed in section 5.4.1,

consistent with the CASAC recommendation (Henderson, 2006c) that for asthmatic children a lung function response defined in terms of FEV_1 decrement ≥ 10% serves as an indicator of potential adverse health effects for this group, risk estimates have been calculated for lung function decrements (i.e., ≥ 10% and ≥ 20% reductions in FEV_1) in asthmatic school age children for 5 urban areas that are a subset of the 12 urban areas included for all children.

As shown in Table 6-3, for the three recent years (2002 - 2004), from about 160,000 to about 250,000 asthmatic school age children (9 to 15% of asthmatic school age children) are estimated to experience 1 or more moderate lung function responses (i.e., ≥ 10% reduction in FEV_1) in the 5 urban areas combined. Similar to the risk estimates for all school age children discussed above, when air quality is adjusted to simulate just meeting the current 8-hr standard, there are significant reductions in estimated health outcomes. Depending on which year is adjusted to just meet the current 8-hr standard, from about 70,000 to about 130,000 children (4 to 8% of asthmatic school age children) are estimated to experience moderate (i.e., ≥ 10% reduction in FEV_1) lung function responses in these 5 urban areas combined upon just meeting the current standard. Among asthmatic school age children, these estimates indicate that the number of children likely to experience one or more moderate or greater lung function decrements (i.e., ≥ 10% FEV_1 decrement) drops by about 50 to 60% when the current standard is just met based on adjusting 2002, 2003, and 2004 air quality.

Asthmatic school age children are estimated to experience a greater number of repeated occurrences of moderate or greater lung function responses per individual responding compared to all school age children. The results in Table 6-3 indicate that among asthmatic school age children, on average an individual is likely to experience 8 to 10 occurrences of moderate or greater lung function responses during an O_3 season. As discussed above, the more likely distribution is that many children will experience one or only a few occurrences of moderate or greater lung function decrements (≥ 10% decrement in FEV_1), while some may experience a very large number, based on these estimates. Recognizing that nationally over 14% of school age children have asthma, these numbers raise concern about the potential number of children with asthma who could experience a large number of occurrences of moderate or greater lung function decrements (≥ 10% decrement in FEV_1) even with air quality just meeting the current 8-hr standard.

Table 6-3. Summary of Number and Percent of Asthmatic School Age Children (5-18) in 5 Urban Areas Estimated to Experience Lung Function Responses and the Number of Occurrences[1] Associated with 8-Hour Ozone Exposures While Engaged in Moderate Exertion[2] for 2002, 2003, and 2004 Air Quality and Just Meeting the Current 8-Hour Standard[3]

Air Quality Scenario	$FEV_1 \geq 10\%$		$FEV_1 \geq 20\%$	
	Children (% of Asthmatic Children in)[1]	Occurrences[1] [% reduction from recent air quality][1]	Children (% of Asthmatic Children)[1]	Occurrences[1] [% reduction from recent air quality][1]
Recent Air Quality (2002)	250,000 (16%)	1,890,000	50,000 (3%)	120,000
Just Meeting Current Standard	130,000 (8%)	1,040,000 [45% reduction]	10,000 (0.7%)	40,000 [70% reduction]
Recent Air Quality (2003)	210,000 (13%)	1,590,000	40,000 (2%)	90,000
Just Meeting Current Standard	90,000 (6%)	760,00 0 [53% reduction]	6,000 (0.4%)	20,000 [79% reduction]
Recent Air Quality (2004)	160,000 (10%)	1,390,000	20,000 (1%)	60,000
Just Meeting Current Standard	70,000 (4%)	680,000 [51% reduction]	3,000 (0.2%)	10,000 [76% reduction]

[1]Estimates for persons and person days greater than 10,000 were rounded to the nearest 10,000. Estimates for persons and person days less than 10,000 were rounded to the nearest thousand. Percentages less than 1 are rounded to the nearest tenth, percentages greater than or equal to 1 are rounded to the nearest percent.

[2]Moderate exertion is defined as having an 8-hr average equivalent ventilation rate $\geq$ 13 l-min/m^2

[3]Estimates are the aggregate results based on 5 urban areas (Atlanta, Chicago, New York, Houston, and Los Angeles) based on Tables 5-8 and 5-9 for estimates based on 2002 and 2004 air quality and Tables 3-8, 3-9, 3-24, 3-26, and 3-27 in the Risk Assessment TSD for estimates based on 2003 air quality. Estimates are for the ozone season which is all year in Houston, and Los Angeles, and March or April to September or October for the remaining urban areas.

As discussed in section 3.6.3, for people with asthma, large lung function responses (i.e., $\geq$ 20% decrement in FEV_1), would likely interfere with normal activities for most individuals and would also increase the likelihood that these individuals would use additional medication or seek medical treatment. Single occurrences would be considered to be adverse to the individuals and would be cause for concern. As shown in Table 6-3, for the three recent years (2002 - 2004), there were estimated to be from about 120,000 to about 60,000 occurrences of large lung function responses (i.e., $\geq$ 20% reduction in FEV_1) in asthmatic school age children in the 5 urban areas combined. When air quality is adjusted to simulate just meeting the current 8-hr standard, there are significant reductions in estimated health outcomes. Depending on which

year is adjusted to just meet the current 8-hr standard, there are estimated to be from about 30,000 to about 10,000 occurrences of large lung function responses (i.e., ≥ 20% reduction in FEV_1) in asthmatic children in these 5 urban areas combined. Among asthmatic school age children, these estimates indicate that occurrences of large lung function decrements (i.e., ≥ 20% FEV_1 decrement) in these 5 urban areas are reduced by about 70 to 79% when the current standard is just met based on adjusting 2002, 2003, and 2004 air quality.[9]

Respiratory symptoms in moderate to severe asthmatic children. Risk estimates were developed for several respiratory symptoms (i.e., chest tightness, shortness of breath, and wheeze) during the O_3 season in children of age 0 to 12 years with moderate to severe asthma (as defined by the use of maintenance asthma medications), living in the Boston area.[10] About 40% of the children with asthma in the Boston area are estimated to be on controller medications and would be considered moderate-to-severe asthmatics.[11] In this population of 25,000 children with moderate-to-severe asthma, as shown in Tables 5-10 and 5-11, the estimated incidence of symptom days of chest tightness (across 4 models reflecting 2 different lags and O_3 alone vs. inclusion of $PM_{2.5}$ in the model) ranges from almost 6,900 to 10,800 based on a year (2002) with poorer air quality, and from 5,300 to 8,400 based on a year with better (2004) air quality.

As indicated in Figure 5-10, the current standard reduces the incidence of symptom days for chest tightness by relatively small and consistent amounts across the 4 models specified. Risk estimates for the other symptom endpoints, shortness of breath and wheeze, show similar patterns as the risk estimates for chest tightness. The reduction of risks across the 4 models for chest tightness is shown in Table 6-4. Averaging the median estimates of symptom days indicates that just meeting the current 8-hr standard is estimated to reduce the total number of symptom days for chest tightness in children with moderate to severe asthma by 11% (8,700 to 7,700) based on a year (2002) with generally poorer air quality and by 15% (from 6,700 to 5,700) based on a year (2004) with generally better air quality. The current standard clearly does not provide the same degree of protection against respiratory symptoms in moderate to severe

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[9]Risk estimates for all and asthmatic children school age children (i.e., ≥ 15% and ≥ 10% reductions in FEV_1, respectively) are discussed in more detail in Chapter 5 and in the Risk Assessment TSD. Risk estimates for large lung function decrements in all and asthmatic children (i.e., ≥ 20% reductions in FEV_1), are included in the Risk Assessment TSD.

[10]To minimize uncertainty, this risk assessment was performed for the Boston area because that is the urban area nearest to where the epidemiological study was conducted that is the basis for the exposure-response function used in the assessment.

[11]The estimated percent of asthmatic children using maintenance medications (40%) was obtained via email 4-05-06 from Jeanne E. Moorman, Survey Statistician, National Center for Environmental Health, CDC. The email communication has been placed in the docket for this review.

asthmatic children as it provides against moderate or greater lung function decrements in all children.

Looking at percent of total incidence of symptom days, even after the current 8-hr standard is met in a year with generally better air quality, among children with moderate to severe asthma in the Boston area, as many as one symptom day in 8 during the O_3 season is estimated to be attributable to O_3 exposure. In a year with generally poorer air quality, as many as one symptom day in 6 is estimated to be attributable to O_3 exposure. These results support the human clinical and epidemiological evidence that people with asthma are more likely to experience effects related to O_3 exposure than the general population, and provide evidence that the current 8-hr O_3 standard is not as protective for children with moderate to severe asthma in the Boston area as it is for all school age children in the 12 urban areas evaluated.

Respiratory-related hospital admissions in New York City. For unscheduled hospital admissions, risk estimates for the New York City area[12] associated with O_3 levels above background for the period from April to September are shown in Table 6-5 for recent air quality (2002, 2004), and for just meeting the current 8-hr standard based on adjusting a recent 3-year period (2002-2004). The current 8-hr standard reduces the incidence of respiratory-related hospital admissions by about 16% in a year with poorer air quality (2002) and about 18% in a year with better air quality (2004). The incidence of asthma-related hospital admissions (which are a subset of total respiratory hospital admissions) were reduced by about the same amount in each of the two scenarios. This results in an incidence per 100,000 of 4.6 to 6.4 for respiratory-related hospital admissions, and 3.9 to 5.5 for asthma-related hospital admissions, based on two air quality years, after the current standard is met.

[12]To minimize uncertainty, this risk assessment was performed for the New York City area because that is where the epidemiological study was conducted that is the basis for the exposure-response function used in the assessment.

Table 6-4. Incidence of Respiratory Symptom Days for Chest Tightness Associated with Recent (2004, 2002) Air Quality and Just Meets the Current Standard Based on Adjusting 2004 and 2002 Air Quality in Moderate to Severe Asthmatic Children in Boston, MA

Respiratory Symptoms in Moderate to Severe Asthmatic Children on Controller Meds	Year	Average Incidence of Chest Tightness Associated with Air Quality (range of median estimates)[1,2] [% reduction from recent air quality]	
		Recent Air Quality	Just Meets 0.08 ppm
Incidence	2002	8,700 (6,900 - 10,800)	7,700 [11% reduction] (6,100 - 9,600)
	2004	6,700 (5,300 - 8,400)	5,700 [15% reduction] (4,500 - 7,200)
Incidence per 100,000	2002	34,400 (27,200 - 42,700)	30,600 (24,100 - 38,100)
	2004	26,400 (20,700 - 33,100)	22,600 (17,700 - 28,400)
Percent of Total Incidence	2002	12% - 20%	11% - 17%
	2004	9% - 15%	8% - 13%

[1] Incidence rounded to nearest 100. Percentages rounded to nearest tenth.

[2] Range of median estimates across models using lag 0 and lag 1 day and O_3 only and including $PM_{2.5}$ in the model.

Total non-accidental and cardiorespiratory mortality in 12 urban areas. Table 6-6 summarizes risk estimates for non-accidental mortality in 12 urban areas associated with O_3 levels above background for the period from April to September based on the 95-city function reported in Bell et al. (2004) for non-accidental mortality. This table includes risks for two recent years of air quality (2002 and 2004) and risks associated with just meeting the current 8-hr standard over a recent 3-year period (2002-2004).[13] We chose to present the multi-city function risk estimates here because they are available for all 12 urban areas, while single-city estimates

[13]The information presented in Table 6-6 is based on Tables 5-10 and 5-11 in this Staff Paper which summarize the risk estimates for non-accidental mortality in 12 urban areas for recent air quality (2002, 2004) and Tables 5C-13 to 5C-16 in Appendix 5C of this Staff Paper and Tables 4-15 and 4-18 of the Risk Assessment TSD which present risk estimates for just meeting the current 8-hr standard based on adjusting the 3-year period (2002-2004).

are only available for 7 of the 12 urban areas and because the multi-city risk estimates are less subject to publication bias. In comparing estimates between recent air quality and just meeting the current standard, similar patterns are seen in terms of relative reductions regardless of whether single- or multi-city functions are used.[14] Across the 12 urban areas, the estimates of mortality incidence per 100,000 relevant population range from 0.4 to 2.8 (for 2002) and from 0.5 to 1.5 (for 2004). Meeting the current standard results in a reduction of the incidence per 100,000 to a range of 0.3 to 2.4 based on adjusting 2002 air quality and a range of 0.3 to 1.2 based on adjusting 2004 air quality. Estimates for cardiorespiratory mortality show similar patterns (Tables 5-14, 5-15).

[14]Additional risk estimates for cardiorespiratory mortality are included in the Risk Assessment TSD for 8 of the 12 urban areas based on Huang et al. (2005).

Table 6-5. Risks of Respiratory- and Asthma-related Hospital Admissions Associated with Recent (2004, 2002) Air Quality and Air Quality Adjusted to Just Meets Current Standard Based on Adjusting 2004 and 2002 Air Quality in New York City, NY

Unscheduled Hospital Admissions	Air Quality Scenario	Incidence[1] (Range)[2] [% reduction from recent]		Incidence per 100,000 (Range)[2]		Percent Total Incidence (Range)[2]	
		2002	2004	2002	2004	2002	2004
Respiratory	Recent	610 (150 – 1070)	450 (110 – 790)	7.6 (1.8 - 13.3)	5.6 (1.4 - 9.8)	1.7% (0.4 - 3%)	1.3% (0.3 - 2.2%)
	Just Meets 0.08 ppm	510 (120 – 900) [16% reduction]	370 (90 - 640) [18 % reduction]	6.4 (1.5 - 11.3)	4.6 (1.1 - 8)	1.5% (0.4 - 2.6%)	1% (0.3 - 1.8%)
Asthma (subset of respiratory)	Recent	520 (110 – 930)	380 (80 – 680)	6.5 (1.4 - 11.6)	4.8 (1 - 8.5)	4% (0.8 - 7.1%)	2.9% (0.6 - 5.2%)
	Just Meets 0.08 ppm	440 (90 - 780) [16% reduction]	310 (70 - 560) [18% reduction]	5.5 (1.2 - 9.8)	3.9 (0.8 - 7)	3.3% (0.7 - 6%)	2.4% (0.5 - 4.3%)

[1]Incidence rounded to the nearest 10. Incidence per 100,000 and percent of total incidence rounded to the nearest tenth.

[2]Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient

Table 6-6. Risks of Non-accidental Mortality Associated with Recent (2004, 2002) Air Quality and Air Quality Adjusted to Just Meets Current Standard Based on Adjusting 2004 and 2002 Air Quality

Location	Air Quality Scenario	Estimated Risk of Non-accidental Mortality[1,2]					
		Incidence (range)[3]		Incidence per 100,000 (range)[3]		Percent of Total Incidence (range)[3]	
		2002	2004	2002	2004	2002	2004
Atlanta	Recent	17 (6 - 29)	12 (4 - 20)	1.2 (0.4 - 1.9)	0.8 (0.3 - 1.4)	0.4% (0.1 - 0.6%)	0.3% (0.1 - 0.4%)
	Just Meets 0.08 ppm	14 (5 - 23)	9 (3 - 15)	0.9 (0.3 - 1.6)	0.6 (0.2 - 1)	0.3% (0.1 - 0.5%)	0.2% (0.1 - 0.3%)
Boston	Recent	10 (3 - 17)	7 (2 - 12)	1.5 (0.5 - 2.5)	1.0 (0.3 - 1.7)	0.4% (0.1 - 0.7%)	0.3% (0.1 - 0.5%)
	Just Meets 0.08 ppm	9 (3 - 15)	6 (2 - 9)	1.3 (0.4 - 2.1)	0.8 (0.3 - 1.4)	0.3% (0.1 - 0.6%)	0.2% (0.1 - 0.4%)
Chicago	Recent	69 (23 - 115)	49 (16 - 81)	1.3 (0.4 - 2.1)	0.9 (0.3 - 1.5)	0.3% (0.1 - 0.5%)	0.2% (0.1 - 0.4%)
	Just Meets 0.08 ppm	55 (18 - 91)	33 (11 - 55)	1 (0.3 - 1.7)	0.6 (0.2 - 1)	0.3% (0.1 - 0.4%)	0.2% (0.1 - 0.3%)
Cleveland	Recent	38 (13 - 64)	17 (6 - 28)	2.8 (0.9 - 4.6)	1.2 (0.4 - 2)	0.5% (0.2 - 0.9%)	0.2% (0.1 - 0.4%)
	Just Meets 0.08 ppm	31 (10 - 52)	12 (4 - 20)	2.2 (0.8 - 3.7)	0.9 (0.3 - 1.4)	0.4% (0.1 - 0.7%)	0.2% (0.1 - 0.3%)
Detroit	Recent	29 (10 - 48)	17 (6 - 28)	1.4 (0.5 - 2.3)	0.8 (0.3 - 1.4)	0.3% (0.1 - 0.5%)	0.2% (0.1 - 0.3%)
	Just Meets 0.08 ppm	24 (8 - 39)	12 (4 - 20)	1.1 (0.4 - 1.9)	0.6 (0.2 - 1)	0.3% (0.1 - 0.4%)	0.1% (0 - 0.2%)
Houston	Recent	14 (5 - 24)	17 (6- 28)	0.4 (0.1 - 0.7)	0.5 (0.2 - 0.8)	0.2% (0.1 - 0.3%)	0.2% (0.1 - 0.3%)
	Just Meets 0.08 ppm	9 (3 - 15)	11 (4 - 18)	0.3 (0.1 - 0.4)	0.3 (0.1 - 0.5)	0.1% (0% - 0.2%)	0.1% (0% - 0.2%)

Location	Air Quality Scenario	Estimated Risk of Non-accidental Mortality[1,2]					
		Incidence (range)[3]		Incidence per 100,000 (range)[3]		Percent of Total Incidence (range)[3]	
		2002	2004	2002	2004	2002	2004
Los Angeles	Recent	110 (37 - 184)	133 (45 - 221)	1.2 (0.4 - 1.9)	1.4 (0.5 - 2.3)	0.4% (0.1 - 0.7%)	0.5% (0.2 - 0.8%)
	Just Meets 0.08 ppm	52 (17 - 86)	67 (22 - 111)	0.5 (0.2 - 0.9)	0.7 (0.2 - 1.2)	0.2% (0.1 - 0.3%)	0.2% (0.1 - 0.4%)
New York	Recent	105 (35 - 174)	60 (20 - 100)	1.2 (0.4 - 2)	0.7 (0.2 - 1.1)	0.3% (0.1 - 0.6%)	0.2% (0.1 - 0.3%)
	Just Meets 0.08 ppm	84 (28 - 139)	43 (15 - 72)	0.9 (0.3 - 1.6)	0.5 (0.2 - 0.8)	0.3% (0.1 - 0.4%)	0.2% (0.1 - 0.3%)
Philadelphia	Recent	37 (12 - 62)	23 (8 - 38)	2.4 (0.8 - 4.1)	1.5 (0.5 - 2.5)	0.5% (0.2 - 0.8%)	0.3% (0.1 - 0.5%)
	Just Meets 0.08 ppm	30 (10 - 50)	17 (6 - 28)	2 (0.7 - 3.3)	1.1 (0.4 - 1.8)	0.4% (0.1 - 0.6%)	0.2% (0.1 - 0.3%)
Sacramento	Recent	23 (8 - 39)	18 (6 - 29)	1.9 (0.6 - 3.2)	1.4 (0.5 - 2.4)	0.6% (0.2 - 0.9%)	0.4% (0.1 - 0.7%)
	Just Meets 0.08 ppm	18 (6 - 30)	12 (4 - 21)	1.5 (0.5 - 2.4)	1 (0.3 - 1.7)	0.4% (0.1 - 0.7%)	0.3% (0.1 - 0.5%)
St. Louis	Recent	6 (2 - 10)	3 (1 - 5)	1.7 (0.6 - 2.8)	0.9 (0.3 - 1.5)	0.3% (0.1 - 0.5%)	0.2% (0.1 - 0.3%)
	Just meets 0.08 ppm	5 (2 - 8)	2 (1 - 4)	1.4 (0.5 - 2.3)	0.7 (0.2 - 1.1)	0.2% (0.1% - 0.4%)	0.1% (0%- 0.2%)
Washington, DC	Recent	15 (5 - 25)	8 (3 - 14)	2.6 (0.9 - 4.4)	1.5 (0.5 - 2.4)	0.6% (0.2 - 0.9%)	0.3% (0.1 - 0.5%)
	Just Meets 0.08 ppm	14 (5 - 23)	7 (2 - 12)	2.4 (0.8 - 3.9)	1.2 (0.4 - 2.1)	0.5% (0.2 - 0.8%)	0.3% (0.1 - 0.4%)

[1]All results are for mortality (among all ages) associated with short-term exposures to O_3. All results are based on single-pollutant model from Bell et al. (2004) 95-cities model.

[2]Incidence was quantified down to estimated policy relevant background levels. Incidences are rounded to the nearest whole number; incidences per 100,000 relevant population and percents are rounded to the nearest tenth.

[3]Note: Numbers in parentheses are 95% confidence or credible intervals based on statistical uncertainty surrounding the O_3 coefficient.

6.3.1.3 CASAC and Public Commenters' Views on the Adequacy of the Current Standard

Beyond the evidence- and risk/exposure-based information discussed above, staff has also taken into account the comments and advice of CASAC (Henderson, 2006a, b, c), based on their review of the CD and earlier drafts of this document and the related technical support documents, as well as comments on earlier drafts of these documents provided by public commenters.[15] The range of views summarized here generally reflects differing judgments as to the relative weight to place on various types of evidence, the exposure- and risk-based information, and the associated uncertainties, as well as differing judgments about the importance of various O_3-related health effects from a public health perspective.

In a letter to the Administrator (Attachment 1), the CASAC O_3 Panel, with full endorsement of the chartered CASAC, unanimously concluded that there is "no scientific justification for retaining" the current primary O_3 standard, and the current standard "needs to be substantially reduced to protect human health, particularly in sensitive subpopulations" (Henderson, 2006c, pp. 1-2).[16] The Panel's rationale for this conclusion is outlined in their letter, beginning with their conclusion that "*new evidence supports and build-upon key, health-related conclusions drawn in the 1997 Ozone NAAQS review.*" (id., p. 3). The Panel points to studies discussed in Chapter 3 and Appendix 3B of this document in noting that several new single-city studies and large multi-city studies have provided more evidence for adverse health effects at concentrations lower than the current standard, and that these epidemiological studies are backed-up by evidence from controlled human exposure studies. The Panel specifically noted evidence from the recent Adams (2006) study that reported statistically significant decrements in the lung function of healthy, moderately exercising adults at a 0.08 ppm exposure level, and importantly, also reported adverse lung function effects in some individuals at 0.06 ppm. In concluding that these results indicate that the current standard "is not sufficiently health-protective with an adequate margin of safety," the Panel noted that that while similar studies in sensitive groups such as asthmatics have yet to be conducted, "people with asthma, and particularly children, have been found to be more sensitive and to experience larger

[15]All written comments submitted to the Agency are available in the docket for this rulemaking, as are transcripts of the public meetings held in conjunction with CASAC's review of earlier drafts of this document and of draft and final versions of the CD on which this document is based.

[16]Comments of individual Panel members are available as Attachment D to the CASAC O_3 Panel letter (Henderson, 2006c); the letter without its attachments is reproduced as Attachment 1 to this document and the attachments to the letter, including individual Panel member comments, can be found in the docket and online at http://www.epa.gov/sab/pdf/casac-07-001.pdf.

decrements in lung function in response to ozone exposures than would healthy volunteers (Mortimer *et al.*, 2002)." (Henderson, 2006c, p. 4).

The CASAC Panel also highlighted a number of adverse health effects, beyond lung function decrements, that are attributable to low-concentration exposure to ambient O_3, below the level of the current standard, based on a broad range of epidemiological and controlled exposure studies (id.). These adverse health effects include increases in school absenteeism, respiratory hospital emergency department visits among asthmatics and patients with other respiratory diseases, hospitalizations for respiratory illnesses, symptoms associated with adverse health effects (including chest tightness and medication usage, and mortality (non-accidental, cardiorespiratory deaths) reported at exposure levels well below the current standard. "*The CASAC considers each of these findings to be an important indicator of adverse health effects*." (id.). The Panel further noted that the risk assessment (discussed above in chapter 5) estimated that beneficial reductions in some adverse health effects would occur upon meeting the lowest standard level (0.064 ppm) considered in the assessment.

With regard to the justification discussed in the second draft of this Staff Paper for consideration of retaining the current standard,[17] the CASAC Panel felt that more emphasis should be placed on numbers of subjects in controlled human exposure studies with FEV_1 decrements greater than 10%, which can be clinically significant, rather than on the relatively small average decrements. The Panel also emphasized significant O_3-related inflammatory responses and markers of injury to the epithelial lining of the lung that are independent of spirometric responses. Further, the Panel expressed the view that the justification for considering retaining the current standard discussed in the earlier draft of this document did not place enough emphasis on serious morbidity (e.g., hospital admissions) and mortality observed in epidemiology studies. On the basis of the large amount of recent data evaluating adverse health effects at levels at and below the current O_3 standard, it was the unanimous opinion of the CASAC that the current primary O_3 standard is not adequate to protect human health, that the relevant scientific data do not support consideration of retaining the current standard, and that the current standard needs to be substantially reduced to be protective of human health, particularly in sensitive subpopulations (id., pp. 4-5).

Further, the CASAC letter noted that "*there is no longer significant scientific uncertainty regarding the CASAC's conclusion that the current 8-hr primary NAAQS must be lowered*." (id., p. 5). The Panel noted that a "large body of data clearly demonstrates adverse human health effects at the current level" of the standard, such that "[R]etaining this standard would continue

[17]See second draft O_3 Staff Paper (U.S. EPA, 2006c, e.g., p. 6-50) for the justification referred to by CASAC. In the second draft Staff Paper staff concluded that "consideration could be given" to retaining the current standard, as discussed below in section 6.3.1.4 which presents staff's final conclusions.

to put large numbers of individuals at risk for respiratory effects and/or significant impact on quality of life including asthma exacerbations, emergency room visits, hospital admissions and mortality" (id.). The Panel also noted that "scientific uncertainty does exist with regard to the lower level of ozone exposure that would be fully protective of human health," concluding that "it is possible that there is no threshold for an ozone-induced impact on human health and that some adverse events may occur at policy-relevant background" (id.).

Consistent with the advice of CASAC, several public commenters supported revising the primary O_3 standard to provide increased public health protection.[18] In considering the available evidence as a basis for their views, these commenters generally noted that the controlled human exposure studies, showing statistically significant declines in lung function, increases in respiratory symptoms, airway inflammation and airway responsiveness at a 0.08 ppm exposure level, were conducted with healthy adults, not members of sensitive groups including people with asthma and active children generally. Further, recognizing the substantial variability in response between subjects, some of these commenters felt that the number of subjects included in these studies was too small to ascertain the full range of responses, especially for sensitive groups. Such considerations in part were the basis for these commenters' view that an O_3 standard set at 0.08 ppm is not protective of public health and has no margin of safety for sensitive groups.

In considering the results of the human exposure and health risk assessment, this group of commenters generally expressed the view that these assessments substantially underestimate the public health impacts of exposure to O_3. For example, several commenters noted that the assessments are done for a limited number of cities, they do not address risks to important sensitive subpopulations (e.g., outdoor workers, active people who spend their summers outdoors, children up to 5 years of age), and they do not include many health effects that are important from a public health perspective (e.g., school absences, restricted activity days). Further, some of these commenters expressed the view that the primary O_3 standard should be set to protect the most exposed and most vulnerable groups, and the fact that some children are frequently indoors, and thus at lower risk, should not weigh against setting a standard to protect those children who are active outdoors. To the extent the exposure and risk estimates are considered, some of these commenters felt that primary consideration should be given to the estimates based on 2002 air quality, for which most areas had relatively higher O_3 levels than in 2004, so as to ensure public health protection even in years with relatively worse O_3 air quality levels. Some commenters also felt that the exposure analysis should focus on exposures of

[18] This group of commenters included a public health advocacy group, a medical association, a State agency, and a regional State organization.

concern down to at least 60 ppb, the lower end of the range of alternative standards advocated by the CASAC Panel during their public meeting in August 2006.

In contrast to the views discussed above, several other public commenters supported retaining the current standards.[19] In considering the available evidence as a basis for their views, these commenters challenged a number of aspects of the interpretation of the evidence presented in the CD. For example, some of these commenters asserted that EPA generally overestimates the magnitude and consistency of the results of short-term exposure epidemiological studies (e.g., for respiratory symptoms, school absences, hospital admissions, mortality), mistakenly links statistical significance and consistency with strength of associations, and underestimates the uncertainties in interpreting the results of such studies. Further, these commenters generally express the view that there is significant uncertainty related to the reliability of estimates from time-series studies, in that ambient monitors do not provide reliable estimates of personal exposures, such that the small reported morbidity and mortality risks are unlikely to be attributable to people's exposures to O_3. Rather, these commenters variously attribute the reported risks to the inability of time series studies to account for key model specification factors such as smoothing for time-varying parameters, meteorological factors, removal of O_3 by building ventilation systems, and confounding by co-pollutants. In particular, these commenters generally asserted that reported associations between short-term O_3 exposure and mortality are not causal, in that the reported relative risks are too small to provide a basis for inferring causality and the associations are most likely due to confounding, inappropriately specified statistical models, or publication bias.

In considering the results of the human exposure and health risk assessment, this group of commenters generally expressed the view that these assessments are based on a number of studies that should not be used in quantitative risk assessment. For example, some commenters variously asserted that the results of time-series studies should not be used at all in quantitative risk assessments, that risk estimates from single city time-series studies should not be used since they are highly heterogeneous and influenced by publication bias, and that risk estimates from multi-city studies should not be used in estimating risk for individual cities. This group of commenters also generally expressed the view that the assessments generally overestimate the public health impacts of exposure to O_3. Noting that the risk assessment used a nonlinear exposure-response function to estimate decreased lung function risks, some commenters expressed the view that a nonlinear approach should also be used to assess other acute morbidity effects and mortality. This view was in part based on judgments that it is not possible to determine if thresholds exist using time-series analyses and that the lack of association of O_3 to

[19]This group of commenters included industry associations and corporations.

mortality in the winter season is highly supportive of the likelihood of the existence of an effect threshold. With regard to the risk assessment based on controlled human exposure studies of lung function decrements, some commenters expressed the view that the assessment should not rely on what they characterized as "outlier" information to define exposure-response relationships, with reference to the data in the Adams (2006) study at the 0.06 and 0.04 ppm exposure levels, but rather should focus on group central tendency response levels. Further, some commenters expressed the view that the air quality rollback algorithm used may result in overestimates in benefits from emission reductions. Some commenters noted that potential beneficial effects of O_3 in shielding from UV-B radiation are not quantified in the assessment, and that the assessment should discuss the evidence for both adverse and beneficial effects with the same objectivity. Finally, some of these commenters asserted that since estimates of exposures of concern (defined as 0.080 ppm) and lung function decrements are substantially below the estimates available when the current O_3 standard was set in 1997, retaining the current standard is the most appropriate policy alternative.

6.3.1.4 Staff Conclusions on the Adequacy of the Current Standard

As discussed above, we have considered new evidence from controlled human exposure, toxicological, and epidemiological studies as well as estimates of O_3-related exposures of concern and risks upon meeting the current O_3 standard in many urban areas across the U.S., together with associated uncertainties. As an initial matter, we note that there is general agreement among staff, CASAC, and all interested parties who commented on earlier drafts of this document that this information supports consideration of a primary O_3 standard that is at least as protective as the current standard, with no one supporting consideration of an O_3 standard that is any less protective.

In considering whether the current standard is adequate or should be revised to provide increased public health protection, we first note that in the second draft of this Staff Paper, retention of the current standard was included among the policy options that we identified for consideration. At that time, while we recognized that there was substantial evidence that could be interpreted as calling into question the adequacy of the current standard, we chose not to exclude the option of retaining the current standard from the provisional conclusions presented in that draft document. We wanted to have the benefit of receiving CASAC views and public comments on the strength of the available evidence, the results of the exposure/risk assessments, and their interpretation of the evidence with regard to judging the adequacy of the current standard before reaching final conclusions. As discussed below, based on the available information and taking into account the views of CASAC and public comments, we now conclude that the overall body of evidence clearly calls into question the adequacy of the current

standard in protecting sensitive groups, notably including asthmatic children and other people with lung disease, as well as all children and older adults, especially those active outdoors, and outdoor workers,[20] against an array of adverse health effects that range from decreased lung function to serious indicators of respiratory morbidity including ED visits and hospital admissions for respiratory causes, and possibly cardiovascular effects and mortality. We believe the available information provides strong support for consideration of an O_3 standard that would provide increased health protection for these sensitive groups.

In discussing information related to the adequacy of the current standard, we have noted that evidence of a range of respiratory-related morbidity effects seen in the last review has been strengthened, both through toxicological and controlled human exposure (see Chapter 3, e.g., Table 3-1 and Appendix 3C) studies as well as through many new panel and epidemiological studies (see Chapter 3, e.g., Figure 3-4 and Appendix 3B). In addition, new evidence identifies people with asthma as an important susceptible population for which estimates of respiratory effects in the general population may underestimate the magnitude or importance of the effect. New evidence about mechanisms of toxicity helps to explain the biological plausibility of O_3-induced respiratory effects and is beginning to suggest mechanisms that may link O_3 exposure to cardiovascular effects. Further, there is now relatively strong evidence for associations between O_3 and total nonaccidental and cardiopulmonary mortality, even after adjustment for the influence of season and PM. Relative to the information that was available to inform the Agency's 1997 decision to set the current standard, the newly available evidence increases our confidence that a broad array of adverse health effects, especially indicators of respiratory morbidity, are causally related to O_3 exposures, and that mortality is likely associated with O_3 exposures during the O_3 season.

In examining the entire body of evidence and considering CASAC's views and advice on interpreting the evidence with regard to the adequacy of the current standard, we conclude that there is important new evidence demonstrating that exposures to O_3 at levels below the level of the current standard cause or are clearly associated with a broad array of adverse health effects in sensitive populations. For example, we note new direct evidence of transient and reversible lung function effects and respiratory symptoms in some healthy individuals at exposure levels below the level of the current standard. In addition, there is now epidemiological evidence of statistically significant O_3-related associations with lung function and respiratory symptom effects, respiratory-related ED visits and hospital admissions, as well as possibly increased mortality, in areas that likely would have met the current standard. There are also many

[20] In defining sensitive groups this way we are including both groups with greater inherent sensitivity and those more likely to be exposed.

epidemiological studies done in areas that likely would not have met the current standard but which nonetheless report statistically significant associations that generally extend down to ambient O_3 concentrations that are well below the level of the current standard. Further, there are a few studies that have examined subsets of data that include only days with ambient O_3 concentrations below the level of the current standard, or below even much lower O_3 concentrations, and continue to report statistically significant associations. Our level of confidence in the findings from these studies is not related to whether they were done in areas that likely would or would not have met the current standard. However, we agree with the views expressed by the CASAC O_3 Panel[21] that uncertainty in epidemiological findings increases at the low end of the ranges of concentrations observed in these studies, which are generally well below the level of the current standard, because of limitations in interpreting the results that relate, for example, to poor correlations between ambient concentrations and personal exposure and to questions of plausibility that are more salient at relatively low concentrations.

Based on the strength of the currently available evidence of adverse health effects, especially indicators of respiratory morbidity, and on the extent to which the evidence indicates that such effects likely result from exposures to ambient O_3 concentrations well below the level of the current standard, we conclude that the available evidence clearly calls into question the adequacy of the current standard and provides strong support for giving consideration to revising the standard to provide increased protection, especially for sensitive groups, against a broad array of adverse health effects. As discussed below, we have also considered the results of the exposure and risk assessments conducted for this review to provide some quantitative perspective on the extent to which sensitive groups are likely to experience exposures of concern and on the risk of experiencing various adverse health effects when air quality is adjusted to simulate meeting the current standard in a number of urban areas in the U.S.

In considering the results of the exposure and risk assessments, we first note that the CASAC Panel has expressed the view that the exposure analysis represents a state of the art modeling approach, that the risk assessment is well-done and balanced, and that the results of both are appropriate input to the decision on the O_3 NAAQS. Moreover, the additional uncertainty and sensitivity analyses conducted after CASAC review of the second draft Staff Paper have increased our overall confidence in the results of these assessments. Accordingly, in considering the adequacy of the current standard, we have placed substantial weight on these results, both as direct measures of a limited set of O_3-related risks to public health and as indicators of the potential for a range of other types of adverse health effects for which currently available information is too limited to allow for direct estimates of risk.

[21]See, for example, the written comments of Dr. Vedal (Henderson, 2006c, Appendix D, p. D-74).

Turning to the results of the exposure assessment, we note that estimating exposures of concern provides an important indication of the potential magnitude of the incidence of health outcomes that we cannot currently evaluate in a quantitative risk assessment, such as, increased airway responsiveness, increased pulmonary inflammation, including increased cellular permeability, and decreased pulmonary defense mechanisms. These physiological effects, which have been demonstrated to occur in healthy people at O_3 exposures as low as 0.080 ppm, are associated with aggravation of asthma, increased medication use, increased school and work absences, increased susceptibility to respiratory infection, increased visits to doctors' offices and EDs, increased admissions to hospitals, and possibly to cardiovascular system effects and chronic effects such as chronic bronchitis or long-term damage to the lungs that can lead to reduced quality of life. In considering whether the current standard provides a margin of safety against such serious respiratory morbidity effects not just in healthy adults but in sensitive groups, such as people with asthma or other lung diseases, children, and older adults, we believe it is appropriate to consider the extent to which the current standard reduces exposures of concern not only at and above the 0.080 ppm benchmark level, but more importantly also at lower benchmark levels. Therefore, we have focused on the extent to which the current standard reduces exposures of concern at the 0.070 and 0.060 ppm benchmark levels, noting that 0.060 ppm is the lowest level at which potentially adverse lung function decrements have been observed in healthy people. While we believe that exposures of concern at these lower benchmark levels are an important indicator of the potential for adverse health effects especially in sensitive groups, as discussed above, we note that due to individual variability in responsiveness only a subset of individuals in these groups with exposures of concern can be expected to experience such adverse health effects.

Based on the aggregate estimates summarized above in Table 6-1a-c for the 12 U.S., urban areas included in the exposure analysis, we first note that there is substantial year-to-year variability across the three years included in this analysis, ranging up to an order of magnitude or more, in estimates of the number of people and the number of occurrences of exposures of concern at each of the benchmark levels. We believe it is appropriate to consider not just the average estimates across all years, but also to consider public health impacts in years with relatively poorer air quality. In so doing, we note that even when considering the benchmark level of $\geq$ 0.080 ppm, an exposure level at which adverse respiratory effects have been demonstrated in healthy adults, approximately 100,000 asthmatic children (and over 700,000 total children) in these 12 cities alone are estimated to experience such exposure levels in the worst of the three years when the current standard is met. In looking at the lower benchmark levels that are more relevant to providing a margin of safety for sensitive groups, over 500,000 (~20%) asthmatic children (and over 3 million total children) in these 12 cities are estimated to

experience exposures ≥ 0.070 ppm in the worst of the three years; even in the mid-year, approximately 150,000 (~6%) asthmatic children (and approximately one million total children) are estimated to experience such exposures. These estimates are roughly 2 to 4 times higher when considering exposures at the benchmark level of ≥ 0.060 ppm.

We also note that there is substantial city-to-city variability in these estimates, as summarized in Table 6-7, and we believe it is appropriate to consider not just aggregate estimates across all cities, but also to consider public health impacts in cities that receive relatively less protection from the current standard. For example, in considering the benchmark level of ≥ 0.070 ppm, while the aggregate percentage of asthmatic children estimated to experience such exposures of concern across all 12 cities is ~ 6% in the mid-year and ~20% in the worst year when the current standard is met, these estimates range up to 12% and 41%, respectively, in the city with the least degree of protection from the current standard. As seen in Table 6-7, such percentages are substantially higher when considering exposures at the benchmark level of ≥ 0.060 ppm, ranging up to 38 to 69% in the mid to worst years. Estimates of the percent of all children exposed are generally similar or slightly lower than those for asthmatic children.

Table 6-7. Estimates of Percent of Children Exposed While at Moderate Exertion to 8-Hour Daily Maximum Ozone Concentrations ≥ 0.070 ppm and ≥ 0.060 ppm Combined for 12 Urban Areas in the U.S., and the Range of Estimates for Each of the 12 Cities – Just Meeting Current Standard

Exposure of Concern Benchmark Level	Percent of All Children (5-18 yrs old) (18.3 million children) -- Aggregated across 12 cities (Range for each of 12 cities)			Percent of Asthmatic Children (5-18 yrs old) (2.6 million children) – Aggregated across 12 cities (Range for each of 12 cities)		
	2002	2003	2004	2002	2003	2004
≥ 0.070 ppm	18 (1 – 38)	5 (1 – 11)	1 (0 – 7)	20 (1 – 41)	6 (1 – 12)	1 (0 – 7)
≥ 0.060 ppm	44 (7 – 66)	22 (8 – 39)	10 (1 – 26)	47 (7 – 69)	24 (9 – 38)	11 (1 – 28)

With regard to estimates of risks of health effects in sensitive populations likely to remain upon meeting the current standard, we note that some such estimates related to relatively less serious lung function effects are now appreciably lower than in the last review, whereas risk

estimates related to more serious effects, such as hospital admissions, are as high or higher than previously estimated. In addition, unlike in the last review, there is now evidence that supports estimating risks for respiratory symptoms in asthmatic children and O_3-related mortality.

Based on Tables 6-2 and 6-3, we note that meeting the current O_3 standard substantially reduces the estimated risk of moderate lung function decrements (i.e., ≥ 15% FEV_1 decrement) in all school age children across 12 urban areas. In asthmatic children the reduction in the estimated risk of moderate lung function decrements (i.e., ≥ 10% FEV_1 decrement) is not as large, with about 4 to 8% of asthmatic school age children estimated to experience one or more occurrences of moderate lung function decrements even when the current standard is met, resulting in over 1 million occurrences just in 5 urban areas in a year with relatively poorer air quality (2002). Moreover, the estimated number of occurrences of moderate or greater lung function decrements per child is on average approximately 5 to 7 in all children and 8 to 10 in asthmatic children in an O_3 season, even when the current standard is met. In the 1997 review of the O_3 standard a general consensus view of the adversity of such moderate responses emerged as the frequency of occurrences increases, with the judgment that repeated occurrences of moderate responses, even in otherwise healthy individuals, may be considered adverse since they may well set the stage for more serious illness.

Large lung function decrements (i.e., ≥ 20% FEV_1 decrement) would likely interfere with normal activities in many healthy individuals, therefore single occurrences would be considered to be adverse. In people with asthma, large lung function responses (i.e., ≥ 20% FEV_1 decrement), would likely interfere with normal activities for most individuals and would also increase the likelihood that these individuals would use additional medication or seek medical treatment. Not only would single occurrences be considered to be adverse to asthmatic individuals under the ATS definition, but they also would be cause for medical concern. While the current standard reduces the occurrences of large lung function decrements in all children and asthmatic children overall from about 60 to 80%, in a year with relatively poorer air quality (2002) there are estimated to be about 500,000 occurrences in all school children across 12 urban areas, and about 40,000 occurrences in asthmatic children across just 5 urban areas. As noted above, it is clear that even when the current standard is met over a three-year period, air quality in each year can vary considerably, as evidenced by relatively large differences between risk estimates based on 2002 to 2004 air quality. We believe it is appropriate to consider this yearly variation in air quality allowed by the current standard in judging the extent to which impacts on members of sensitive groups in a year with relatively poorer air quality remains of concern from a public health perspective.

As seen in Tables 6-4 through 6-6, risks of respiratory symptom days in moderate to severe asthmatic children, respiratory-related hospital admissions, and non-accidental and

cardiorespiratory mortality, respectively, are not reduced to as great an extent by meeting the current standard as are lung function decrements. For example, just meeting the current standard reduces the estimated average incidence of chest tightness in moderate to severe asthmatic children living in the Boston urban area by 11 to 15%, based on adjusting 2002 and 2004 air quality, respectively, resulting in an incidence per 100,000 relevant population of approximately 23,000 to 31,000 children, attributable to O_3 exposure (Table 6-4). The current standard reduces the estimated incidence of respiratory-related hospital admissions in the New York City urban area by 16 to 18%, based on adjusting 2002 and 2004 air quality, respectively, resulting in an incidence per 100,000 population of approximately 4.6 to 6.4, respectively (Table 6-5). Across the 12 urban areas considered in this assessment, the estimates of non-accidental mortality incidence per 100,000 relevant population range from 0.4 to 2.6 (for 2002) and 0.5 to 1.5 (for 2004) (Table 6-6). Meeting the current standard results in a reduction of the estimated incidence per 100,000 population to a range of 0.3 to 2.4 based on adjusting 2002 air quality and a range of 0.3 to 1.2 based on adjusting 2004 air quality. Estimates for cardiorespiratory mortality show similar patterns.

Staff notes that in considering the estimates of the proportion of population affected and the number of occurrences of the health effects that are included in the risk assessment, these limited estimates are indicative of a much broader array of O_3-related health endpoints that are part of a "pyramid of effects" that include various indicators of morbidity that could not be included in the risk assessment (e.g., school absences, increased medication use, ED visits) and which primarily affect members of sensitive groups. While we had sufficient information to estimate and consider the number of symptom days in children with moderate to severe asthma, we recognize that there are many other effects that may be associated with symptom days, such as increased medication use, school and work absences, or visits to doctors' offices, that we did not have sufficient information to estimate but are important to consider in assessing the adequacy of the current standard. The same is true for more serious, but less frequent effects. We estimated hospital admissions, but we did not have sufficient information to estimate ED visits in a quantitative risk assessment. Consideration of such unquantified risks for this array of health effects, in conjunction with risk estimates for health effects that we did quantify, leads us to conclude that they are indicative of risks to sensitive groups that can reasonably be judged to be important from a public health perspective. These risk-based considerations reinforce our conclusion that consideration should be given to revising the standard so as to provide increased public health protection, especially for sensitive groups such as people with asthma or other lung diseases, as well as children and older adults, particularly those active outdoors, and outdoor workers.

Taking into account the above evidence- and exposure/risk-based considerations, staff concludes that the body of information that is now available supports consideration of revising the current primary O_3 standard so as to afford greater public health protection, especially to sensitive groups, and that it does not support retention of the current standard. The following sections on indicator, averaging time, level, and form are intended to help inform consideration of an appropriate range of alternative standards.

6.3.2 Indicator

In the last review EPA focused on a standard for O_3 as the most appropriate surrogate for ambient photochemical oxidants. In this review, while the complex atmospheric chemistry in which O_3 plays a key role has been highlighted, no alternatives to O_3 have been advanced as being a more appropriate surrogate for ambient photochemical oxidants.

It is generally recognized that control of ambient O_3 levels provides the best means of controlling photochemical oxidants of potential health concern. Further, among the photochemical oxidants, the acute exposure chamber, panel and field epidemiological human health database provides evidence only for O_3 at levels of photochemical oxidants commonly reported in the ambient air, in part because few other photochemical oxidants are routinely measured. However, recent investigations on copollutant interactions have used simulated urban photochemical oxidant mixes. These investigations suggest the need for similar studies to help in understanding the biological basis for effects observed in epidemiological studies that are associated with air pollutant mixtures, where O_3 is used as the surrogate for the mix of photochemical oxidants. Meeting the O_3 standard can be expected to provide some degree of protection against potential health effects that may be independently associated with other photochemical oxidants but which are not discernable from currently available studies indexed by O_3 alone. Since the precursor emissions that lead to the formation of O_3 generally also lead to the formation of other photochemical oxidants, measures leading to reductions in population exposures to O_3 can generally be expected to lead to reductions in population exposures to other photochemical oxidants.

6.3.3 Averaging Time

6.3.3.1 Short-Term and Prolonged (1 to 8 Hours)

The current 8-hr averaging time for the primary O_3 NAAQS was set in 1997. The decision to revise the averaging time of the primary standard from 1 to 8 hr was supported by the following key observations and conclusions (62 FR 38861):

(1) The 1-hr averaging time of the previous NAAQS was originally selected on the basis of health effects associated with short-term (i.e., 1- to 3-hr) exposures.

(2) Substantial health effects information was available for the 1997 review that demonstrated associations between a wide range of health effects (e.g., moderate to large lung function decrements, moderate to severe symptoms and pulmonary inflammation) and prolonged (i.e., 6- to 8-hr) exposures below the level of the NAAQS.

(3) Results of the quantitative risk analyses showed that reductions in risks from both short-term and prolonged exposures could be achieved through a primary standard with an averaging period of either 1 or 8 hr.

(4) The 8-hr averaging time is more directly associated with health effects of concern at lower O_3 concentrations than the 1-hr averaging time. It was thus the consensus of CASAC "that an 8-hour standard was more appropriate for a human health-based standard than a 1-hour standard." (Wolff, 1995)

In looking at the new information that is discussed in section 7.6.2 of the CD, epidemiological studies have used various averaging periods for O_3 concentrations, most commonly 1-hr, 8-hr and 24-hr averages. As described more specifically below, in general the results presented from U.S. and Canadian studies (Appendix 3B) show no consistent difference for various averaging times in different studies.

Only a few studies presented results for different O_3 averaging periods using the same data set. Two of the recent multi-city mortality studies reported associations for multiple averaging times (Bell et al., 2004; Gryparis et al., 2004). Both reported that the effect estimates for different averaging times were not statistically different, though the effect estimates for associations with 1-hr daily maximum O_3 concentrations were somewhat larger than those for longer averaging times, especially 24-hr average O_3. In addition, Gent et al., (2003) reported that associations for 1-hr and 8-hr average O_3 with respiratory symptoms were not significantly different.

Among the single-city epidemiological studies, Peters et al. (2001) reported positive, but not statistically significant associations between O_3 and the incidence of myocardial infarction (CD, p. 7-55); this study differs from most since the short-term O_3 concentration used was the time period preceding the health event, not the highest daily short-term average concentration. The effect estimate for the association with O_3 averaged over a 2-hr period prior to the myocardial infarction was substantially larger than that reported for an association with 24-hr average O_3 (Peters et al., 2001). The CD reports results for a number of single-city results that generally reported effect estimate sizes that were larger when comparing 1-hr or 8-hr daily maximum O_3 concentrations with the 24-hr concentration, but the results did not differ statistically (CD, p. 7-120). The CD observes that the various O_3 average concentrations were generally very highly correlated with one another, so it is not surprising that effect estimates

would be similar. The CD concludes that the epidemiological study results were generally comparable for the three O_3 averaging times (CD, p. 7-120).

Because the 8-hr averaging time continues to be more directly associated with health effects of concern from controlled human exposure studies at lower concentrations than do shorter averaging periods, we have not focused on alternative averaging times in this review and have not conducted exposure or risk assessments for standards with averaging times other than 8 hours. In its letter to the Administrator, the CASAC O_3 Panel supported the continued use of an 8-hr averaging time for the primary O_3 standard (Henderson, 2006c, p. 2), as did many commenters.

Some other commenters expressed the view that consideration should be given to setting or reinstating a 1-hr standard, in addition to maintaining the use of an 8-hr averaging time, to protect people in those parts of the country with relatively more "peaky" exposure profiles. These commenters point out that when controlled exposure studies using triangular exposure patterns (with relatively higher 1-hr peaks) have been compared to constant exposure patterns with the same aggregate O_3 dose (in terms of concentration x time), "peaky" exposure patterns are seen to lead to higher risks. The California Air Resources Board made particular note of this point, expressing the view that a 1-hr standard would more closely represent actual exposures, in that many people spend only 1 to 2 hours a day outdoors, and that it would be better matched to O_3 concentration profiles along the coasts where O_3 levels are typically high for shorter averaging periods than 8 hours.

In considering the information discussed above, CASAC views and public comments on the earlier draft of this Staff Paper, staff concludes that the 8-hr averaging time remains the most appropriate averaging time for a human health-based standard. This conclusion is based on the observations summarized above, particularly: (1) the fact that the 8-hr averaging time is more directly associated with health effects of concern at lower O_3 concentrations than are averaging times of shorter duration and (2) results from quantitative risk analyses showing that attaining an 8-hr standard reduces the risk of experiencing health effects associated with both 8-hr and shorter duration exposures. Furthermore, the CASAC O_3 Panel unanimously agreed that the health-based standard should be an 8-hr average in 1995 (Wolff, 1995) and made no comment in 2006 (Henderson, 2006c) to suggest that any averaging time other than 8-hr was appropriate for the health-based standard.

In addition to quantitative risk analyses, we conducted an analysis of a recent three-year period of air quality data (2002 to 2004) to determine whether the comparative 1- and 8-hr air quality patterns that were observed in the last review continue to be observed based on more recent air quality data. This updated air quality analysis (McCluney, 2007) is very consistent with the analysis done in the last review in that it indicates that only two urban areas of the U.S.

have such "peaky" air quality patterns that the ratio of 1-hr to 8-hr design values is greater than 1.5. This suggests that based on recent air quality data, it is reasonable to again conclude that an 8-hr average standard at or below the current level would generally be expected to provide protection equal to or greater than the previous 1-hr standard of 0.12 ppm in almost all urban areas. Thus, staff concludes that setting a standard with an 8-hr averaging time can effectively limit both 1- and 8-hr exposures of concern and is appropriate to provide adequate and more uniform protection of public health from both short-term and prolonged exposures to O_3 in the ambient air. Therefore, we recommend that the 8-hr averaging time be retained and do not recommend consideration of a separate 1-hr standard at this time.

6.3.3.2 Long-Term

During the last review, there was a large animal toxicological database for consideration that provided clear evidence of associations between long-term (e.g., from several months to years) exposures and lung tissue damage, with additional evidence of reduced lung elasticity and accelerated loss of lung function. However, there was no corresponding evidence for humans, and the state of the science had not progressed sufficiently to allow quantitative extrapolation of the animal study findings to humans. For these reasons, consideration of a separate long-term primary O_3 standard was not judged to be appropriate at that time, recognizing that the 8-hr standard would act to limit long-term exposures as well as short-term and prolonged exposures.

In the current review, long-term animal toxicological studies continue to support the relationship between O_3 exposure and structural alterations in several regions of the respiratory tract and identify the CAR as the most affected region. In addition, animal toxicological studies that utilized exposure regimens to simulate seasonal exposure patterns also report increased lung injury compared to conventional long-term, stable exposures. (CD, p. 8-85) Collectively, the evidence from animal studies strongly suggest that O_3 is capable of damaging the distal airways and proximal alveoli, resulting in lung tissue remodeling leading to apparently irreversible changes. Compromised pulmonary function and structural changes due to persistent inflammation may exacerbate the progression and development of chronic lung disease (CD, p. 8-70). Recent epidemiological studies observed that reduced lung function growth in children was associated with seasonal exposure to O_3; however, cohort studies investigating the effect of annual or multiyear O_3 exposure observed little clear evidence for impacts of longer-term, relatively low-level O_3 exposure on lung function development in children.

Collectively, the epidemiological studies are inconclusive, but suggestive of respiratory health effects from long-term O_3 exposure. While there continues to be evidence of structural changes in the respiratory tract in animal studies, with some very weak support from epidemiological studies in children, it is highly uncertain as to what long-term patterns of

exposure or O_3 concentrations in humans may be required to produce the morphological changes found in the animal studies and it is not currently possible to characterize the possible magnitude or severity of any such effects occurring in humans in response to ambient O_3 exposures at levels observed in the U.S.. Further, to the extent that meeting an 8-hr O_3 standard in some cases is expected to result in lower long-term average concentrations, the 8-hr standard would provide some protection against effects that may be associated with long-term O_3 exposures.

In its letter to the Administrator, the CASAC O_3 Panel offered no views on the long-term exposure evidence, nor did it suggest that consideration of a primary O_3 standard with a long-term averaging time was appropriate (Henderson, 2006c, p. 2). Similarly, no commenters expressed support for considering such a standard.

Staff concludes that a health-based standard with a longer-term averaging time than 8 hours is not warranted at this time. While potentially more serious health effects have been identified as being associated with longer-term exposure studies of laboratory animals and in epidemiology studies, there remains substantial uncertainty regarding how these data could be used quantitatively to develop a basis for setting a long-term health standard. Because long-term air quality patterns would be improved in areas coming into attainment with an 8-hr standard, the potential risk of health effects associated with long-term exposures would be reduced in any area meeting an 8-hr standard. Furthermore, the CASAC O_3 Panel offered no advice either in 1995 (Wolff, 1995) or in 2006 (Henderson, 2006c) that a long-term health-based standard should be considered. Thus, staff does not recommend consideration of a long-term, health-based standard at this time.

6.3.4 Level

In considering alternative O_3 standard levels that would provide greater protection against the array of O_3-related adverse health effects than that afforded by the current standard, staff has taken into account both evidence- and exposure/risk-based considerations, as well as the comments and advice of CASAC and public commenters' views. The discussion of alternative levels in this section builds upon the information presented above in the discussion of the adequacy of the current standard (section 6.3.1) to help inform staff's evaluation of the range of levels that would be appropriate for consideration.

As an initial matter, we have considered whether it is appropriate to continue to specify the level of the O_3 standard to the nearest hundredth (two decimal places) ppm, or whether the precision with which ambient O_3 concentrations are measured supports specifying the standard level to the nearest thousandth ppm (i.e., to the nearest part per billion (ppb)). As discussed above in Chapter 2 (section 2.4.2), staff conducted an analysis to determine the impact of ambient O_3 measurement error on calculated 8-hr average O_3 design value concentrations, which

are compared to the level of the standard to determine whether the standard is attained (Cox and Camalier, 2006). The results of this analysis suggest that instrument measurement error, or possible instrument bias, contribute very little to the uncertainty in design values. More specifically, measurement imprecision was determined to contribute less than 1 ppb to design value uncertainty, and a simulation study indicated that randomly occurring instrument bias could contribute approximately 1 ppb. Staff has interpreted this analysis as being supportive of specifying the level of the standard to the nearest thousandth ppm. This information was provided to the CASAC O_3 Panel and made available to the public at the August 24-25, 2006 public meeting. The Panel concluded that current monitoring technology "allows accurate measurement of O_3 concentrations with a precision of parts per *billion*" and recommended that the specification of the level of the O_3 standard should reflect this degree of precision (Henderson, 2006c).[22] Based on these considerations, staff recommends that consideration be given to specifying the level of an alternative 8-hr O_3 standard to the nearest thousandth ppm. If the current standard were to be specified to this degree of precision, the current standard would effectively be at a level of 0.084 ppm, reflecting the data rounding conventions that are part of the definition of the current 0.08 ppm 8-hr standard.

6.3.4.1 Evidence-based Considerations

In taking into account evidence-based considerations, staff has evaluated available evidence from controlled human exposure studies and epidemiological studies, as well as the uncertainties and limitations in that evidence. In so doing, we focused primarily but not exclusively on U.S. and Canadian studies. In particular, we have considered the extent to which controlled human exposure studies provide evidence of lowest-observed-effects levels and the extent to which epidemiological studies provide evidence of associations that extend down to the lower levels of O_3 concentrations observed in the studies or some indication of potential effect thresholds in terms of 8-hr average O_3 concentrations.

In considering the available controlled human exposure studies, as discussed above in Chapter 3 (section 3.3.1.1), we note that FEV_1 decrements and various measures of respiratory symptoms were observed in some healthy adults following a 6.6-hr exposure level of 0.06 ppm (reflecting exposures of 0.060 ± 0.003 ppm) during moderate exertion. More specifically, Adams (2002) reports that in an earlier study (Adams, 1998) 20% of subjects (6 of 30 subjects) had notable responses (FEV_1 decrements > 10%) at the 0.06 ppm exposure level, and data underlying the Adams (2006) study show that 7% of healthy adult subjects had $\geq$ 10% FEV_1

[22] We also note that the 8-hr O_3 standard adopted by the state of California in 2006 is specified to the nearest thousandth part per million (at a level of 0.070 ppm) (http://www.arb.ca.gov/research/aaqs/ozone-rs/ozone-rs.htm).

decrements at the 0.06 ppm exposure level in that study. Notably, in Adams (2006), total respiratory symptoms (which includes pain on deep inspiration, shortness of breath, and cough) following 5.6 and 6.6-hr exposures at 0.06 ppm (during a triangular exposure pattern, that is more representative of those encountered in summer air pollution episodes than a square-wave exposure pattern) reached statistical significance.

In considering the controlled human exposure study results discussed above in the context of the broader body of controlled human exposure studies, we conclude that these studies provide evidence of a lowest-observed-effects level of 0.060 ppm for potentially adverse lung function decrements and respiratory symptoms in some healthy adults while at prolonged moderate exertion. We further conclude that since people with asthma, particularly children, have been found to be more sensitive and to experience larger decrements in lung function in response to O_3 exposures than would healthy adults, the 0.060 ppm exposure level also can be interpreted as representing a level likely to cause adverse lung function decrements and respiratory symptoms in children with asthma and more generally in people with respiratory disease.

In considering controlled human exposure studies of pulmonary inflammation, airway responsiveness, and impaired host defense capabilities, we note that these studies provide evidence of a lowest-observed-effects level for such effects in healthy adults at prolonged moderate exertion of 0.08 ppm (generally reflecting exposures of 0.080 ppm $\pm$ 0.004 ppm). As discussed above, these physiological effects have been linked to aggravation of asthma and increased susceptibility to respiratory infection, potentially leading to increased medication use, increased school and work absences, increased visits to doctors' offices and EDs, and increased hospital admissions. Further, pulmonary inflammation is related to increased cellular permeability in the lung, which may be a mechanism by which O_3 exposure can lead to cardiovascular system effects, and to potential chronic effects such as chronic bronchitis or long-term damage to the lungs that can lead to reduced quality of life. These are all indicators of adverse O_3-related morbidity effects, which are consistent with and lend plausibility to the adverse morbidity effects and mortality effects observed in epidemiological studies.

In considering epidemiological studies, we first recognize that the available evidence neither supports nor refutes the existence of effect thresholds at the population level for morbidity and mortality effects. As discussed above, based on a consideration of studies that have explored the question of potential thresholds and of seasonal studies that show no consistent O_3-related effects during the cold season when O_3 concentrations are generally low, we conclude that if a population threshold level does exist, it would likely be well below the level of the current O_3 standard and possibly within the range of background levels. More specifically, as discussed above in Chapter 3 (section 3.4.5) and more fully in the CD (Chapter 7, section 7.6.5),

a number of studies reported some suggestive evidence of possible thresholds for morbidity and mortality outcomes in terms of 24-hr, 8-hr, and 1-hr averaging times. These results, taken together, provide some indication of possible 8-hr average threshold levels from below about 25 to 35 ppb up to approximately 50 ppb. Other studies, however, observe linear concentration-response functions suggesting no effect threshold. In considering this information, we conclude that the statistically significant associations between ambient O_3 concentrations and lung function decrements, respiratory symptoms, indicators of respiratory morbidity including increased ED visits and hospitals admissions, and possibly mortality reported in a large number of studies likely extend down to ambient O_3 concentrations that are well below the level of the current standard. Toward the lower end of the range of O_3 concentrations observed in such studies, ranging down to background levels, however, we conclude that there is increasing uncertainty as to whether the observed associations remain plausibly related to exposures to ambient O_3, rather than to the broader mix of air pollutants present in the ambient atmosphere.

We have also considered studies that did subset analyses that include only days with ambient O_3 concentrations below the level of the current standard, or below even lower O_3 concentrations, and continue to report statistically significant associations. Notably, as discussed above, Bell et al. (2006) conducted a subset analysis that continued to show statistically significant associations even when only days with a maximum 8-hr average O_3 concentration below a value of approximately 61 ppb were included.[23] Also of note is the large multi-city NCICAS (Mortimer et al., 2002) that reported statistically significant associations between ambient O_3 concentrations and lung function decrements even when days with 8-hr average O_3 levels greater than 80 ppb were excluded (which consisted of less than 5% of the days in the eight urban areas in the study).

Being mindful of the uncertainties and limitations inherent in interpreting the available evidence, staff believes that the range of alternative O_3 standards appropriate for consideration in this review should take into account information on lowest-observed-effects levels in controlled human exposure studies as well as indications of possible effects thresholds reported in some epidemiological studies and questions of biological plausibility in attributing associations observed down to background levels to O_3 exposures alone. Based on the evidence and these considerations, we conclude that the upper end of the range of consideration should be somewhat below 0.080 ppm, the lowest-observed-effects level for effects such as pulmonary inflammation, increased airway responsiveness and impaired host-defense capabilities in healthy adults while at prolonged moderate exertion. As discussed above, these physiological effects have been linked

[23] Bell et al. (2006) referred to this level as being approximately equivalent to 120 $\mu g/m^3$, daily 8-hr maximum, the World Health Organization guideline and European Commission target value for O_3.

to aggravation of asthma and increased susceptibility to respiratory infection, potentially leading to increased medication use, increased school and work absences, increased visits to doctors' offices and EDs, and increased hospital admissions. Further, pulmonary inflammation is related to increased cellular permeability in the lung, which may be a mechanism by which O_3 exposure can lead to cardiovascular system effects, and to potential chronic effects such as chronic bronchitis or long-term damage to the lungs that can lead to reduced quality of life. These are all indicators of adverse O_3-related morbidity effects, which are consistent with and lend plausibility to the adverse morbidity effects and mortality effects observed in epidemiological studies reporting statistically significant associations with ambient O_3 concentrations that range down to levels well below 0.080 ppm. Based on the evidence, we also conclude that the lower end to the range of alternative O_3 standards appropriate for consideration should be at least as low as the lowest-observed-effects level for potentially adverse lung function decrements and respiratory symptoms in healthy adults, 0.060 ppm, which is also a level likely to cause adverse effects in sensitive groups, and above the level where there is some indication of possible effects thresholds in epidemiological studies. In considering a lower end of the range for consideration, we also recognize that control strategies designed to attain an O_3 standard set at a particular level within an urban area, as measured at the monitor reporting the highest O_3 design value, would cause the entire distribution of O_3 concentrations across the area to be reduced, thus lowering not only concentrations above the level of the standard but also those below that level as well. Thus, we believe that it is appropriate to also consider the results of the exposure and risk assessments that are based on modeling changes in the entire distribution of ambient O_3 concentrations to simulate just meeting alternative standards, discussed below, in reaching conclusions about an appropriate lower end of the range for consideration.

6.3.4.2 Exposure/Risk-based Considerations

In addition to the evidence-based considerations, staff has also considered quantitative exposures and health risks estimated to occur upon meeting the current and alternative standards to help inform judgments about a range of standard levels for consideration that could provide an appropriate degree of public health protection. In so doing, we are mindful of the important uncertainties and limitations that are associated with the exposure and risk assessments, as discussed above in section 6.3.1.2 and more fully in Chapters 4 and 5. For example, important uncertainties affecting the exposure estimates are related to modeling human activity patterns over an O_3 season (especially repetitive exposures), modeling microscale variations in ambient concentrations, and modeling building air exchange rates. With regard to the risk assessment, important uncertainties include, for example, those related to exposure estimates (for children engaged in moderate or greater exertion), as well as those related to estimation of concentration-

response functions, specification of concentration-response models, the possible role of copollutants in interpreting reported associations with O_3, and inferences of a likely causal relationship between O_3 exposure and non-accidental mortality (for risk estimates based on epidemiological studies). As noted above, after considering the key uncertainties, the CASAC Panel expressed the view that the exposure analysis represents a state of the art modeling approach, that the risk assessment is well-done and balanced, and that the results of both are appropriate input to the decision on the O_3 NAAQS.

Beyond these uncertainties, we and CASAC also recognize important limitations to the exposure and risk analyses. For example, we did not have sufficient information to evaluate all relevant sensitive groups (e.g., outdoor workers) or all O_3-related health outcomes (e.g., increased medication use, school absences, ED visits), and the scope of our analyses was generally limited to estimating exposures and risks in 12 urban areas across the U.S., and to only five or just one area for some risk analyses. Thus, it is clear that national-scale public health impacts of ambient O_3 exposures are much larger than the quantitative estimates of O_3-related incidences of adverse health effects and the numbers of children likely to experience exposures of concern associated with meeting the current or alternative standards. Taking these limitations into account, the CASAC advised us not to rely solely on the results of the exposure and risk assessments in considering alternative standards, but also to place significant weight on the body of evidence of O_3-related health effects in drawing conclusions about an appropriate range of levels for consideration. We concur with this important caveat.

Turning to the results of the exposure assessment, we examine the extent to which alternative standard levels below the current standard are estimated to reduce exposures of concern at the 0.070 and 0.060 ppm benchmark levels, for all and asthmatic school age children in the 12 urban areas included in the assessment. The alternative standard levels evaluated include standards set at: 0.080 ppm, 4th daily maximum (i.e., the 80/4 scenario); 0.074 ppm, 5th daily maximum (i.e., the 74/5 scenario); 0.074 ppm, 4th daily maximum (i.e., the 74/4 scenario); 0.074 ppm, 3rd daily maximum (i.e., the 74/3 scenario); 0.070 ppm, 4th daily maximum (i.e., the 70/4 scenario); and, 0.064 ppm, 4th daily maximum (i.e., the 64/4 scenario).[24] Exposure estimates for 14 scenarios are examined (i.e., meeting 6 alternative standard level/form combinations, based on adjusting 2002 and 2004 air quality for all 6 alternative standards, and

[24] The abbreviated notation used to identify the current and alternative standards in the figures showing reductions in risk estimates in this chapter is in terms of ppm and the nth highest daily maximum. For example, the current standard is identified as "0.084/4." This notation is equivalent to the abbreviated labeling used in Chapters 4 and in the text and tables in this chapter which is in terms of ppb and the nth highest daily maximum (e.g., the current standard is labeled "84/4").

adjusting 2003 air quality for the 74/4 and 64/4 alternative standards[25]) at the 0.060 and 0.070 ppm O_3 benchmark levels, for all children and asthmatic children. Individual city estimates of the percent of all children likely to experience exposures of concern are given in the exhibits in Chapter 4. Estimates of the percent of asthmatic children, the number of all children and asthmatic children, and the number of occurrences of exposures of concern, both the aggregate estimates across the 12 urban areas and the individual city estimates, are shown in Appendix 4A. The estimates are for the number and percent of all children and asthmatic children exposed, and the number of person-days (occurrences) of exposures, with daily 8-hr maximum exposures at or above the 0.060 ppm and the 0.070 ppm benchmark levels while at intermittent moderate or greater exertion. For the purpose of this discussion, recommending an appropriate range of levels for consideration, we will focus on scenarios with the same form as the current O_3 standard (i.e. the 80/4, 74/4, 70/4 and 64/4 scenarios) and will address consideration of alternative forms in the next section.

As shown in the exhibits in Chapter 4 and Appendix 4A, the percent of population exposed at any given level is very similar for all and asthmatic school age children. Substantial year-to-year variability in exposure estimates is observed, ranging to over an order of magnitude at the higher alternative standard levels, in estimates of the number of children and the number of occurrences of exposures of concern at both of the benchmark levels. For example, for the 80/4 scenario, approximately 2.2 million children (~12%) (and 330,000 asthmatic children) based on 2002 air quality, to approximately 100,000 children (~0.5%) (and about 10,000 asthmatic children) based on 2004 air quality, are estimated to experience one or more exposures of concern at the benchmark level of $\geq$ 0.070 ppm O_3. For the 74/4 and 64/4 scenarios, we estimated exposures of concern at the two benchmark levels, based on air quality in 2003, which was intermediate between 2002 and 2004. Estimates of exposures of concern for this year are between the estimates for 2002 and 2004. Across the alternative standard levels, in the year with poorer air quality (2002) estimates of the number of all children exposed one or more times ranges from 2.2 million (80/4 scenario) to 30,000 (64/4 scenario); for asthmatic children the range is from 330,000 (80/4 scenario) to about 5,000 (64/4 scenario), at the benchmark level of $\geq$ 0.070 ppm O_3. These results suggest reductions of approximately 35% (80/4 scenario) to nearly 100% (64/4 scenario) across the range of alternative standards in the number of all children and asthmatic children at this exposure of concern level.

The estimates of exposures of concern are considerably larger at the benchmark level of $\geq$ 0.060 ppm O_3, and the pattern of year-to-year variability remains. For example, for the 80/4

[25] Estimates for exposures of concern for the year 2003 were developed since the second draft of this Staff Paper for only 2 alternative standard levels (i.e., 74/4 and 64/4) due to time constraints.

scenario, approximately 6.7 million children (~37%) (> 1,000,000 asthmatic children) based on 2002 air quality, to more than 1,000,000 children (~6%) (and > 100,000 asthmatic children) based on 2004 air quality, are estimated to experience one or more exposures of concern at the benchmark level of ≥ 0.060 ppm O_3. In the year with poorer air quality (2002), across the alternative standard levels, estimates of the number of all children exposed one or more times ranges from 6.7 million (80/4 scenario) to more than 900,000 children (64/4 scenario); for asthmatic children the range is more than 1,000,000 (80/4 scenario) to more than 100,000 (64/4 scenario) at the benchmark level of ≥ 0.060 ppm O_3. At this benchmark level, in the year with poorer air quality, these results suggest reductions of about 15% (80/4 scenario) to about 90% (64/4 scenario) in the percent of children estimated to be exposed across the range of alternative standards.

We also note that there is substantial city-to-city variability in these estimates, as summarized in Table 6-8, and we believe it is appropriate to consider not just the aggregate estimates across all cities, but also to consider the public health impacts in cities that receive relatively less protection from the alternative standards. For example, in considering the benchmark level of ≥ 0.070 ppm, for the 74/4 scenario, while the aggregate percentage of all or asthmatic children estimated to experience such exposures of concern across all 12 cities is about 4% in the worst year, it ranges up to 14% in the city with the least degree of protection from that alternative standard. This pattern of city-to-city variability also occurs at the benchmark level of ≥ 0.060 ppm O_3. While the aggregate percentage of all and asthmatic children estimated to experience one or more exposures of concern across all 12 cities, for the 74/4 scenario, is about 25% in the worst year, it ranges up to approximately 50% in the city with the least degree of protection from that alternative standard.

Turning to the estimates from the risk assessment, Figures 6-1 through 6-6 show the percent reduction in risk estimates from just meeting the current standard (the 84/4 scenario) to just meeting the same alternative standards discussed above, based on adjusting 2002 and 2004 air quality data. These figures also provide perspective on the extent to which the risks in these recent years (i.e., 2002 and 2004) are greater than those estimated to occur upon meeting the current standard (in terms of a negative percent reduction relative to the 84/4 scenario). Figures 6-1 and 6-2 show the percent reduction in the numbers of school age children estimated to experience at least one decrement in FEV_1 ≥ 15% in each of the 12 urban areas for 2002 and 2004, respectively, and Figures 6-3 and 6-4 show the percent reduction in the number of

Table 6-8. Daily Maximum Ozone Concentrations ≥ 0.07 ppm and ≥ 0.06 ppm Combined for 12 Urban Areas in the U.S., and the Range of Estimates for Each of the 12 Cities – Just Meeting Alternative Standards

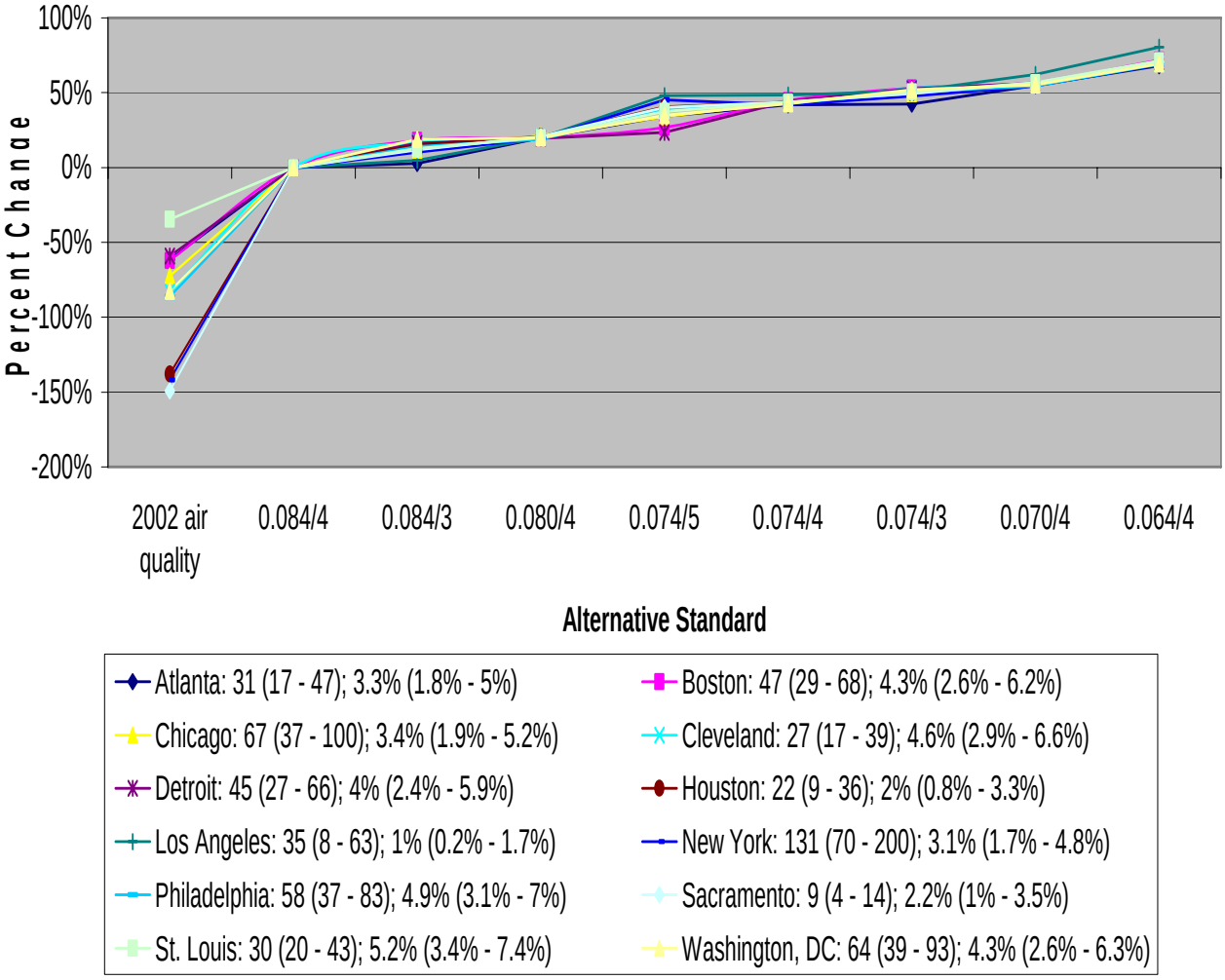
Exposure of Concern Benchmark Level	Alternative Standard Level/ Form	Percent of All Children (5-18 yrs old) (18.3 million children) -- Aggregated across 12 cities (Range for each of 12 cities)			Percent of Asthmatic Children (5-18 yrs old) (2.6 million children) – Aggregated across 12 cities (Range for each of 12 cities)		
		2002	2003	2004	2002	2003	2004
≥ 0.07 ppm	80/4	12 (1 – 28)		1 (0 – 3)	13 (0 – 31)		0 (0 – 3)
	74/5	6 (0 – 18)		0 (0 - 1)	7 (0 - 17)		0 (0 - 1)
	74/4	4 (0 – 13)	1 (0 - 2)	0 (0 - 1)	5 (0 - 14)	1 (0 - 2)	0 (0 - 1)
	74/3	3 (0 - 8)		0 (0 - 1)	3 (0 - 9)		0 (0 - 1)
	70/4	1 (0 - 5)		0 (0 - 0)	2 (0 - 6)		0 (0 - 0)
	64/4	0 (0 - 1)	0 (0 - 0)	0 (0 - 0)	0 (0 - 1)	0 (0 - 0)	0 (0 - 0)
≥ 0.06 ppm	80/4	37 (4 – 60)		6 (1 – 18)	40 (4 – 63)		6 (0 – 19)
	74/5	28 (1 – 52)		3 (0 - 11)	30 (2 - 53)		3 (0 - 11)
	74/4	25 (1 – 48)	5 (2 - 14)	2 (0 - 9)	27 (1 - 51)	7 (2 - 14)	2 (0 -9)
	74/3	21 (1 - 41)		1 (0 - 8)	23 (1 - 46)		1 (0 - 7)
	70/4	16 (1 - 36)		1 (0 - 4)	18 (0 - 41)		1 (0 - 3)
	64/4	5 (0 - 17)	0 (0 - 1)	0 (0 - 1)	6 (0 -16)	0 (0 - 1)	0 (0 - 1)

asthmatic school age children estimated to experience at least one decrement in $FEV_1 \geq 10\%$ in only 5 urban areas for 2002 and 2004, respectively.[26] Figures 6-5 and 6-6 show the percent reduction in risk estimates of non-accidental mortality for the 12 urban areas, for 2002 and 2004, respectively. The legend under each figure lists the estimated number of cases (and 95% credible intervals) when O_3 concentrations just meet the current standard next to the name of each location. There were two health outcomes that we evaluated in one city only, respiratory symptom days in moderate to severe asthmatic children (Boston, Table 6-9) and respiratory-related hospital admissions (New York City, Table 6-10), because the concentration-response functions were developed in these cities and we did not want to introduce additional uncertainties by applying these functions to other locations. We believe, however, that it is reasonable to assume that these results would be generally applicable to other locations.

As shown in Figures 6-1 and 6-2, we first note that just meeting the 80/4 scenario is estimated to result in about a 20% reduction in the number of all school age children estimated to experience moderate lung function decrements (≥ 15% reduction in FEV_1) relative to the current standard. Reducing the level of the standard to the 74/4 scenario results in about a 40 to 50% reduction in the number of all school age children estimated to experience moderate lung function decrements, depending on whether 2002 or 2004 air quality is the basis for adjustment. As shown in Figures 6-3 and 6-4, for asthmatic school age children, reducing the level of the standard to the 74/4 scenario results in about a 30 to 45% reduction in estimated risks across the 5 urban areas relative to the current standard. An alternative standard set at the 64/4 scenario provides an appreciably greater reduction of about 65 to 80% in the number of all school age children estimated to experience moderate lung function decrements (≥ 15% reduction in FEV_1) relative to the current standard (depending on the year adjusted and the urban area). This same 64/4 scenario reduces estimates of moderate lung function decrements (≥ 10% reduction in FEV_1) in asthmatic school age children by about 55 to 65% in most of the areas, with 1 area having reductions of about 75%.

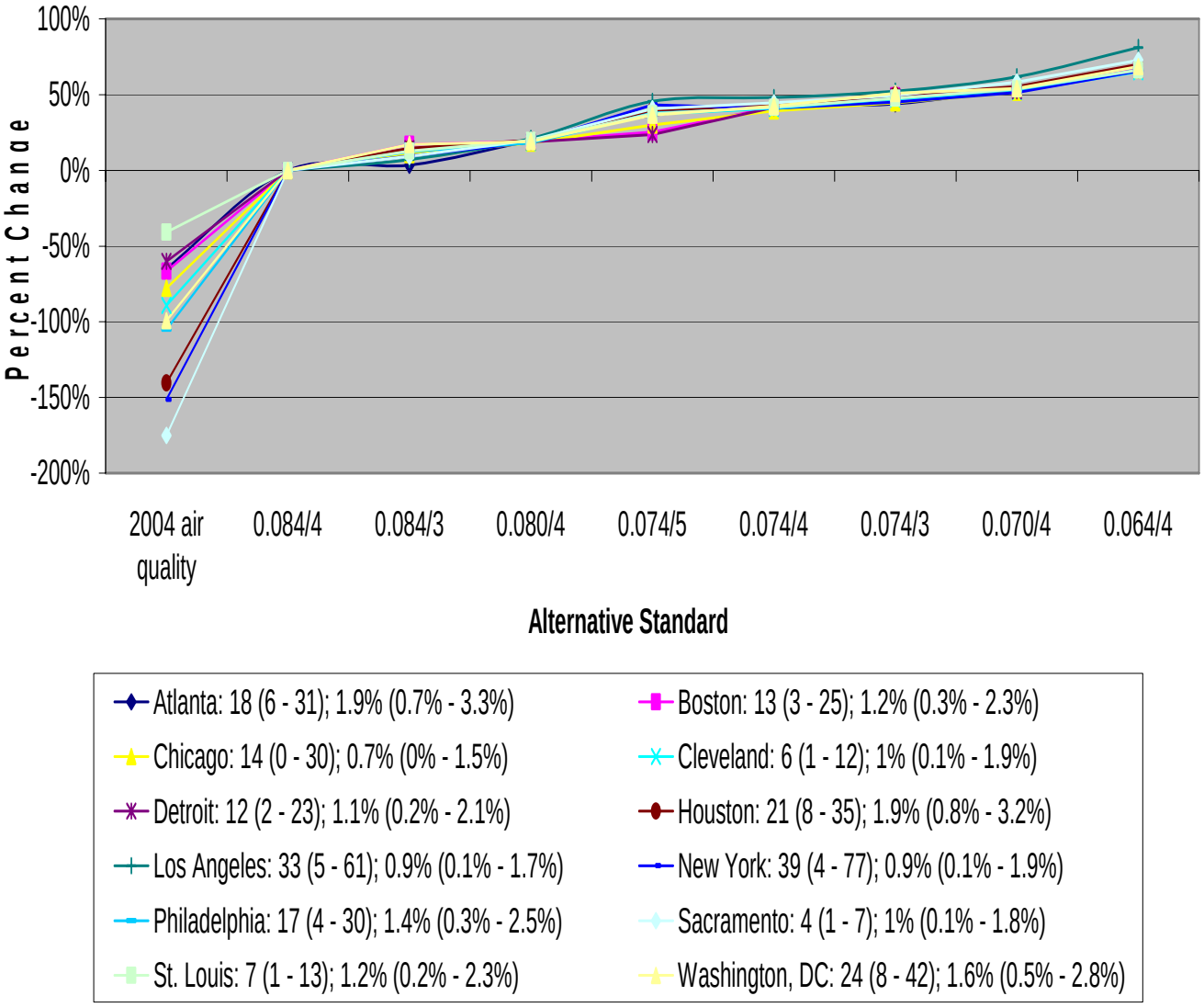
[26] The health risk assessment for lung function decrements for asthmatic school age children was conducted for 5 of the 12 urban areas and for a more limited set of alternative standards due to time constraints. The areas were selected to be geographically diverse including urban areas that are not meeting the current O_3 standard in the northeast, southeast, deep south, Midwest, and southern California.

Figure 6-1. Percent Changes in Numbers of School Age Children Experiencing at Least One Decrement in FEV1 >15% when O3 Concentrations are Reduced from Those Just Meeting the Current Standard to Those that Would Just Meet Each Alternative Standard, Based on Adjusting 2002 Data*



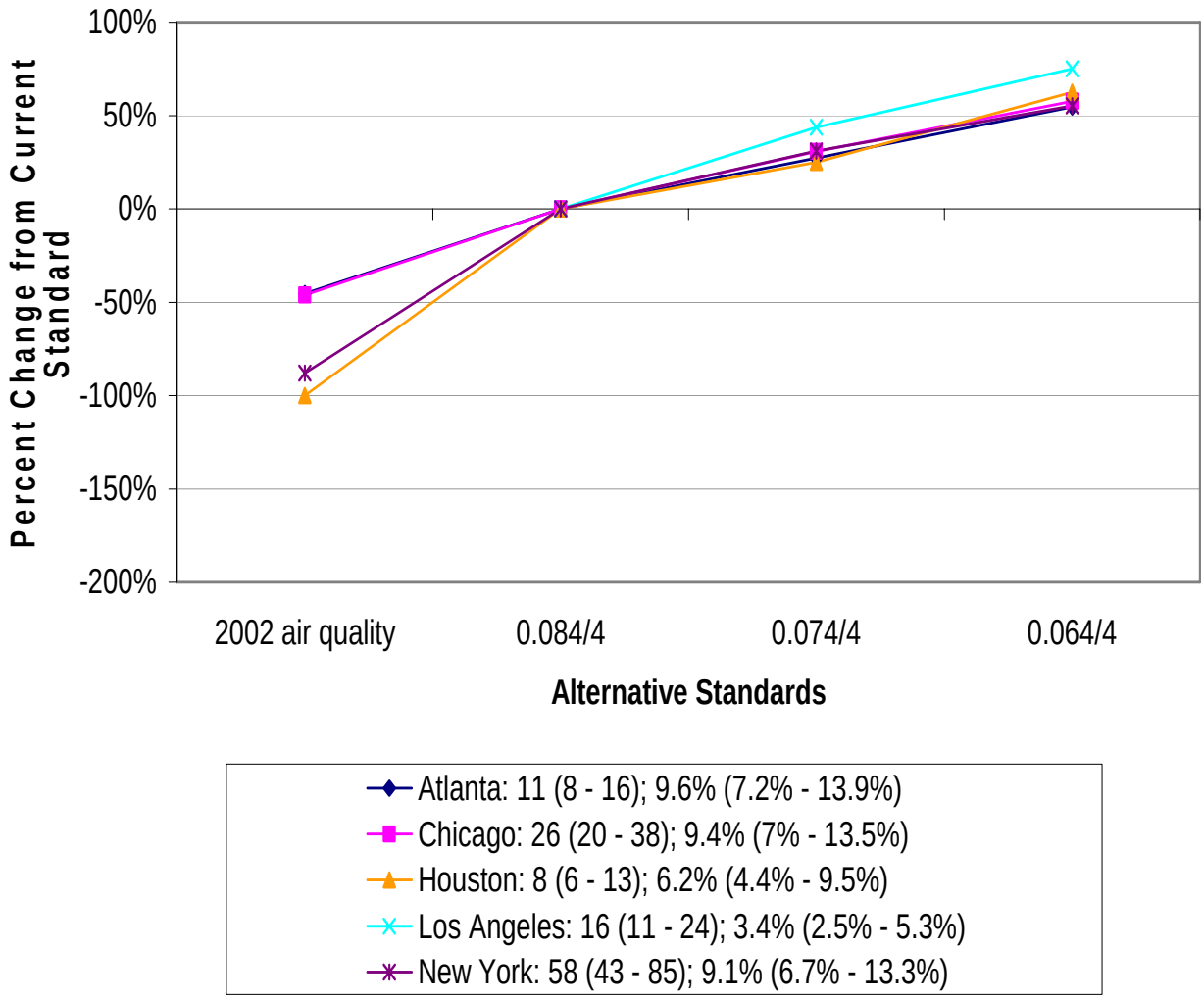
*The numbers in the box below the figure show for each urban area the number of children estimated to experience moderate lung function decrements ($FEV_1 \geq 15\%$), in thousands (and 95% credible interval) and the percent of children (and 95% credible interval) estimated to experience these effects when O_3 concentrations just meet the current 0.084/4 8-hr standard. The 8-hr average standards shown in this figure, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum form. For example, the current standard is 0.084/4 -- 0.084 ppm, 4th-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality. The percent change from the current standard (0.084/4) to a recent year of air quality was omitted for Los Angeles because it was so large in magnitude (–528% in 2002).

Figure 6-2. Percent Changes in Numbers of School Age Children Experiencing at Least One Decrement in FEV1 >15% when O3 Concentrations are Reduced from Those Just Meeting the Current Standard to Those that Would Just Meet Each Alternative Standard, Based on Adjusting 2004 Data*



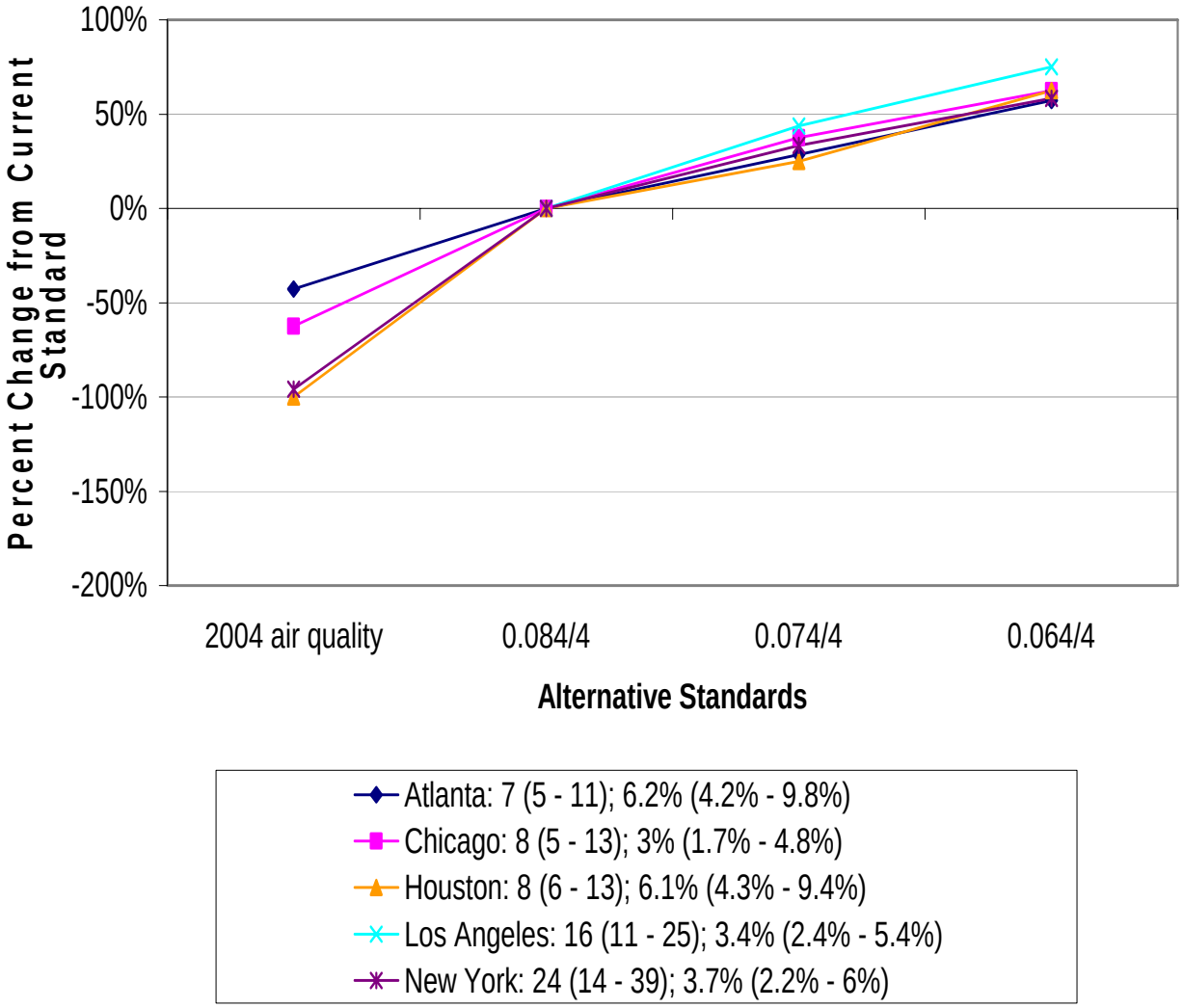
* The numbers shown in the box below the figure for each urban area represent the number of children estimated to experience moderate lung function decrements ($FEV_1 \geq 15\%$), in thousands (and 95% credible interval) and the percent of children (and 95% credible interval) estimated to experience these effects when O_3 concentrations just meet the current 0.084/4 8-hr standard. The 8-hr average standards shown in this figure, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum form. For example, the current standard is 0.084/4 -- 0.084 ppm, 4th-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality. The percent change from the current standard (0.084/4) to a recent year of air quality was omitted for Los Angeles because it was so large in magnitude (-553% in 2004).

Figure 6-3. Percent Changes in Numbers of Asthmatic School Age Children Experiencing at Least One Decrement in FEV1 >10% when O3 Concentrations are Reduced from Those Just Meeting the Current Standard to Those that Would Just Meet Each Alternative Standard, Based on Adjusting 2002 Data*



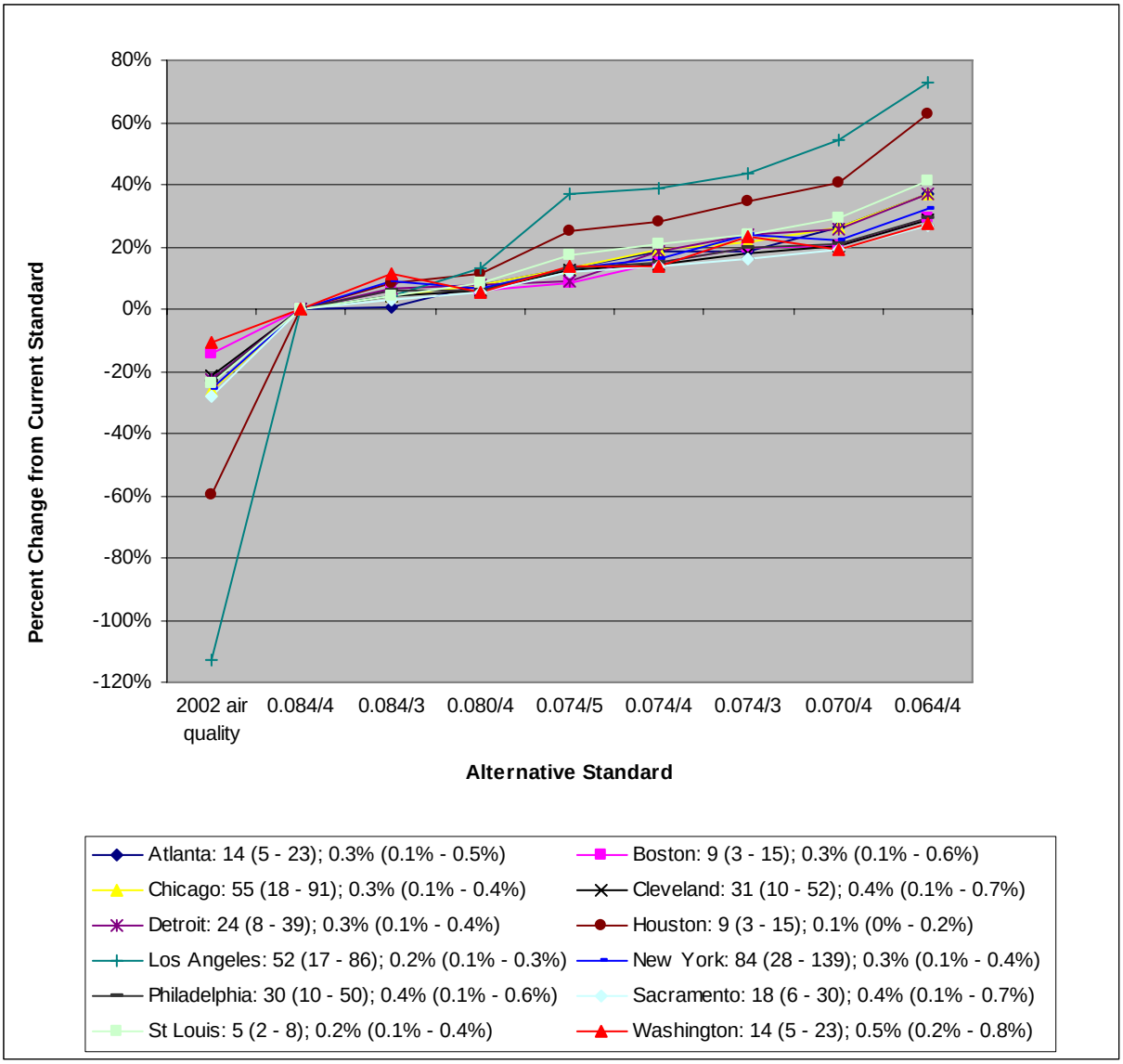
*The numbers shown in the box below the figure show for each urban area the number of asthmatic children estimated to experience moderate lung function decrements ($FEV_1 \geq 10\%$), in thousands (and 95% credible interval) and the percent of asthmatic children (and 95% credible interval) estimated to experience these effects when O_3 concentrations just meet the current 0.084/4 8-hr standard. The 8-hr average standards shown in this figure, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum form. For example, the current standard is 0.084/4 -- 0.084 ppm, 4th-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality. The percent change from the current standard (0.084/4) to a recent year of air quality was omitted for Los Angeles because it was so large in magnitude (-275% in 2002).

Figure 6-4. Percent Changes in Numbers of Asthmatic School Age Children Experiencing at Least One Decrement in FEV1 > 10% when O3 Concentrations are Reduced from Those Just Meeting the Current Standard to Those that Would Just Meet Each Alternative Standard, Based on Adjusting 2004 Data*



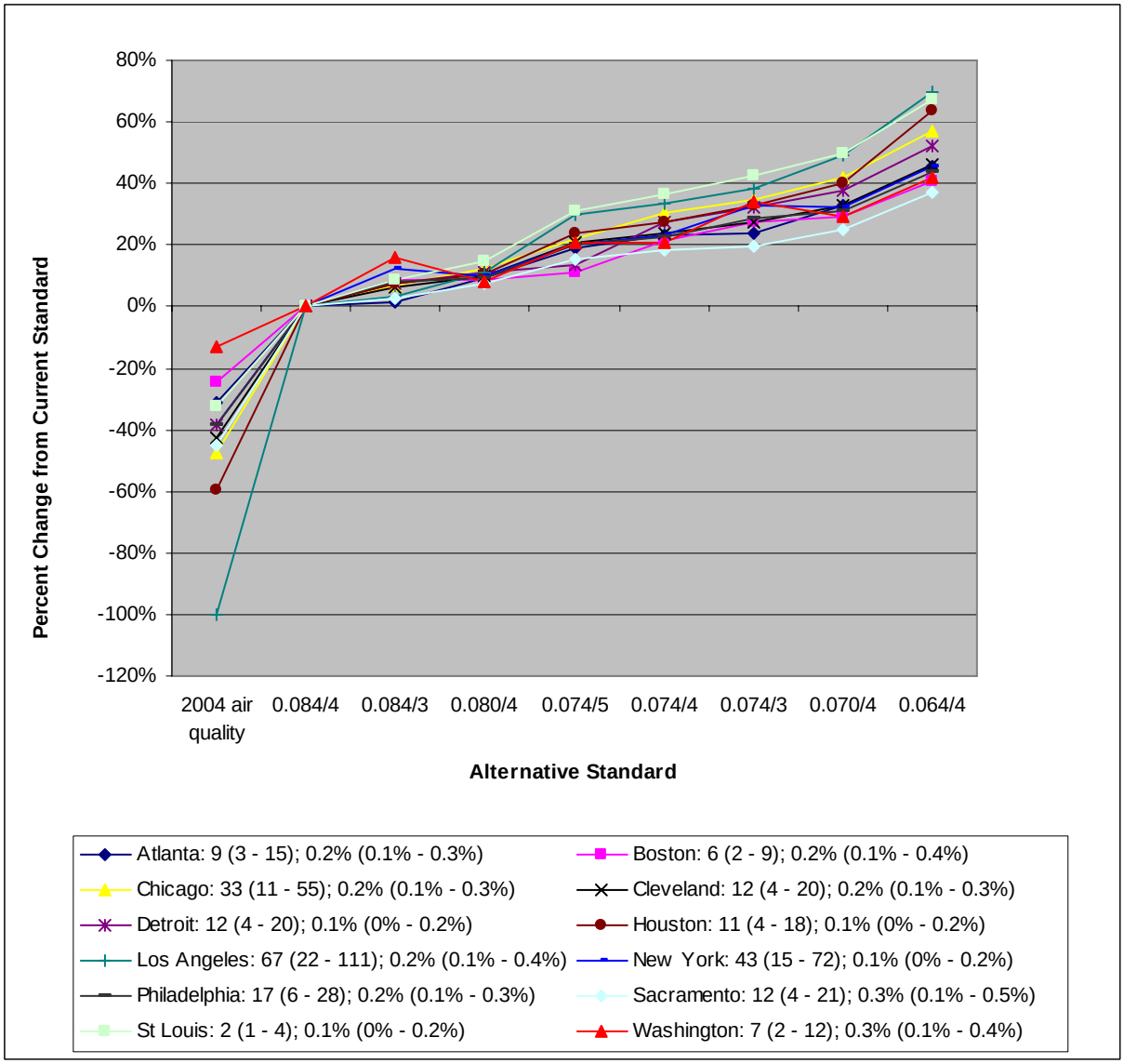
*The numbers in the box below the figure show for each urban area the number of asthmatic children estimated to experience moderate lung function decrements ($FEV_1 \geq 10\%$), in thousands (and 95% credible interval) and the percent of asthmatic children (and 95% credible interval) estimated to experience these effects when O_3 concentrations just meet the current 0.084/4 8-hr standard. The 8-hr average standards shown in this figure, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum form. For example, the current standard is 0.084/4 -- 0.084 ppm, 4^{th}-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality. The percent change from the current standard (0.084/4) to a recent year of air quality was omitted for Los Angeles because it was so large in magnitude (-281% in 2004).

Figure 6-5. Percent Changes in O3-Related Non-Accidental Mortality Incidence when O3 Concentrations are Reduced from Those Just Meeting the Current Standard to Those that Would Just Meet Each Alternative Standard, Based on Adjusting 2002 Data* (Using Bell et al., 2004 – 95 U.S. Cities)



*The numbers in the box below the figure show for each urban area the number of cases (and 95% credible interval) and the percent of total incidence (and 95% credible interval) of O_3-related non-accidental mortality when O_3 concentrations just meet the current standard (0.084/4). The 8-hr average standards shown in this figure, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum form. For example, the current standard is 0.084/4 -- 0.084 ppm, 4^{th}-highest daily maximum 8-hr average. The figure also compares the current standard to a recent year of air quality.

Figure 6-6. Percent Changes in O3-Related Non-Accidental Mortality Incidence When O3 Concentrations are Reduced from Those Just Meeting the Current Standard to Those that Would Just Meet Each Alternative Standard, Based on Adjusting 2004 Data* (Using Bell et al., 2004 – 95 U.S. Cities)



**The numbers in the box below the figure show for each urban area the number of cases (and 95% credible interval) and the percent of total incidence (and 95% credible interval) of O_3-related non-accidental mortality when O_3 concentrations just meet the current standard (0.084/4). The 8-hr average standards shown in this figure, denoted m/n, are characterized by a concentration of m ppm and an nth-highest daily maximum form. For example, the current standard is 0.084/4 -- 0.084 ppm, 4^{th}-highest daily maximum 8-hr average The figure also compares the current standard to a recent year of air quality.

As shown in Figures 6-5 and 6-6, we first note that just meeting the 80/4 scenario is estimated to result in about a 5 to 15% reduction in estimated incidences of O_3-related non-accidental mortality relative to the current standard. Reducing the level of the standard to the 74/4 scenario results in the estimated incidences of non-accidental mortality being reduced by about 15 to nearly 40% (depending on the year adjusted and the urban area) relative to the current standard. Just meeting the 64/4 scenario is estimated to provide appreciably greater reduction relative to the current standard in some areas, with an estimated reduction of about 30 to 40% in most areas and about 60 to 70% in two areas (depending on the year adjusted and the urban area) in the estimated incidences of O_3-related non-accidental mortality.

With regard to respiratory symptom days for chest tightness in moderate to severe asthmatic children in the Boston area, as shown in Table 6-9, the alternative standards provide incremental protection beyond that offered by the current standard. From the 80/4 scenario to the 64/4 scenario, the estimated incidence of respiratory symptom days is reduced by 5 to 25% in the worst year, and by approximately 7 to 31% in the best year. In the worst of the two years, the estimated percent of total incidence, or the percent of respiratory-symptom days attributable to O_3 exposure ranges from about 14% for just meeting the current standard to about 10% for the 64/4 scenario. This means that even under the most stringent alternative standard evaluated, as many as one symptom day in 10 would be estimated to be attributable to O_3 exposure in the O_3 season.

Risk estimates for respiratory-related hospital admissions attributable to O_3 exposure in New York City are shown in Table 6-10. Across the range of alternative standards from the 80/4 scenario to the 64/4 scenario, the estimated number of O_3-related hospital admissions declines by about 6 to 29% in the worst year (2002), and about 7 to 34% in the best year (2004). The percent of total respiratory-related hospital admissions attributable to O_3 exposure declines from about 1.5% for the current standard to about 1% or less for the 64/4 scenario.

Table 6-9. Risks of Respiratory Symptom Days for Chest Tightness Associated with Just Meeting the Current and Alternative Ozone Standards Based on Adjusting 2002 and 2004 Air Quality in Moderate to Severe Asthmatic Children in Boston, MA[1]

Current and Alternative Standards	Risk Metric	Average Risks of Chest Tightness Associated with Air Quality[2,3] [percent reduction from current standard]	
		2002	2004
Current Standard (84/4)	Incidence	7800	5700
	Percent of Total Incidence	14%	10%
80/4	Incidence	7400 [5% reduction]	5400 [7% reduction]
	Percent of Total Incidence	13%	10%
74/4	Incidence	6800 [12% reduction]	4900 [15% reduction]
	Percent of Total Incidence	12%	9%
70/4	Incidence	6400 [17% reduction]	4500 [22% reduction]
	Percent of Total Incidence	12%	8%
64/4	Incidence	5900 [25% reduction]	3900 [31% reduction]
	Percent of Total Incidence	10%	7%

[1]It is estimated that there are 25,000 children with moderate to severe asthma in the Boston area.
[2]Incidence rounded to nearest 100.
[3]Average of median estimates for models using lag 0 and lag 1 day and O_3 only and including $PM_{2.5}$ in the model.

Table 6-10. Risks of Hospital Admissions for Respiratory Illness Associated with Just Meeting the Current and Alternative Ozone Standards Based on Adjusting 2002 and 2004 Air Quality in New York, NY

Current and Alternative Standards	Risk Metric	Hospital Admissions for Respiratory Illness Associated with Ozone Exposures[1,2] [percent reduction from current standard]	
		2002	2004
Current Standard (84/4)	Incidence	513	366
	Percent of Total Incidence	1.5%	1.0%
80/4	Incidence	483 [6% reduction]	341 [7% reduction]
	Percent of Total Incidence	1.4%	1.0%
74/4	Incidence	439 [14% reduction]	304 [17% reduction]
	Percent of Total Incidence	1.2%	0.9%
70/4	Incidence	410 [20% reduction]	278 [24% reduction]
	Percent of Total Incidence	1.2%	0.8%
64/4	Incidence	365 [29% reduction]	241 [34% reduction]
	Percent of Total Incidence	1.0%	0.7%

[1]Incidence rounded to nearest whole number.
[2]95 % credible intervals based on statistical uncertainty surrounding the O_3 coefficient are presented in tables in Appendix 5C of this Staff Paper.

6.3.4.3 CASAC and Public Commenters' Views on the Level of the Standard

As noted above in section 6.3.1.4, staff recognizes that the exposure- and risk-based information can be considered both in terms of whether the risks estimated to remain upon attaining the current standard are important from a public health perspective and/or whether additional reductions in risk estimated to be associated with alternative, more protective standards are important from a public health perspective. Judgments about the importance of the estimates of exposures and risks need to take into account the important uncertainties associated with such estimates. We recognize that public health policy judgments, including the weight to place on various types of evidence and how to weigh the importance of estimated risks in a public health perspective, are ultimately decisions left to the Administrator. To help inform those judgments with regard to the level of the primary O_3 standard, the views expressed by CASAC as well as the views of interested parties who have commented on earlier drafts of this document are summarized here.

As stated in its letter to the Administrator, "*the CASAC unanimously recommends that the current primary ozone NAAQS be revised and that the level that should be considered for the revised standard be from 0.060 to 0.070 ppm*" (Henderson, 2006c, p. 5). The CASAC coupled this recommended range of levels with a range of forms, as discussed in the next section below. This recommendation follows from their more general recommendation, discussed above in section 6.3.2, that the current standard of 0.08 ppm needs to be substantially reduced to be protective of human health, particularly in sensitive subpopulations. The lower end of this range reflects CASAC's views that "[W]hile data exist that adverse health effects may occur at levels lower than 0.060 ppm, these data are less certain and achievable gains in protecting human health can be accomplished through lowering the ozone NAAQS to a level between 0.060 and 0.070 ppm." (id.).

The same group of commenters that expressed the view that the current primary O_3 standard is not adequate also submitted comments that supported revising the level of the primary O_3 standard to within the same or even lower range of levels than the range recommended by CASAC. The basis for these commenters' views on the level of the standard is generally reflected in the discussion above on the basis for their views on the adequacy of the current standard and in the rationale given by CASAC. In addition, some of these commenters also noted that the World Health Organization's guidelines for O_3 air quality are in the range of 51 to 61 ppb. The other group of commenters who expressed the view that the current standard is adequate did not provide any provisional views on alternative levels that would be appropriate for consideration should the Administrator consider revisions to the standard.

6.3.4.4 Staff Conclusions on the Level of the Standard

Staff's consideration of alternative levels of the primary O_3 standard builds upon our conclusion, discussed above in section 6.3.1, that the overall body of evidence clearly calls into question the adequacy of the current standard in protecting sensitive groups, notably including asthmatic children and other people with lung disease, as well as all children and older adults, especially those active outdoors, and outdoor workers, against an array of adverse health effects that range from decreased lung function to serious indicators of respiratory morbidity including ED visits and hospital admissions for respiratory causes, and possibly cardiovascular-related effects and mortality. Thus, we believe that the available information provides strong support for consideration of a range of standard levels that is below the level of the current standard so as to afford increased health protection for these sensitive groups. We have also concluded, as an initial matter, that it is appropriate to consider specifying the level of the O_3 standard to the nearest thousandth ppm.

As discussed above in section 6.3.4.1., based on the evidence, we conclude that it is appropriate to consider a range of levels for the primary O_3 standard from somewhat below 0.080 ppm down to at least as low as 0.060 ppm. This evidence-based recommendation takes into account information on lowest-observed-effects levels in controlled human exposure studies as well as indications of possible effects thresholds reported in some epidemiological studies and questions of biological plausibility in attributing associations observed down to background levels to O_3 exposures alone. The upper end of this range is somewhat below the lowest-observed-effects level for effects such as pulmonary inflammation, increased airway responsiveness and impaired host-defense capabilities in healthy adults while at prolonged moderate exertion. These effects have been linked to aggravation of asthma and increased susceptibility to respiratory infection, potentially leading to increased medication use, increased school and work absences, increased visits to doctors' offices and EDs, and increased hospital admissions, and pulmonary inflammation is also related to increased cellular permeability in the lung, which may be a mechanism by which O_3 exposure can lead to cardiovascular system effects, and to potential chronic effects such as chronic bronchitis or long-term damage to the lungs that can lead to reduced quality of life. These indicators of adverse O_3-related morbidity effects lend plausibility to the adverse morbidity effects and mortality effects observed in epidemiological studies reporting statistically significant associations with ambient O_3 concentrations that range down to levels well below 0.080 ppm. The lower end of this range reflects the lowest-observed-effects level for potentially adverse lung function decrements and respiratory symptoms in some healthy adults, 0.060 ppm, which is also a level likely to cause these adverse effects in sensitive groups, and is above the level where there is some indication of possible effects thresholds in epidemiological studies.

Having reached this evidence-based conclusion on an appropriate range of levels for consideration, we have also focused on considering the public health implications of selecting different levels within this range (< 0.08 to 0.060 ppm O_3). In so doing, we have looked to the results of the analyses of exposure and risk for the 74/4 scenario to represent the public health impacts of selecting a standard level in the upper part of the range, the results of the analyses of the 70/4 scenario to represent the impacts in the middle part of the range, and the results of the analyses of the 64/4 scenario to represent the lower part of the range.

As discussed above in section 6.3.4.2, for each of these alternative standard levels, we have considered exposures of concern at the two benchmark levels discussed above (i.e., 0.070 ppm and 0.060 ppm), that serve as indicators of health outcomes for which there is insufficient information to do quantitative risk assessments. We have also considered the quantitative estimates of risk for moderate lung function decrements in all and asthmatic children, respiratory symptom days in moderate to severe asthmatic children, respiratory-related hospital admissions, and non-accidental mortality. In considering both exposures of concern and quantitative risk estimates, we again note that there is substantial year-to-year variability across the three years included in this analysis (2002 to 2004) in the estimates of the number of children and the number of occurrences of exposures of concern at the benchmark levels and in the quantitative risk estimates. We also note the substantial city-to-city variability in these estimates of exposures of concern and quantitative risk. We believe that it is appropriate and important to consider not just the average estimates across all years or all cities included in the analyses, but to consider the public health impacts in years and locations with relatively poorer air quality and in locations receiving relatively less protection from any alternative standard.

We turn now to considering the public health implications of setting the standard in the upper, middle and lower parts of the range. A standard set in the upper part of this range (e.g., the 74/4 scenario) would result in an aggregate estimate of about 4% of all school age children[27] (~ 770,000 children in 12 urban areas) likely to experience exposures of concern at the $\geq$ 0.070 ppm benchmark level in the worst (2002) of the 3 years evaluated, while the estimates range up to 13% of all school age children (~ 150,000 children) in the single city with the least degree of protection from this standard. In the mid-year (2003), in aggregate about 1% of all school age children (~110,000 children) are estimated to experience exposures of concern at this level; in the city with the least degree of protection from this standard the estimate is less than 2% of all school age children (~ 26,000 children). At the benchmark level of $\geq$ 0.060 ppm, in aggregate in

[27] We note that the percent of all school age children and asthmatic school age children estimated to experience exposures of concern (aggregate and individual city estimates) are very similar, and the results for all school age children are presented in the exhibits in Chapter 4, thus for ease of discussion we present results for all school age children here.

the worst year about 25% of all school age children (~ 4.4 million children) are estimated to experience exposures of concern; this estimate ranges up to about 48% of all school age children (~ 1 million children) in the single city with the least degree of protection from this standard. Even in the mid-year, in aggregate about 7% of all school age children (~ 1.3 million children) are estimated to experience exposures of concern, ranging up to 15% of all school age children (~290,000 children) in the single city with the least degree of protection from this standard. A standard set at this level would reduce the number of all and asthmatic school age children estimated to experience one or more moderate lung function decrements by about 30 to 50% relative to the current standard (Figures 6-1 through 6-4), with city-to-city differences accounting for most of the variability in estimates. A standard set at this level would reduce non-accidental mortality by about 10 to 40%, with most of the variability occurring across the 12 city estimates (Figures 6-5 and 6-6).

There were two health outcomes that we evaluated in one city only, respiratory symptom days in moderate to severe asthmatic children (Boston, Table 6-9) and respiratory-related hospital admissions (New York City, Table 6-10). In the worst year, a standard set at this level (the 74/4 scenario) is estimated to reduce the incidence of symptom days in children[28] with moderate to severe asthma in the Boston area to 6,800 days, a 12% reduction relative to the current standard. Even with this reduction, it is estimated that 1 respiratory symptom day in 8 during the O_3 season would be attributable to O_3 exposure. Estimated incidence of respiratory-related hospital admissions was reduced by 14 to 17% by a standard set at this level relative to the current standard, in the year with worst and best air quality respectively. A standard set at this level reduces exposures of concern at the $\geq$ 0.070 benchmark level much more than exposures of concern at the $\geq$ 0.06 ppm benchmark level, placing relatively less weight on the evidence from the controlled human exposure studies showing lung function decrements and respiratory symptoms in some healthy adults at 0.060 ppm O_3, as well as evidence from epidemiological studies showing an array of respiratory morbidity effects occurring at levels well below the current standard. It would place relatively more weight on the uncertainties associated with the exposure and risk estimates, suggesting less importance for the implications of exposures at the 0.060 ppm benchmark level from a public health policy perspective.

A standard set in the middle part of this range (e.g., the 70/4 scenario) would result in an aggregate estimate of about 1% of all school age children (> 250,000 children in 12 urban areas) likely to experience exposures of concern at the $\geq$ 0.070 ppm benchmark level even in the worst year (2002); in the city with the least protection about 5% of all school age children (~ 64,000

[28] Since there are estimated to be about 25,000 moderate to severe asthmatic children in the Boston area, this incidence rate is per 25,000 children.

children) are estimated to experience exposures of concern at this level.[29] At the benchmark level of $\geq$ 0.060 ppm, in aggregate in the worst year about 16% of all school age children (~ 3 million children) are estimated to experience exposures of concern; this number ranges up to 36% of all school age children (~ 600,000 children) in the single city with the least degree of protection from this standard. A standard set at this level would reduce the number of all school age children[30] estimated to experience one or more moderate lung function decrements by about 50 to 65% over the current standard, with city-to-city differences accounting for most of the variability in estimates. A standard set at this level would reduce non-accidental mortality by about 20 to 55%, with most of the variability occurring across the 12 city estimates. In the worst year, a standard set at this level is estimated to reduce the incidence of symptom days in children with moderate to severe asthma in the Boston area only slightly over the standard set at the upper end of the range, to 6,400 days (a 12% reduction). With this reduction, it is estimated that about 1 respiratory symptom day in 8 during the O_3 season would be attributable to O_3 exposure. Estimated incidence of respiratory-related hospital admissions was reduced by about 20 to 24% in the year with worst and best air quality, respectively.

A standard set in the middle part of the recommended range, as indicated by the estimates for the 70/4 scenario, would reduce the exposures of concern at the 0.070 ppm level substantially over the current standard, even in the worst of the three years and in the city with the least degree of protection. However, it reduces exposures of concern at the 0.060 ppm benchmark level much less so, leaving relatively large percentages of all school age children unprotected in the worst year or the city with the least protection from this standard. It provides incremental additional protection for members of sensitive groups, over the current O_3 standard, against respiratory morbidity effects such as lung function decrements, respiratory symptom days and hospital admissions, as well as non-accidental mortality.

A standard set in the lower part of this range (e.g., the 64/4 scenario) would result in an aggregate estimate of less than 0.5% of all school age children (~ 30,000 children) likely to experience exposures of concern at the 0.070 ppm benchmark level in the worst year and only 1% of all school age children (9,900 children) in the city with the least degree of protection from this standard. In the mid-year (2003), estimates of exposures of concern go close to zero, even in the city with the least degree of protection. At the benchmark level of 0.060 ppm, in aggregate in the worst year about 5% of all school age children (~ 950,000 children) are estimated to experience exposures of concern; this number ranges up to 17% of all school age children

[29] Estimates were not developed for the mid-year (2003) for this alternative standard.

[30] Estimates for asthmatic children were not developed for this alternative standard.

(~200,000 children) in the city with the least degree of protection from this standard. In the mid-year exposures of concern at this level are reduced substantially, resulting in an aggregate estimates of less than 0.5% of all school age children (~ 70,000 children), ranging up to only 1% of all school age children (~ 13,000 children) in the city with the least degree of protection from this standard. A standard set at this level would reduce the number of all and asthmatic school age children estimated to experience one or more moderate lung function decrements by about 50 to 80% over the current standard, and non-accidental mortality by about 25 to 75%, with most of the variability occurring across the 12 city estimates. In the worst year, a standard set at this level is estimated to reduce the incidence of symptom days in children with moderate to severe asthma in the Boston area to 5,900 days, about a 25% reduction over the current standard. But even with this reduction, it is estimated that 1 respiratory symptom day in 10 during the O_3 season is attributable to O_3 exposure. Estimated incidence of respiratory-related hospital admissions was reduced by 30 to 35% over the current standard, in the year with worst and best air quality respectively.

These results indicate that setting a standard in the lower part of the range would essentially eliminate exposures of concern at the benchmark level of 0.070 ppm, even in the worst of the three years and in the city with the least degree of protection. It would also substantially reduce exposures of concern at the benchmark level of 0.060 ppm, especially in the mid-year of the three years evaluated. It provides additional incremental protection for members of sensitive groups over the current O_3 standard and the alternative standards at the upper to middle part of the range, against respiratory morbidity effects such as lung function decrements, respiratory symptom days and hospital admissions, as well as non-accidental mortality. A standard set in the lower part of the range would place relatively more weight on the evidence from the controlled human exposure studies showing lung function decrements and respiratory symptoms in some healthy adults at 0.060 ppm O_3, as well as evidence from epidemiological studies showing an array of respiratory morbidity effects occurring at levels below the current standard. It would place relatively less weight on the uncertainties associated with the exposure and risk estimates, and reflect the greater importance, from a policy perspective, of the public health implications of exposures at the 0.060 ppm benchmark level.

The CASAC recommended a range down to 0.060 ppm for the level of the O_3 standard, noting that "achievable gains in protecting public health" (Henderson 2006c, p. 5) can be accomplished by setting the level of the standard down to 0.060 ppm O_3. The results of the exposure and risk assessments support this recommendation. Staff concludes that important improvements in protecting the health of sensitive groups can be made by setting the level of the O_3 standard within the range of < 0.080 ppm to 0.060 ppm O_3. Standard levels within this range considered in staff exposure and risk assessments include 0.074, 0.070, and 0.064 ppm, which

are representative of levels within the upper, middle, and lower parts of this range, respectively. Moreover, these assessment results indicate that even in the lower end of the range there are benefits to the health of sensitive groups that warrant consideration.

To provide some perspective on the implications of alternative 8-hr primary standards (within the range of levels recommended above and within the range of forms discussed in the next section below), staff assessed (based on 2002 and 2004 air quality data) the percentage of counties, and the populations in those counties, that likely would not attain various 8-hr O_3 standards. This assessment, shown in Appendix 6B for various forms and levels of the 8-hr standards, was not considered as a basis for the above staff conclusions and recommendations.

6.3.5 Form

In 1997 the primary O_3 NAAQS was changed from a "1-expected-exceedance" form[31] to a concentration-based statistic, specifically the 3-year average of the annual fourth-highest daily maximum 8-hr concentrations. The principal advantage of the concentration-based form is that it is more directly related to the ambient O_3 concentrations that are associated with the health effects. With a concentration-based form, days on which higher O_3 concentrations occur would weigh proportionally more than days with lower concentrations, since the actual concentrations are used in determining whether the standard is attained. That is, given that there is a continuum of effects associated with exposures to varying levels of O_3, the extent to which public health is affected by exposure to ambient O_3 is related to the actual magnitude of the O_3 concentration, not just whether the concentration is above a specified level.

In evaluating alternative forms for the primary standard in conjunction with specific standard levels, staff considers the adequacy of the public health protection provided by the combination of the level and form to be the foremost consideration. In addition, we recognize that it is important to have a form of the standard that is stable and insulated from the impacts of extreme meteorological events that are conducive to O_3 formation. Such instability can have the effect of reducing public health protection, because frequent shifting in and out of attainment due of meteorological conditions can disrupt an area's ongoing implementation plans and associated control programs. Providing more stability is one of the reasons that EPA moved to a concentration-based form in 1997.

During the 1997 review, consideration was given to a range of alternative forms, including the second-, third-, fourth- and fifth-highest daily maximum 8-hr concentrations in an O_3 season, recognizing that the public health risks associated with exposure to a pollutant

[31]The 1-expected-exceedance form essentially requires that the fourth-highest air quality value in 3 years, based on adjustments for missing data, be less than or equal to the level of the standard for the standard to be met at an air quality monitoring site.

without a clear, discernable threshold can be appropriately addressed through a standard that allows for multiple exceedances to provide increased stability, but that also significantly limits the number of days on which the level may be exceeded and the magnitude of such exceedances. Consideration was given to setting a standard with a form that would provide a margin of safety against possible, but uncertain chronic effects, and would also provide greater stability to ongoing control programs. The fourth-highest daily maximum was selected because it was decided that the differences in the degree of protection against potential chronic effects afforded by the alternatives within the range were not well enough understood to use any such differences as a basis for choosing the most restrictive forms. On the other hand, the relatively large percentage of sites that would experience O_3 peaks well above 0.08 ppm and the number of days on which the level of the standard may be exceeded even when attaining a fifth-highest 0.08 ppm concentration-based standard, argued against choosing that form.

In selecting alternative standards to include in our exposure and risk analyses, we considered two concentration-based forms, the nth-highest maximum concentration and a percentile-based form. A percentile-based statistic is useful for comparing datasets of varying length because it samples approximately the same place in the distribution of air quality values, whether the dataset is several months or several years long. However, a percentile-based form would allow more days with higher air quality values in locations with longer O_3 seasons relative to places with shorter O_3 seasons. An nth-highest maximum concentration form would more effectively ensure that people who live in areas with different length O_3 seasons receive the same degree of public health protection. For this reason, our exposure and risk analyses were based on a form specified in terms of an nth-highest concentration, with n ranging from 3 to 5.

The results of some of these analyses are shown in Figures 6-1 through 6-4, discussed above in section 6.3.4.2. These figures illustrate the estimated percent change in risk estimates for the incidence of moderate or greater decrements in lung function ($\geq$ 15% FEV_1) in all school age children and moderate or greater lung function decrements ($\geq$ 10% FEV_1) in asthmatic school age children, associated with going from meeting the current standard to meeting alternative standards with alternative forms. Figures 6-5 and 6-6 illustrate the estimated percent change in the estimated incidence of non-accidental mortality, associated with going from meeting the current standard to meeting alternative standards. These results are generally representative of the patterns found in all of the analyses. The estimated reductions in risk associated with different forms of the standard, ranging from third- to fourth-highest daily maximum concentrations at 0.084 ppm, and from third- to fifth-highest daily maximum concentrations at 0.074 ppm, are generally less than the estimated reductions associated with the different levels that were analyzed. As seen in these figures, there is much city-to-city variability, particularly in the percent changes associated with going from a fourth-highest to

third-highest form at the current level of 0.084 ppm, and with estimated reductions associated with the fifth-highest form at a 0.074 ppm level. In most cities, there are generally only small differences in the estimated reductions in risks associated with the third- to fifth-highest forms at a level of 0.074 ppm.

In their letter to the Administrator, CASAC recommended that "*a range of concentration-based forms from the third- to the fifth-highest daily maximum 8-hr average concentration"* be considered (Henderson, 2006c, p. 5). Further, CASAC recommended that the Agency conduct a broader evaluation of alternative concentration-based forms to evaluate the implications of a broader range of alternative forms on public health protection and stability (i.e., with respect to yearly variability to ensure a stable target for control programs).

The same group of commenters that expressed the view that the current primary O_3 standard is not adequate also submitted comments that supported a more health-protective form of the standard than the current form (e.g., a second- or third-highest daily maximum form). The other group of commenters who expressed the view that the current standard is adequate did not provide any provisional views on alternative forms that would be appropriate for consideration should the Administrator consider revisions to the standard.

Staff notes that there is not a clear health-based threshold for selecting a particular nth-highest daily maximum form of the standard from among the ones analyzed.[32] We further note that the changes in the form considered in our analyses result in only small differences in the estimated reductions in risks in most cities, although in some cities larger differences are estimated.

Staff concludes that a range of concentration-based forms from the third- to the fifth-highest daily maximum 8-hr average concentration is appropriate for consideration in setting the standard. Given that there is a continuum of effects associated with exposures to varying levels of O_3, the extent to which public health is affected by exposure to ambient O_3 is related to the actual magnitude of the O_3 concentration, not just whether the concentration is above a specified level. The principal advantage of a concentration-based form is that it is more directly related to the ambient O_3 concentrations that are associated with health effects. Robust, concentration-based forms, in the range of the third- to fifth-highest daily maximum 8-hr average concentration, including the current 4^{th}-highest daily maximum form, minimize the inherent lack of year-to-year stability of exceedance-based forms and provide insulation from the impacts of

[32] Staff consideration of an alternative form based on looking at the nth-highest daily maximum 8-hr average concentration over three years (specifically the 12^{th}-highest value in three years), rather than the current form that is based on the 3-year average of annual nth-highest concentrations, did not identify a specific alternative form that was appreciably more consistent across areas than the range of forms previously considered.

extreme meteorological events. Such instability can have the effect of reducing public health protection by disrupting ongoing implementation plans and associated control programs.

6.3.6 Summary of Staff Conclusions and Recommendations on the Primary O_3 NAAQS

Staff conclusions and recommendations on the elements of the primary O_3 standard for the Administrator's consideration in making decisions on the primary O_3 standard are summarized below, together with supporting conclusions from sections 6.3.1 to 6.3.5 above. These standard elements, including indicator, averaging time, level, and form, collectively determine the public health protection afforded by the standard.

We recognize that selecting from among alternative standards will necessarily reflect consideration of qualitative and quantitative uncertainties inherent in the relevant evidence and in the assumptions of the quantitative exposure and risk assessments. Any such standard should protect public health against health effects associated with exposure to O_3, alone or in combination with related photochemical oxidants, taking into account both evidence-based and exposure- and risk-based considerations, and the nature and degree of uncertainties in such information. In recommending these ranges of alternative standards for consideration, we are mindful that the Act requires standards that, in the judgment of the Administrator, are requisite to protect public health with an adequate margin of safety. The standards are to be neither more nor less stringent than necessary. Thus, the Act does not require that NAAQS be set at zero-risk levels, but rather at levels that reduce risk sufficiently to protect public health with an adequate margin of safety.

(1) It is appropriate to continue to use O_3 as the indicator for a standard that is intended to address effects associated with exposure to O_3, alone or in combination with related photochemical oxidants. Based on the available information, and consistent with the views of CASAC and public commenters, we concluded that there is no basis for considering any alternative indicator at this time. Staff notes that while the new body of time-series epidemiological evidence cannot resolve questions about the relative contribution of other photochemical oxidant species to the range of morbidity and mortality effects associated with O_3 in these types of studies, control of ambient O_3 levels is generally understood to provide the best means of controlling photochemical oxidants in general, and thus of protecting against effects that may be associated with individual species and/or the broader mix of photochemical oxidants, independent of effects specifically related to O_3.

(2) It is appropriate to continue to use an 8-hr averaging time for the primary O_3 standard. We conclude that a standard with an 8-hr averaging time can effectively limit both 1- and 8-hr exposures of concern and that an 8-hr averaging time is appropriate to provide adequate and more uniform protection of public health from both short-term (1- to 3-hr) and prolonged (6- to 8-hr) exposures to O_3 in the ambient air. Therefore, we recommend retaining the 8-hr averaging time and do not recommend consideration of a separate 1-hr standard at this time. We also conclude that consideration of a standard with a longer-term averaging time (e.g., annual) is not warranted at this time.

(3) We conclude that the overall body of evidence clearly calls into question the adequacy of the current standard and provides strong support for consideration of an O_3 standard that would provide increased health protection for sensitive groups, including asthmatic children and other people with lung disease, as well as all children and older adults, especially those active outdoors, and outdoor workers, against an array of adverse health effects that range from decreased lung function and respiratory symptoms to serious indicators of respiratory morbidity including ED visits and hospital admissions for respiratory causes, and possibly cardiovascular-related effects and mortality. We also conclude that risks projected to remain upon meeting the current standard, based on the exposure and risk assessment, are indicative of risks to sensitive groups that can reasonably be judged to be important from a public health perspective, which reinforces our conclusion that consideration should be given to revising the level of the standard so as to provide increased public health protection.

   (a) We recommend that consideration be given to a standard level within the range of somewhat below 0.080 ppm to 0.060 ppm, reflecting our judgment that a standard set within this range could provide an appropriate degree of public health protection and would result in important improvements in protecting the health of sensitive groups. Standard levels within this range that were considered in staff analyses of air quality, exposure, and risk include 0.074, 0.070, and 0.064 ppm, representative of levels within the upper, middle, and lower parts of this range, respectively.

   (b) We further recommend that consideration be given to specifying the level of the primary standard to the nearest thousandth ppm, reflecting the degree of precision with which ambient O_3 concentrations can be measured and design values can be calculated.

(4) We conclude that it is appropriate to consider a form in the range of the annual third- to fifth-highest daily maximum 8-hr average concentration, which includes the current form of the annual fourth-highest daily maximum 8-hr average concentration, averaged over three years. It is appropriate to consider a form within this range in conjunction with a standard level within the recommended range, so as to provide an appropriate degree of increased public health protection.

6.4 SUMMARY OF KEY UNCERTAINTIES AND RESEARCH RECOMMENDATIONS RELATED TO SETTING A PRIMARY O_3 STANDARD

We believe it is important to continue to highlight the uncertainties associated with establishing standards for O_3 during and after completion of the NAAQS review process. Research needs go beyond what is necessary to understand health and welfare effects, population exposures, and the risks of exposure for purposes of setting standards. Research can also support the development of more efficient and effective control strategies. It should be noted, however, that a thorough discussion of research needs related to control strategy development is beyond the scope of this document.

Following completion of the 1996 Ozone Staff Paper (U.S. EPA, 1996), the EPA held a research needs workshop and produced a draft document[33] for review by the CASAC at a public meeting held November 16, 1998. Based on our review of scientific information contained in the 2006 CD, we have concluded that O_3 health research needs and priorities have not changed substantially since the above document was written. Key uncertainties and research needs that continue to be high priority for future reviews of the health-based primary standards are identified below:

(1) An important aspect of risk characterization and decision making for air quality standard levels for the O_3 NAAQS is the characterization of the shape of exposure-response functions for O_3, including the identification of potential population threshold levels. Recent controlled human exposure studies conducted at levels below 0.08 ppm O_3 provide evidence that measurable lung function effects occur in some individuals for 6-8 hr exposures in the range of 0.08 to as low as 0.04 ppm. A major limitation of these data is that they were collected in one laboratory located in an area of the U.S. that typically experiences higher ambient air levels of O_3; therefore, prior attenuation of subject response may have been a factor in the responses observed. Considering the importance of estimating health risks in the range of 0.04 to 0.08 ppm O_3, additional research is

[33] "Ozone Research Needs to Improve Health and Ecological Risk Assessment" (U.S. EPA, 1998).

needed to evaluate responses in healthy and asthmatic individuals in the range of 0.04 to 0.08 ppm for 6-8 hr exposures while engaged in moderate exertion.

(2) Similarly, for health endpoints reported in epidemiological studies such as hospital admissions, ED visits, and premature mortality, an important aspect of characterizing risk is the shape of concentration-response functions for O_3, including identification of potential population threshold levels. Most of the recent studies and analyses continue to show no evidence for a clear threshold in the relationships between O_3 levels and these health endpoints or have suggested that any such thresholds must be at very low levels approaching policy relevant background levels. Whether or not exposure errors, misclassification of exposure, or potential impacts of other copollutants may be obscuring potential population thresholds is still unknown.

(3) The extent to which the broad mix of photochemical oxidants and more generally other copollutants in the ambient air (e.g., PM, NO_2, SO_2, etc.) may play a role in modifying or contributing to the observed associations between ambient O_3 and various morbidity effects and mortality continues to be an important research question. Ozone has long been known as an indicator of health effects of the entire photochemical oxidant mix in the ambient air and has served as a surrogate for control purposes. A better understanding of sources of the broader pollutant mix, of human exposures, and of how other pollutants may modify or contribute to the health effects of O_3 in the ambient air, and vice versa, is needed to better inform future NAAQS reviews.

(4) As epidemiological research has become a more important factor in assessing the public health impacts of O_3, methodological issues in epidemiological studies have received greater visibility and scrutiny. Investigations of questions on the use of generalized additive models in time-series epidemiological studies have raised model specification issues. There remains a need for further study on the selection of appropriate modeling strategies and appropriate methods to control for time-varying factors, such as temperature, and to better understand the role of copollutants in the ambient air.

(5) Limited controlled human exposure and epidemiology research has provided suggestive evidence of both direct and indirect effects of O_3 on the cardiovascular system, cardiovascular hospital admissions, and cardiovascular mortality. However, additional work will be needed to examine biologically plausible mechanisms of cardiovascular effects and to determine the extent to which O_3 is directly implicated or works together

with other pollutants in causing adverse cardiovascular effects in sensitive individuals and in the general population.

(6) Most epidemiological studies of short-term exposure effects have been time-series studies in large populations. Time-series studies remain subject to uncertainty due to use of ambient fixed-site data serving as a surrogate for ambient exposures, to the difficulty of determining the impact of any single pollutant among the mix of pollutants in the ambient air, to limitations in existing statistical models, or to a combination of all of these factors. Independent variables for air pollution have generally been measurements made at stationary outdoor monitors, but the accuracy with which these measurements actually reflect subjects' exposure is not yet fully understood. Also, additional research is needed to improve the characterization of the degree to which discrepancy between stationary monitor measurements and actual pollutant exposures introduces error into statistical estimates of pollutant effects in time-series studies.

(7) Improved understanding of human exposures to ambient O_3 and to related copollutants is an important research need. Population-based information on human exposure for healthy adults and children and susceptible or at-risk populations including asthmatics to ambient O_3 concentrations, including exposure information in various microenvironments, is needed to better evaluate current and future O_3 exposure models. Such information is needed for sufficient periods to facilitate evaluation of exposure models throughout the O_3 season.

(8) Information is needed to improve inputs to current and future population-based O_3 exposure and health risk assessment models. Collection of time-activity data over longer time periods is needed to reduce uncertainty in the modeled exposure distributions that form an important part of the basis for decisions regarding air quality standard for O_3 and other air pollutants. Research addressing energy expenditure and associated breathing rates in various population groups, particularly healthy and asthmatic children, in various locations, across the spectrum of physical activity, including sleep to vigorous physical exertion is needed.

(9) An important consideration in the O_3 NAAQS review is the characterization of policy relevant background levels. There still remain significant uncertainties in the characterization of 8-hr daily maximum O3 background concentrations. Further research to improve the evaluation of the GEOS-CHEM model which has been used to characterize estimates of policy relevant background levels would help reduce

uncertainties in estimating health risks relevant for standard setting (i.e., those risks associated with exposure to O_3 in excess of policy relevant background levels) and would aid in the development of associated control programs.

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7. POLICY-RELEVANT ASSESSMENT OF WELFARE EFFECTS EVIDENCE

7.1 INTRODUCTION

This chapter presents information critical to the review of the secondary NAAQS for O_3. Welfare effects addressed by a secondary NAAQS 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. Of these welfare effects categories, the effects of O_3 on vegetation, including agricultural crops, trees in managed and unmanaged forests, and herbaceous and woody species growing in natural settings are of most concern at concentrations typically occurring in the U.S. As stated in earlier reviews, "of the phytotoxic compounds commonly found in the ambient air, O_3 is the most prevalent, impairing crop production and injuring native vegetation and ecosystems more than any other air pollutant" (U.S. EPA, 1989, 1996b).

Ozone can also affect other ecosystem components such as soils, water, wildlife, and habitat, either directly, or indirectly, through its effects on vegetation. These individual ecosystem components are associated with one or more of six essential ecological attributes (EEAs) recently described in *A Framework for Assessing and Reporting on Ecological Condition: an SAB report* (Young and Sanzone, 2002) as part of a conceptual framework useful for assessing and reporting on ecological condition (see Figure 7-21 and discussion in section 7.7). This framework can be used to link O_3 effects at the species level to potential impacts at higher levels in the hierarchy (e.g., EEAs). Some of these species level impacts have direct, quantifiable economic value, while others are currently not quantifiable, but still have societal value. In the absence of sufficient research to allow quantification of O_3 impacts at the ecosystem level, including impacts on ecosystem goods and services, only a qualitative discussion is included. However, the staff infers, based on the linkages described in the SAB framework, that increasing protection for vegetation from O_3 related effects would also improve the protection afforded to ecosystems and their related public welfare categories.

Other O_3 related welfare effects categories include damage to certain manmade materials (e.g., elastomers, textile fibers, dyes, paints, and pigments) and climate interactions. The amount of damage to actual in-use materials and the economic consequences of that damage are poorly characterized, however, and the scientific literature contains very little new information to adequately quantify estimates of materials damage from photochemical oxidants (U.S. EPA,

1996a, b, 2006). Therefore, staff judges that there is insufficient information in the materials damage literature to inform secondary standard setting and so it will not be discussed further. Interested readers are referred to Chapter 11 in the CD (U.S. EPA, 2006). In contrast, the welfare impact of O_3 on local, regional and global climates has received more attention in recent years. Ozone enhances the heat capacity of the atmosphere. The overall body of scientific evidence suggests that high concentrations of O_3 on a regional scale could have a discernable influence on climate, leading to surface temperature and hydrological cycle changes. However, the CD stated that confirming this effect will require further advances in monitoring and improvement in chemical transport and regional-scale modeling. Thus, staff concludes that insufficient information is available at this time to quantitatively inform the secondary NAAQS process with regard to this aspect of the O_3-climate interaction and will not address it further. Another aspect, is the potential modification of plant response to O_3 under conditions of changing climate, is included in the discussion of factors that can modify the predicted vegetation responses (see section 7.4.2).

To summarize, this chapter includes an integrated discussion of the key policy relevant science regarding O_3-related effects on vegetation (sections 7.2 through 7.4) and terrestrial ecosystems (section 7.7), as described in the previous CD (U.S. EPA, 1996a) and reiterated in the current CD (U.S. EPA, 2006). The remaining sections (7.5 and 7.6) of this chapter are focused on a discussion of the analyses that have been conducted in support of this current NAAQS review that update and expand upon the exposure, risk and benefits assessments conducted in the last review (U.S. EPA, 1996b). These updated assessments incorporate newer data, models, and approaches, and take into account alternative O_3 air quality scenarios under consideration. The environmental assessment technical support document, *Technical Report on Ozone Exposure, Risk, and Impacts Assessments for Vegetation* (Abt, 2007) (hereafter cited as "Environmental Assessment TSD"), presents a detailed description of the exposure, risk and impacts analysis methodology. Results from these assessments, along with key uncertainties and limitations, are also described in sections 7.5 and 7.6. This information forms the basis for a discussion in Chapter 8 of staff conclusions and recommendations with respect to the secondary O_3 NAAQS.

7.2 MECHANISMS GOVERNING PLANT RESPONSE TO OZONE

The interpretation of predictions of risk associated with vegetation response at ambient O_3 exposure levels can be informed by scientific understanding regarding O_3 impacts at the genetic, physiological, and mechanistic levels. In most cases, the mechanisms of response are similar regardless of the degree of sensitivity of the species. The information assessed in the 1996 CD (U.S. EPA 1996a) regarding the fundamental hypotheses concerning O_3-induced

changes in physiology continues to be valid. However, during the last decade, understanding of the cellular processes within plants has been further clarified and enhanced. Therefore, this section reviews the key scientific conclusions identified in 1996 O_3 CD (U.S. EPA, 1996a), and incorporates new information from the current CD (U.S. EPA, 2006). This section describes: (1) O_3 uptake, (2) cellular to systemic O_3 response, (3) plant compensation and defense mechanisms, (4) O_3–induced changes to plant metabolism, and (5) plant response to chronic O_3 exposures.

7.2.1 Ozone Uptake: Canopy, Plant and Leaf

To cause injury, O_3 must first enter the plant through the stomata of the leaves. Leaves exist in a three dimensional environment called the plant canopy, where each leaf has a unique orientation and receives a different exposure to ambient air, microclimatological conditions, and sunlight. In addition, a plant may be located within a stand of other plants which further modifies ambient air exchange with individual leaves. Not all O_3 entering a plant canopy is absorbed into the leaves, but may be adsorbed to other surfaces e.g., leaf cuticles, stems, and soil (termed non-stomatal deposition) or scavenged by reactions with intra-canopy biogenic VOCs and naturally occurring NOx emissions from soils. Because O_3 does not penetrate the leaf's cuticle, it must reach the stomatal openings in the leaf for absorption to occur. The movement of O_3 and other gases such as CO_2 into and out of leaves is controlled primarily through the stomata. The aperture of the stomata are controlled by guard cells, which respond to a variety of internal species-specific factors as well as external site specific environmental factors such as light, humidity, CO_2 concentration, soil fertility and water status, and in some cases the presence of other air pollutants, including O_3 (see section 7.4.2). These modifying factors produce stomatal conductances that vary across the diurnal cycle, days and seasons. Once O_3 is inside the leaf, a phytotoxic effect will only occur if sufficient amounts of O_3 reach sensitive cellular sites that are subject to the various physiological and biochemical controls within the leaf cells (see the discussion in section 7.2.3 below – Compensation and Detoxification).

A measure of O_3 flux is attractive because it incorporates both relevant environmental factors and physiological processes, and is considered the measure that most closely links exposure to plant response. Unfortunately, measurement of flux is very complex, making it difficult to extrapolate uptake from an individual leaf to that of a whole plant or canopy. Since the last review, interest has been increasing, particularly in Europe, in using mathematically tractable flux models for O_3 assessments at the regional and national scale (Emberson et al., 2000a, b). Though significant new research has been done with respect to flux model development, it has still not advanced to a point of being generally applicable across a range of

species and environments at a national scale. These topics are discussed in more detail in Appendix 7A of this document and in the current CD (U.S. EPA, 2006).

7.2.2 Cellular to Systemic Response

Once O_3 diffuses into the leaf air spaces it can react with a variety of biochemical compounds that are exposed to the air (path 1) or it can be solubilized into the water lining the cell wall of the air spaces (path 2). Having entered the aqueous phase, O_3 can be rapidly altered to form oxidative products that can diffuse more readily into and through the cell and react with many biochemical compounds. The initial sites of membrane reactions seem to involve transport properties and, possibly, the external signal transducer molecules (U.S. EPA, 2006). This alteration in plasma membrane function is clearly an early step in a series of O_3-induced events that leads to leaf injury.

Under certain circumstances, O_3 reacts with organic molecules to generate peroxides, including hydrogen peroxide (H_2O_2). The role of hydrogen peroxide as a signaling molecule in plants is now better understood. The primary set of metabolic reactions that O_3 triggers clearly includes those typical of "wounding" responses generated by cutting of the leaf or by pathogen/insect attack. One aspect of this total response is the production of O_2 and H_2O_2 by the cell (Lamb and Dixon, 1997). The presence of higher-than-normal levels of H_2O_2 within the apoplastic space is a potential trigger for the normal, well-studied pathogen defense pathway.

Ethylene is another compound produced when plants are subjected to biotic or abiotic stressors. Increased ethylene production by plants exposed to O_3 stress was identified as a consistent marker for O_3 exposure decades ago (Tingey et al., 1976). These studies suggested that increased production of stress-ethylene correlated well with the degree of foliar injury that developed within hours or days after O_3 exposure. Thus, one could postulate that O_3 generates a wounding response with the production of ethylene, which would, in turn, generate a change in stomatal conductance and photosynthesis.

7.2.3 Compensation and Detoxification

Ozone injury will not occur if (1) the rate and amount of O_3 uptake is small enough for the plant to detoxify or metabolize O_3 or its metabolites or (2) the plant is able to repair or compensate for the O_3 impacts (Tingey and Taylor, 1982; U.S. EPA, 1996a). Leaves may physically exclude O_3 from sensitive tissues. A few studies have documented a direct stomatal closure or restriction in response to the presence of O_3 ranging from within minutes to hours or days of exposure (Moldau et al., 1990; Dann and Pell, 1989; Weber et al., 1993). However, exclusion of O_3 also restricts the uptake of CO_2, thus limiting photosynthesis and growth.

In addition, plants can also effectively protect tissue against damage by dissipating excess oxidizing power using antioxidants. Since 1996, the role of detoxification in providing a level of

resistance to O_3 has been further investigated. A number of antioxidants, including ascorbate, glutathione peroxidase, and sulfuroxide dimutase, which are highly reactive, can detoxify the chemicals generated by O_3. The pattern of changes in these antioxidant proteins varies greatly among different species and conditions. Most recent reports indicate that ascorbate within the cell wall provides the first significant opportunity for detoxification to occur. The balance between the total O_3 flux and the detoxification process has been defined as the "effective flux" (Dämmgen et al., 1993; Grünhage and Haenel, 1997; Musselman and Massman, 1999).

In spite of the new research, however, it is still not clear as to what extent detoxification protects against O_3 injury. Specifically, data are needed on the potential rates of antioxidant production and on the subcellular location of the antioxidants. Potential rates of antioxidant production are needed to assess whether they are sufficient to detoxify the O_3 as it enters the cell. Data on the subcellular location(s) are needed to assess whether the antioxidants are in the cell wall or plasmalemma locations that permit contact with the O_3 before it has a chance to damage subcellular systems. In addition, generation of these antioxidants in response to O_3-induced stress potentially diverts resources away from other sinks and expends energy. Thus, scientific understanding of the detoxification mechanisms is not yet complete and requires further investigation (U.S. EPA, 2006).

Once O_3 injury has occurred in leaf tissue, some plants are able to repair or compensate for the impacts (Tingey and Taylor, 1982). In general, plants have a variety of compensatory mechanisms for low levels of stress including reallocation of resources, changes in root/shoot ratio, production of new tissue, and/or biochemical shifts, such as increased photosynthetic capacity in new foliage and changes in respiration rates, indicating possible repair or replacement of damaged membranes or enzymes. Since these mechanisms are genetically determined, not all plants have the same complement or degree of tolerance, nor are all stages of a plant's development equally sensitive to O_3. It is not yet known to what degree or how the use of plant resources for repair processes affects the overall carbohydrate budget or subsequent plant response to O_3 or other stresses (U.S. EPA, 1996a, U.S. EPA, 2006).

7.2.4 Changes to Plant Metabolism

Ozone inhibits photosynthesis, the process by which plants produce energy rich compounds (e.g., carbohydrates) in the leaves. This impairment can result from direct impact to chloroplast function and/or O_3-induced stomatal closure resulting in reduced uptake of CO_2. A large body of literature published since 1996 has further elucidated the mechanism of effect of O_3 within the chloroplast. Pell et al. (1997) showed that O_3 exposure results in a loss of Ribulose-1, 5-bisphosphate carboxylase/oxygenase (RuBisCo), the central carboxylating enzyme that plays an important role in the production of carbohydrates. Due to its central importance,

any decrease in RuBisCo may have severe consequences for the plant's productivity. Several recent studies have found that O_3 has a greater effect as leaves age, with the greatest impact of O_3 occurring on the oldest leaves (Fiscus et al., 1997; Reid and Fiscus, 1998; Noormets et al., 2001; Morgan et al., 2004). The loss of RuBisCo and its messenger RNA as a function of increasing O_3 exposure is also linked to an early senescence or a speeding up of normal development leading to senescence. If total plant photosynthesis is sufficiently reduced, the plant will respond by reallocating the remaining carbohydrate at the level of the whole organism (see section 7.3 below) (U.S. EPA, 1996a, 2006).

7.2.5 Plant Response to Chronic/Long-term O_3 Exposures

Many changes that occur with O_3 exposure can be observed within hours, or perhaps days, of the exposure, including those connected with wounding and elicitor-induced changes in gene expression. Other effects due to O_3, however, take longer to occur and tend to become most obvious after long exposures to low-O_3 concentrations. These have been linked to senescence or some other physiological response very closely linked to senescence. The understanding of how O_3 affects long-term growth and resistance to other biotic and abiotic insults in long-lived trees is unclear. Often, the conditions to which a tree is subjected to in one year will affect, or "carry-over", the response of that tree into the next year (U.S. EPA, 2006). In other words, a condition in an earlier year sets the stage for a reaction in the next year; thereby giving a "cause-effect" scenario (U.S. EPA, 2006). In perennial plant species, growth affected by a reduction in carbohydrate storage may result in the limitation of growth the following year (Andersen et al., 1997). Carry-over effects have been documented in the growth of tree seedlings (Hogsett et al., 1989; Sasek et al., 1991; Temple et al., 1993; U.S. EPA, 1996a) and in roots (Andersen et al., 1991; U.S. EPA, 1996a). Accumulation of carry-over effects over time will affect survival and reproduction. Understanding of how O_3 interacts with the plant at a cellular level has dramatically improved in recent years. However, additional work remains to more fully elucidate the translation of those cellular mechanisms into altered cell metabolism, whole plant productivity, and other physiological effects.

7.3 NATURE OF EFFECTS ON VEGETATION

Studies published since the conclusion of the 1996 review continues to support and strengthen key conclusions regarding O_3 effects on vegetation and ecosystems found in the previous CD (U.S. EPA, 1996a) and reiterated in the current CD (U.S. EPA, 2006). For additional detail the reader is referred to Chapter 9 and Annex 9 in the current CD (U.S. EPA, 2006)

7.3.1 Ozone Sensitive Plants and Their Relationship to Public Welfare

Of all the plant species growing within the U.S. (over 43,000 species have been catalogued in the USDA PLANTS database, USDA, NRCS, 2006), only a small percentage have been studied with respect to O_3 sensitivity. Most of the studied species were selected because of their commercial importance or observed O_3-induced visible foliar injury in the field. Given that O_3 impacts to vegetation also include less obvious but often more significant impacts, such as reduced annual growth rates and below ground root loss (see following sections), the paucity of information on other species means the true range of both inter- and intra-species O_3 sensitivity that exists within U.S. vegetation is unknown. Even so, plant species/genotypes with known O_3 sensitivity span a broad range of vegetation types and public use categories. These use categories include food production for human and domestic animal consumption, fiber and materials production, and urban/private landscaping. In addition to these direct uses, a number of O_3 sensitive species have specific relevance to public welfare based on their aesthetic, medicinal, or habitat value. Table 7J-1 in Appendix 7J presents a list of O_3 sensitive species that occur in Federal Class I areas and Table 7J-2 presents O_3 sensitive species in each state.

7.3.2 Vegetation Effects Endpoints

Ozone injury at the cellular level, when it has accumulated sufficiently, will be propagated to the level of the whole leaf or plant. These larger scale effects can include: visible foliar injury and premature senescence; reduced carbohydrate production and reallocation; reduced growth or reproduction; and reduced plant vigor. Much of what is now known about O_3 exposure-plant response relationships, as summarized below, is based on research that was available in the last review. Thus, the present discussion is largely based on the conclusions of the 1978, 1986, and 1996 CDs (U.S. EPA, 1978; 1986; 1996a). Further, research results published since 1996 have supported earlier EPA conclusions (U.S. EPA, 2006) and in some cases have expanded and strengthened those conclusions. The sections below describe the current understanding of the physiological effects of O_3 on vegetation.

7.3.2.1 Visible Foliar Injury and Premature Senescence

Cellular injury can and often does become visible. Acute injury usually appears within 24 hours after exposure to O_3 and, depending on species, can occur under a range of exposures and durations from 0.04 ppm for a period of 4 hours to 0.41 ppm for 0.5 hours for crops and 0.06 ppm for 4 hours to 0.51 ppm for 1 hour for trees and shrubs (Jacobson, 1977). Chronic injury may be mild to severe. In some cases, cell death or premature leaf senescence may occur. The significance of O_3 injury at the leaf and whole plant levels depends on how much of the total leaf area of the plant has been affected, as well as the plant's age, size, developmental stage, and degree of functional redundancy among the existing leaf area. Previous CDs have noted the

difficulty in relating visible foliar injury symptoms to other vegetation effects such as individual tree growth, stand growth, or ecosystem characteristics (U.S. EPA, 1996a) and this difficulty remains to the present day (See discussion in section 7.6.3.2). As a result, it is not presently possible to determine, with consistency across species and environments, what degree of injury at the leaf level has significance to the vigor of the whole plant.

However, visible foliar injury by itself, can impact the public welfare. The presence of visible symptoms due to O_3 exposures can reduce the market value of certain leafy crops (such as spinach, lettuce) and impact the aesthetic value of ornamentals (such as petunia, geranium, and poinsettia) in urban landscapes and scenic vistas in protected natural areas such as national parks and wilderness areas. Though not quantified, there is likely some level of economic impact to businesses and homeowners from O_3-related injury on sensitive ornamental species due to the cost associated with more frequent replacement and/or increased maintenance (fertilizer or pesticide application). In addition, because O_3 not only results in discoloration of leaves but can lead to more rapid senescence (early shedding of leaves), there is a potential for some lost tourist dollars at sites where fall foliage is less available or attractive.

In recent years, field surveys of visible foliar injury symptoms have become more common, with greater attention to the standardization of methods and the use of reliable indicator species (Campbell et al., 2000; Smith et al., 2003). Specifically, the Unites States Forest Service (USFS) through the Forest Health Monitoring Program (FHM) (1990 - 2001) and currently the Forest Inventory and Analysis (FIA) Program collects data regarding the incidence and severity of visible foliar injury on a variety of O_3 sensitive plant species throughout the U.S. (Coulston et al. 2003, 2004; Smith et al. 2003). Section 7.6.3.2 contains additional information on the use of visible foliar injury incidence on bioindicator species as a measure of the occurrence of phytotoxic levels of O_3 in the ambient air.

To view pictures of O_3-induced foliar injury symptoms to selected sensitive species go to the USDA Agricultural Research Service website (http://www.ars.usda.gov/Main/docs.htm?docid=12463) or the USDA Forest Service Ozone Biomonitoring website (http://www.fiaozone.net/species/index.html).

7.3.2.2 Carbohydrate Production and Allocation

When total plant photosynthesis is sufficiently reduced, the plant will respond by reallocating the remaining carbohydrate at the level of the whole organism. Many studies have demonstrated that root growth is more sensitive to O_3 exposure than stem or leaf growth (U.S. EPA, 2006). When fewer carbohydrates are present in the roots, less energy will be available for root-related functions such as acquisition of water and nutrients. Mycorrhizal fungi in the soil form a symbiotic relationship with many terrestrial plants. For host plants, these fungi improve

the uptake of nutrients, protect the roots against pathogens, produce plant growth hormones, and may transport carbohydrates from one plant to another (U.S. EPA, 1996a). Ozone can disrupt the association between mycorrhizal fungi and host plants by inhibiting photosynthesis and the amount of carbohydrates available for transfer to the roots. This effect has recently been documented in the field. Data from a long-studied pollution gradient in the San Bernardino Mountains of southern California suggest that O_3 substantially reduces root growth in natural stands of ponderosa pine (*Pinus ponderosa*). Root growth in mature trees was decreased at least 87% in a high-pollution site as compared to a low-pollution site (Grulke et al., 1998), and a similar pattern was found in a separate study with whole-tree harvest along this gradient (Grulke and Balduman, 1999). Though effects on other ecosystem components were not examined, a reduction of root growth of this magnitude could have significant implications for the below ground communities at those sites. In contrast, a study in Great Smoky Mountains National Park in Tennessee found no statistically significant effects of O_3 exposure on stem or root biomass for several tree species (Neufeld et al., 2000). The difference in the results from these two studies may reflect the species specific nature of the symbiont-host relationship.

Unlike root systems, effects on leaf and needle carbohydrate content under O_3 stress range from a reduction (Barnes et al., 1990; Miller et al., 1989), to no effect (Alscher et al., 1989), to an increase (Luethy-Krause and Landolt, 1990). Therefore, studies that examine only above-ground vegetative components may miss important O_3-induced changes below ground. These below-ground changes could signal a shift in nutrient cycling with significance at the ecosystem level (Young and Sanzone, 2002).

7.3.2.3 Growth and Reproduction

Studies of the growth response of trees to O_3 have established that individual deciduous trees are generally less sensitive to O_3 than most annual plants, with the exception of a few genera such as *Populus*, which are highly sensitive and in some cases (for instance, quaking aspen and black cherry), are as sensitive to O_3 as annual plants. The O_3 sensitivity of seedlings and mature trees within species and between species varies widely. In general, mature deciduous trees are likely to be more sensitive to O_3 compared to seedlings, while mature evergreen trees are likely to be less sensitive than seedlings. Based on these results, stomatal conductance, O_3 uptake, and O_3 effects cannot be assumed to be equivalent in seedlings and mature trees.

Depending on exposure duration, concentrations of O_3 currently in the United States are sufficient to affect the growth of a number of tree species during the annual growing season. However, these conclusions do not take into account the possibility of "carry over" effects on growth in subsequent years, an important consideration in the case of long-lived species. Given that multiple-year exposures may cause a cumulative effect on the growth of some trees (Hogsett

et al. 1989; Simini et al., 1992; Temple et al., 1993), it is likely that a number of species currently are being impacted.

Other research in the U.S. in the last 10 years has focused on perennial forage crops (U.S. EPA, 2006). Recent results confirm that yields and quality of multiple-year forage crops are reduced at sufficient magnitude to have nutritional and possibly economic implications to their use as ruminant animal feed at O_3 exposures that occur in some years over large areas of the U.S. Ozone may also reduce the quality or nutritive value of annual species.

Recent studies have also further demonstrated O_3 effects on different stages of plant reproduction. Effects of O_3 have been observed on pollen germination, pollen tube growth, fertilization, and abortion of reproductive structures, as reviewed by Black et al. (2000). For seed-bearing plants, reproductive effects will culminate in seed production. The recent scientific literature supports the conclusions of the 1996 CD that ambient O_3 concentrations are reducing the yield of major crops in the U.S. For example, the yield reductions for soybean are generally similar to those reported previously (U.S. EPA, 2006).

7.3.2.4 Reduced Plant Vigor

Though O_3 levels over most of the U.S. are not high enough to kill vegetation directly, current levels have been shown to reduce the ability of many sensitive species and genotypes within species to adapt to or withstand other environmental stresses. These may include increased susceptibility to freezing temperatures, pest infestations and/or root disease, and compromised ability to compete for available resources. For example, when different species are grown together, O_3 exposure can increase the growth of O_3-tolerant species while exacerbating the growth decrease of O_3-sensitive species. In the long run, the result of this loss in vigor may be plant death.

7.4 IMPACTS ON PUBLIC WELFARE

7.4.1 What Constitutes an Adverse Vegetation Impact from Ozone Exposure?

Ozone can cause a variety of effects, beginning at the level of the individual cell and accumulating up to the level of whole leaves, plants, plant populations, communities and whole ecosystems. Not all O_3-related effects, however, have been classified as "adverse" to public welfare. Previous reviews have classified O_3 vegetation effects as either "injury" or "damage" to help in determining adversity. Specifically, "injury" is defined as encompassing all plant reactions, such as reversible changes in plant metabolism (e.g., altered photosynthetic rate), altered plant quality, or reduced growth, that does not impair the intended use or value of the plant (Guderian, 1977). In contrast, "damage" includes those injury effects that also reduce or impair the intended use or value of the plant. Damage includes reductions in aesthetic values

(e.g., visible foliar injury in ornamental species) as well as losses in terms of weight, number, or size of the plant part that is harvested (yield loss). Yield loss also may include changes in crop quality, i.e., physical appearance, chemical composition, or the ability to withstand storage. While this construct has proved useful in the past, it appears most useful in the context of evaluating effects on single plants or species grown in monocultures, such as agricultural crops or managed forests. It is less clear how it might apply to potential effects on natural forests or entire ecosystems, such as shifts in species composition or nutrient cycling where the intended use or value of the system is not specifically quantified.

A more recent construct for assessing risks to forests described in Hogsett et al. (1997) suggests that "adverse effects could be classified into one or more of the following categories: (1) economic production, (2) ecological structure, (3) genetic resources, and (4) cultural values." This expands the context for evaluating the adversity of O_3-related effects beyond the species level. In another recent publication, *A Framework for Assessing and Reporting on Ecological Condition: an SAB report* (Young and Sanzone, 2002), additional support is provided for expanding the consideration of adversity by making explicit the linkages between stress- related effects (e.g., O_3 exposure) at the species level and at higher levels within an ecosystem hierarchy. Staff suggests that consideration of adverse effects undertaken within the context of such a broader paradigm is appropriate in the context of this secondary NAAQS review.

7.4.2 Factors That Modify Functional and Growth Response

The caveat that must be placed on results from any experimental study on the response of living organisms to a stressor in a specific setting is that uncertainty is introduced when attempting to extrapolate or apply those results outside that specific setting (e.g., to a different set of organisms, scales, or exposure/growing conditions). The description of plant response to O_3 exposure is no different. Because staff must necessarily rely on experimental data produced under very specific sets of conditions in conducting this assessment, it is important to understand the range of factors that can influence plant response to O_3 and the magnitude and direction of that response, in order to better assess the likelihood of observing the experimentally predicted response in the ambient environment.

Plant response to O_3 exposure is a function of the plant's ongoing integration of genetic, biological, physical and chemical factors both within and external to the plant. The corollary is also true that O_3 exposure can modify the plant's subsequent integrated response to other environmental factors, both by influencing the plant response directly, and by contributing to altered climatic factors that influence plant response through its greenhouse gas forcing properties.

The 1996 O_3 CD (U.S. EPA, 1996a) concluded with a statement that our understanding regarding modifying factors was too fragmented to permit drawing many general conclusions. Unfortunately, in the interval since the 1996 CD, little additional information has become available, thus, this earlier conclusion remains unchanged. Therefore, only a brief overview of the current understanding from this research is provided. The reader is referred to the 1996 CD (U.S. EPA 1996a) and the current 2006 CD (U.S. EPA 2006) for further information.

7.4.2.1 Genetics

Plant response to O_3 is determined by genes that are directly related to oxidant stress and to an unknown number of genes that are not specifically related to oxidants but instead control leaf and cell wall thickness, stomatal conductance, and the internal architecture of the air spaces. It is unlikely that single genes are responsible for O_3 tolerance, except in rare cases (Engle and Gabelman, 1966). Recent studies using molecular biological tools and transgenic plants have begun to positively verify the role of various genes and gene products in O_3 tolerance and are beginning to increase the understanding of O_3 toxicity and differences in O_3 sensitivity. Specifically, O_3 has been shown to trigger the production of a number of compounds (e.g., ethylene) and the signaling of these molecules determines, in some cases, the O_3 susceptibility of plants (U.S. EPA, 2006). Because the genetic code is species specific, species vary greatly in their responsiveness to O_3. Even within a given species, individual genotypes or populations can also vary significantly with respect to O_3 sensitivity. Thus, caution should be taken when ranking species categorically as having an absolute degree of sensitivity to O_3.

7.4.2.2 Biological Factors

The biological factors within the plant's environment that may directly or indirectly influence its response to O_3 in a positive or negative manner encompass insects, other animal pests, diseases, weeds, and other competing plant species. Ozone and other photochemical oxidants may influence the severity of a disease or infestation either by direct effects on the causal species, or indirectly by affecting the host, or both. Likewise, mutually beneficial relationships or symbioses involving higher plants and bacteria or fungi may also be affected by O_3. Ozone can also have indirect effects on herbivorous animals due to O_3-induced changes in feed quality.

New evidence with regard to insect pests and diseases has done little to remove the uncertainties noted in the 1996 CD (U.S. EPA 1996a). Most of the large numbers of such interactions that may affect crops, forest trees, and other natural vegetation have yet to be studied. With respect to any particular O_3-plant-insect interaction, it is still not possible to predict its likelihood, or its severity. The situation is only a little clearer with respect to interactions involving facultative necrotrophic plant pathogens, with O_3 generally leading to increased

disease. In contrast, with obligate biotrophic fungal, bacterial, and nematode diseases, there are twice as many reports indicating O_3-induced inhibitions than enhancements. At this time, therefore, although some diseases may become more widespread or severe as a result of exposure to O_3, it is still not possible to predict which diseases are likely to present the greatest risks to crops and forests.

The latest studies on O_3 interactions with root symbionts present a more complex picture than was described in the last review. In addition to adverse effects of O_3 on the functioning of tree root symbioses with mycorrhizae (discussed in section 7.3.1 above), there is also evidence that the presence of mycorrhizae may help plants overcome root diseases stimulated by O_3 and/or encourage the spread of mycorrhizae to the roots of uninfected trees.

The few recent studies of the impact of O_3 on intraspecific plant competition have again confirmed that grasses frequently show greater resilience than other types of plants. In grass-legume pastures, the leguminous species suffer greater growth inhibition. Separately, the suppression of ponderosa pine seedling growth by blue wild-rye grass was markedly increased by O_3 (Andersen et al. 2001). Due to the limited number of species studied under competitive situations to date, however, it is still not possible to predict the outcome of O_3 exposure on other specific competitive situations, such as successional plant communities or crop-weed interactions. Clearly, however, O_3 stress creates a selective pressure in some vegetative communities that can lead to a shift in community composition. This community change may be undesirable in some settings.

7.4.2.3 Physical Factors

The interaction of a plant with its physical environment (e.g., light, temperature, relative humidity, soil moisture and wind speed/turbulence) influences the degree and/or nature of the plant response to O_3 exposure. Light is an essential "resource" whose energy content drives photosynthesis and CO_2 assimilation. It has been suggested that increased light intensity may increase the sensitivity of light-tolerant species to O_3 while decreasing the O_3 sensitivity of shade-tolerant species, but this appears to be an oversimplification with many exceptions.

Temperature affects the rates of all physiological processes based on enzyme-catalysis and diffusion, and each process and overall growth (the integral of all processes) has a distinct optimal temperature range. Although some recent field studies have indicated that O_3 impact significantly increases with increased ambient temperature, other studies have revealed little effect of temperature. Temperature is unquestionably an important variable affecting plant response to O_3 in the presence of the elevated CO_2 levels contributing to global climate change (see below). In contrast, evidence continues to accumulate to indicate that exposure to O_3 sensitizes plants to low temperature stress by reducing below-ground carbohydrate reserves,

possibly leading to responses in perennial species ranging from rapid demise to impaired growth in subsequent seasons.

High relative humidity of the ambient air has generally been found to increase the adverse effects of O_3 by increasing stomatal conductance and thereby increasing O_3 flux. Similarly, abundant evidence indicates that the ready availability of soil moisture results in greater sensitivity to O_3. The opposite condition, drought, has been observed in field experiments and modeled in computer simulations to provide partial "protection" against the adverse effects of O_3 as would be expected. This is because, in the short-term, drought causes stomates to close and thus, decrease the exposure to O_3. However, there is also compelling evidence that O_3 can predispose plants to drought stress. Hence, the response will depend to some extent upon the sequence in which the stresses occur, and the species-specific nature of the response. Regardless of the interaction, however, the net result of drought on growth in the short-term is negative, although in the case of tree species, other responses such as increased water use efficiency could be a benefit to long-term survival.

Wind speed and air turbulence affect the thickness of the boundary layers over leaves and canopies and, hence, affects gas exchange rates. These factors can have a significant impact on the relationship between ambient air exposures and actual exposure concentrations at the leaf or canopy surface.

7.4.2.4 Chemical Factors

Mineral nutrients in the soil, other gaseous air pollutants, and agricultural chemicals constitute chemical factors in the environment. The evidence regarding interactions with specific nutrients is still too contradictory to permit any sweeping conclusions. Somewhat analogous with temperature, it appears that any shift away from the nutritional optimum may lead to greater sensitivity, but the shift would have to be substantial before a significant effect on response to O_3 was observed.

Interactions of O_3 with other air pollutants have received relatively little recent attention. The situation with SO_2 remains inconsistent, but seems unlikely to pose any additional risk to those related to the individual pollutants. With NO and NO_2, the situation is complicated by their nutritional value as N sources. In leguminous species, it appears that NO_2 may reduce the impact of O_3 on growth, with the reverse in other species, but the nature of the exposure pattern, i.e., sequential or concurrent, also determines the outcome. Much more investigation is needed before it will be possible will be able to predict the outcomes of different O_3-NO-NO_2 exposure scenarios. The latest research into O_3 × acid rain interactions has confirmed that, at realistic acidities, significant interactions are unlikely. A continuing lack of information precludes offering any generalizations about interactive effects of O_3 with NH_3, HF, or heavy metals.

More evidence has been reported that the application of fungicides affords some protective effects against O_3.

Over the last decade, considerable emphasis has been placed on research into O_3 interactions with two components of global climate change: increased atmospheric CO_2 and increased mean global temperature. Most of these studies, however, have tended to regard increased CO_2 levels and increased mean temperatures as unrelated phenomena, in spite of the crucial role of temperature as a climatic determinant (Monteith and Elston, 1993). Thus, experiments that examine the effects of doubled CO_2 levels at the current mean ambient temperatures are not particularly helpful in trying to assess the impact of climate change on responses to O_3, since most of the biotic and chemical interactions with oxidants may be modified by these climatic changes. Though it is now known from limited experimental evidence and evidence obtained by computer simulation that an atmosphere sufficiently enriched with CO_2 (e.g., 600 + ppm) would more than offset the impact of O_3 on responses as varied as wheat yield or the growth of young Ponderosa pine trees, the concurrent increase in temperature would reduce, but probably not eliminate, the net gain.

Little, if any, experimental evidence exists related to three-way interactions, such as $O_3 \times CO_2 \times$ disease or $O_3 \times CO_2 \times$ nutrient availability. Increased use of computer simulations may be important in suggesting outcomes of the many complex interactions of O_3 and various combinations of environmental factors. However, the results obtained will only be as reliable as the input data used for parameterization. Thus, additional data from organized, systematic studies are needed.

It is important to recognize that wide variations in net impacts of climate change in different geographic areas are expected. Many regions are predicted to experience severe, possibly irreversible, adverse effects due to climate change. The EPA is currently leading a research effort that uses regional-scale climate models with the goal of identifying changes to O_3 and PM concentrations that may occur in a warming climate. An assessment of the results of this effort is expected to be available for consideration in the next review of the O_3 NAAQS.

7.5 CHARACTERIZATION OF VEGETATION EXPOSURES TO OZONE

7.5.1 Key Considerations in Vegetation Exposure Characterization

In the last review, the Administrator chose to make the secondary O_3 NAAQS equal to the primary standard set as the three year average of the annual 4th-highest daily maximum 8-hr average concentrations at the level of 0.08 ppm. While recognizing this as a reasonable policy choice, she also recognized that "a SUM06 seasonal standard is more biologically relevant and, therefore, … also appropriate to consider" (62 FR 38877). This conclusion by the Administrator in 1997 is again supported by the recent body of science reviewed in the 2006 O_3 CD (U.S. EPA,

2006). Staff, therefore, continues to express hourly O_3 monitoring data in terms of both the 8-hr average and seasonal cumulative index forms for comparison. Staff considers the cumulative, concentration weighted SUM06 and W126 index forms discussed in the 1996 Staff Paper (U.S. EPA, 1996b). Staff rationale for including the W126 will emerge from the discussions of current patterns of O_3 air quality and of policy-relevant background (PRB) in the remainder of this section. Further, in a letter to the Administrator on October 24, 2006, CASAC indicated a preference for the W126 form (Henderson, 2006c). Below are the definitions of the three index forms considered in this review and how they will be referred to in the rest of this document:

- 8-hr average form: 4^{th}-highest daily maximum 8-hr average over the O_3 season.

- 12-hr SUM06: 3-month sum of all hourly O_3 concentrations greater than or equal to 0.06 ppm observed during the daily 12-hr period between 8 am and 8 pm. The 3 months are the maximum consecutive 3 months during the O_3 season.

- 12-hr W126: Sigmoidally weighted 3-month sum of all hourly O_3 concentrations observed during the daily 12-hr period between 8 am to 8 pm. The 3 months are the maximum consecutive 3 months during the O_3 season.

  More specifically, W126 is defined in Lefohn et al., 1988 as:

$$W126 = \sum_{i=8AM}^{i<8PM} w_{C_i} C_i, \text{where } C_i = \text{hourly } O_3 \text{ at hour } i, \text{and } w_{C_i} = \frac{1}{1 + 4403e^{-126C_t}}$$

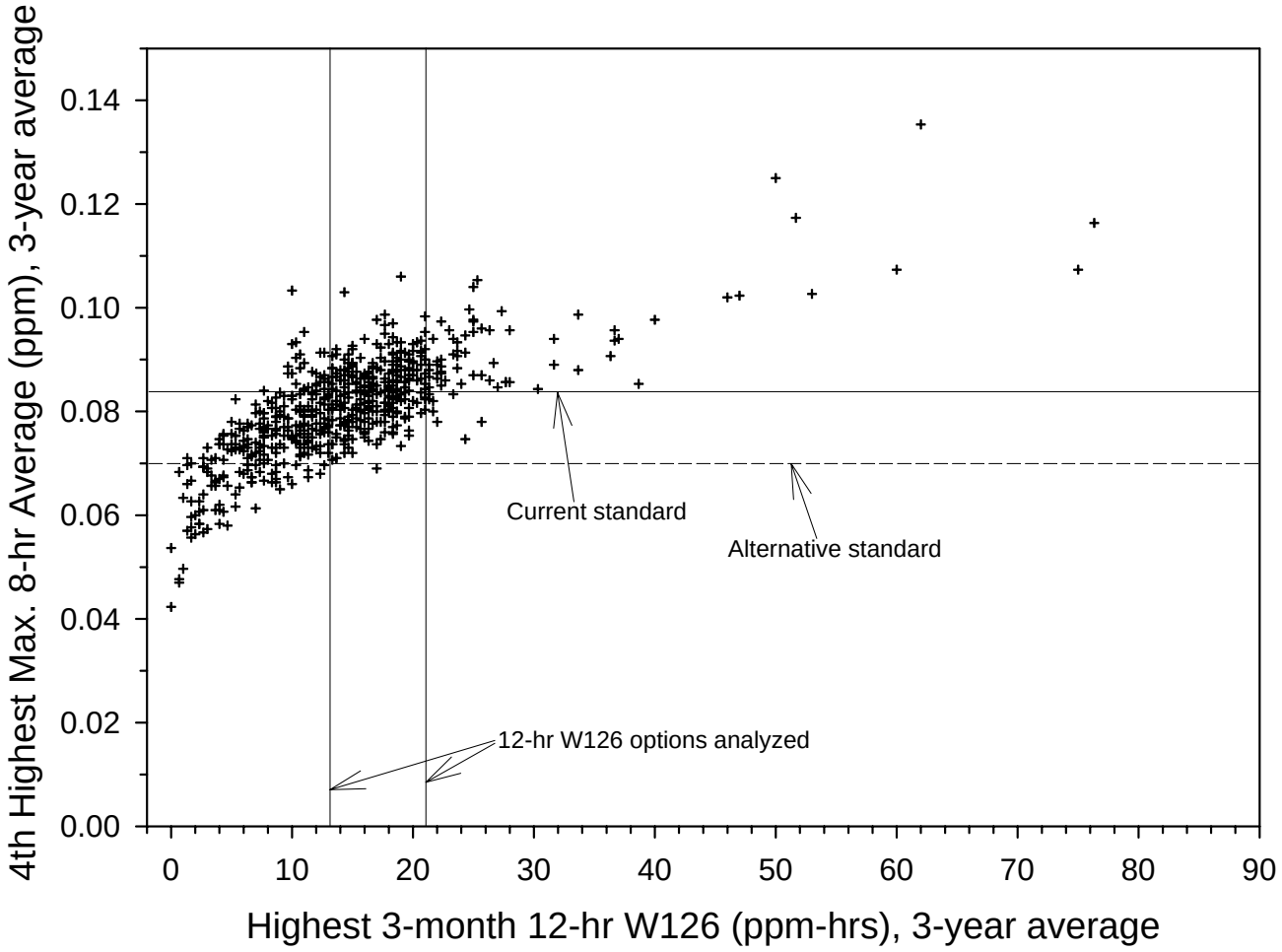
Staff selected two levels of air quality to evaluate for each of these alternative standard forms. Specifically, the levels analyzed were 0.084 and 0.070 ppm, 25 and 15 ppm-hr, and 21 and 13 ppm-hr levels for the 8-hr average, the 12-hr SUM06, and the 12-hr W126 forms, respectively. The current level of the 8-hr average form is 0.084 ppm and the 0.070 ppm level was chosen as a possible alternative. For both the 12-hr SUM06 and 12-hr W126 forms, the two levels were selected on the basis of the associated levels of crop yield loss protection described in the last review. Specifically, both the upper levels of 12-hr SUM06 (25 ppm-hr) and 12-hr W126 (21 ppm-hr) were associated with a level of crop protection of approximately no more than 10% yield loss in 50% of crop cases studied in the National Crop Loss Assessment Network (NCLAN) experiments (section 7.6.2.2.). This level was proposed for a secondary standard in the 1996 review. Alternatively, the lower levels of both SUM06 (15 ppm-hr) and W126 (13 ppm-hr) were associated with a level of crop protection of approximately no more than 10% yield loss in 75% of NCLAN cases. Another level to note is the upper level benchmark of W126

of 31 ppm-hr that approximates the upper end of the SUM06 range analyzed in the last review (U.S. EPA, 1996b) and which was associated with up to 17% yield loss in 50% of crop cases. All approximate equivalency calculations between the 12-hr W126 and 12-hr SUM06 metrics discussed in Chapter 7 and Chapter 8 of this document were done based on NCLAN data (See Appendix 7B).

Since the conclusion of the last review, improvements in O_3 air quality have occurred in some areas of the U.S. In the eastern U.S., these improvements may be attributable in part to the reductions in NOx emissions resulting from the initiation of Phase II of the acid rain program (U.S. EPA, 2004) and the NOx SIP call in 2002 (see Chapter 2 of this SP). In addition, efforts to attain the current NAAQS have no doubt contributed to some air quality improvements, including lower hourly maximum values and fewer occurrences of those maximum values at some sites. One example of this is at the Crestline site in California, where the number of days with concentrations $\geq$ 95 ppb has been declining steadily over the last decade, matched by a decline in peak 1-hr concentrations and 12-hr SUM06 values. These declines match a similar trend in NOx and reactive organic gases (U.S. EPA, 2006, section Annex 9-207, Figure AX9-17) (U.S. EPA 2006; Lee et al 2003). However, not all areas in the U.S. show this declining trend.

The 1997 final rule recognized that "it remained uncertain as to the extent to which air quality improvements designed to reduce 8-hr O_3 concentrations would reduce O_3 exposures measured by a seasonal SUM06 index" (62 FR 38876). In the last review, staff undertook an analysis to explore the relationship between the 8-hr average form and the seasonal SUM06 index. Results of that analysis suggested that many areas that were above the proposed SUM06 standard of 25 ppm-hr were also above the 0.08 primary standard. However, considerable uncertainty remained as to the strength of the relationship, especially between urban O_3 air quality and distributions that occur in non-monitored rural or remote areas. Using recent (2001-2004) county-level air quality data, staff has performed a similar analysis to compare the degree of overlap between the current level of the 8-hr average form and exposure levels of concern for vegetation expressed in terms of the 12-hr W126. Figure 7-1 depicts county O_3 air quality in terms of both the current secondary standard 8-hr average form (Y-axis) and the 3-month 12-hr W126 form (X-axis). This graph shows the relationship between these two forms averaged over three recent years (2002-2004). Both the 21 and 13 ppm-hr levels for the 12-hr W126 are indicated on the graph. For the 3-year average of 2002-2004, only a few counties would have a 12-hr W126 higher than a level of 21 ppm-hr while meeting the 0.08 level of the current 8-hr average form. At the lower 12-hr W126 level of 13 ppm-hr, approximately 135 counties would have a 12-hr W126 higher than a level of 13 ppm-hrs while meeting the 0.08 level of the current 8-hr average form. Based on this comparison, air quality levels associated with adverse

Figure 7-1. The 3-year average (2002-2004) of the 4th-highest maximum 8-hr average (current standard form) versus the 3-year average of the highest 3-month 12-hr W126, by county.



vegetation response can occur in some areas that meet the current 8-hr average secondary NAAQS. The number of counties meeting the current 8-hr average standard and above a 12-hr W126 standard would obviously depend greatly on the level of 12-hr W126 and 8-hr average form selected. In addition, the number of counties also varies depending on the air quality of a particular year or set of years. Figures of the relationship between the current 8-hr average form and 12-hr W126 for 2002 and 2004 are presented in Appendix 7B. These figures demonstrate that the relationship between the current 8-hr average form and 12-hr W126 is not constant and can vary between years. Thus, staff suggests caution should be used in evaluating the likely vegetation impacts associated with a given level of air quality expressed in terms of the 8-hr average form in the absence of parallel 12-hr SUM06 or W126 information. Unfortunately, much of the data published both in this review and in other Agency reports only depicts trend information in terms of the 8-hr average form.

National parks represent areas of nationally recognized ecological and public welfare significance, which are afforded a higher level of protection. Therefore, staff has also focused on air quality in the subset of national park sites and important natural areas. Two recent reports presented some discussion of O_3 trends in a subset of national parks (See discussion in The Ozone Report: Measuring Progress through 2003 (U.S. EPA, 2004) and 2005 Annual Performance and Progress Report: Air Quality in national parks (NPS, 2005). Unfortunately, much of this information is presented only in terms of the current 8-hr average form. Therefore, staff has selected a subset of national parks and other significant natural areas representing 4 general regions of the U.S to analyze available air quality data in terms of the 12-hr W126 levels from 2001 to 2005 (Figures 7-2 and 7-3). These graphs show that many national parks and natural areas have monitored O_3 levels above concentrations that have been shown to decrease plant growth and above the 12-hr W126 levels analyzed in this review. For example, one park in the east and four parks in the west had more than one year with a 12-hr W126 above 21 ppm-hr. This level of exposure has been estimated to cause a 9% biomass loss in 50% of the 49 tree seedling cases studied (Lee and Hogsett, 1996). Sensitive tree seedling species such as black cherry (*Prunus serotina*) and ponderosa pine (*Pinus ponderosa*) have been reported to have 10% biomass losses at levels as low as 5 and 11 ppm-hr (Lee and Hogsett, 1996). Impacts on seedlings may potentially affect long-term growth and survival, ultimately affecting the competitiveness of sensitive species and individual trees.

Another key aspect of evaluating exposure levels of concern to vegetation is distinguishing between pollution levels that can be controlled by U.S. regulations (or through international agreements with neighboring countries) from levels that are generally considered uncontrollable by the U.S., e.g., policy-relevant-background (PRB). As described in Chapter 2 of this SP, the global photochemical transport model GEOS-CHEM (Fiore et al., 2003) was used

Figure 7-2. Highest 3-month 12-hr W126 values from monitors in National Parks and other natural areas in the Southeast (A) and Northeast (B). Monitors designated as GSMNP are found in different areas of the Great Smoky Mountain National Park.

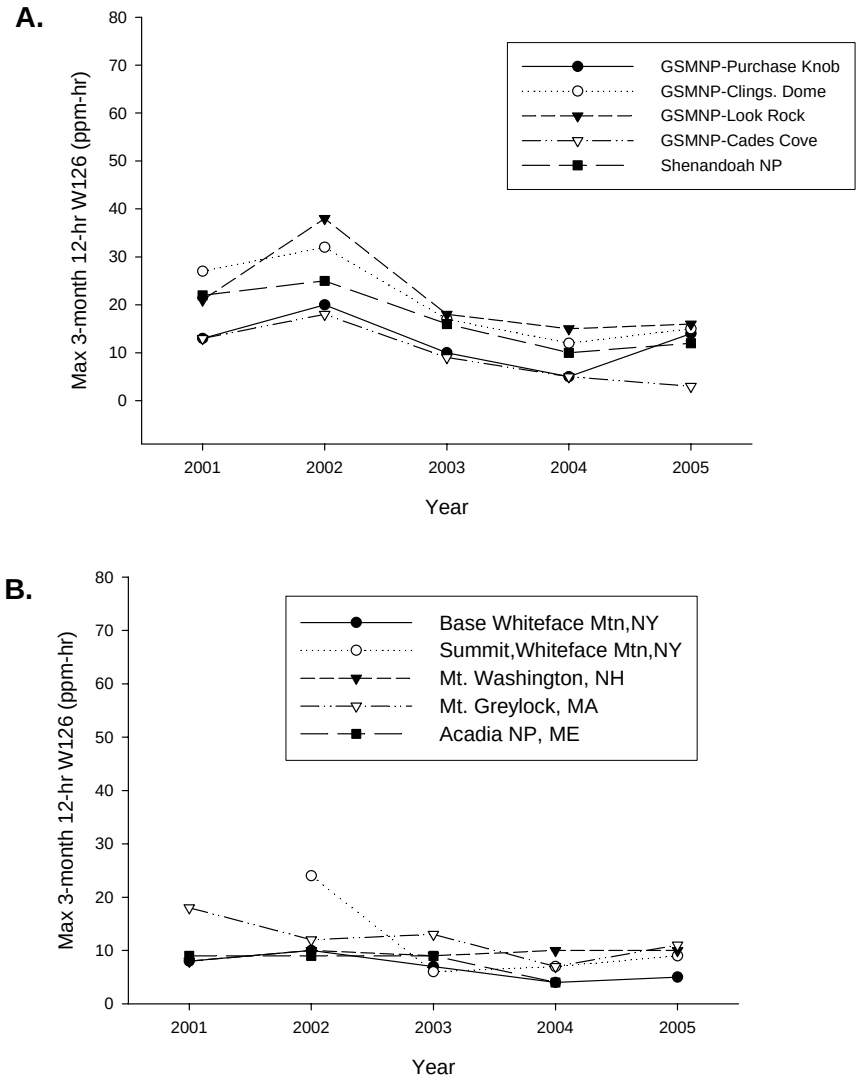
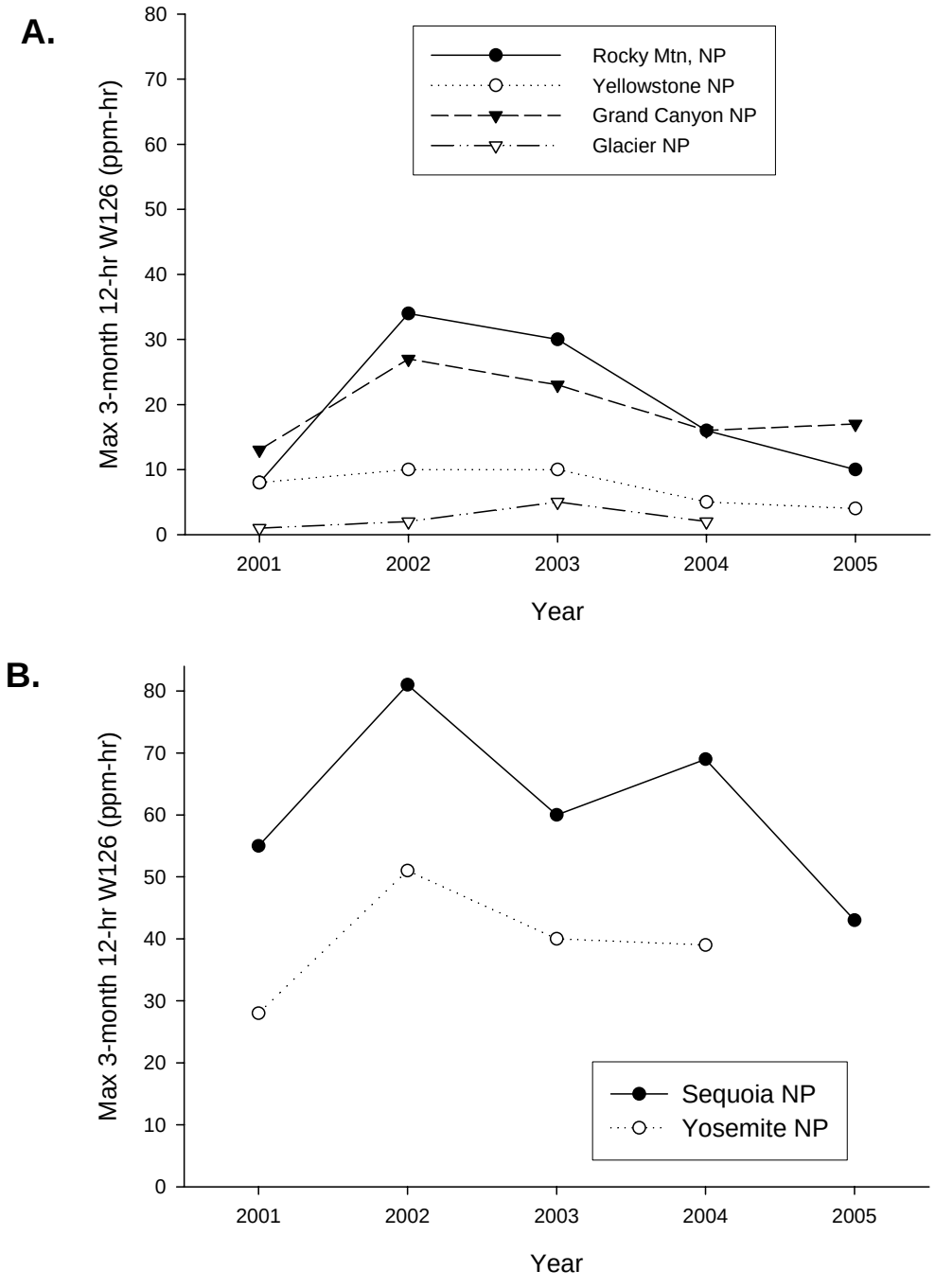


Figure 7-3. Highest 3 month 12-hr W126 values from monitors in National Parks in the Mountain West (A) and California (B).

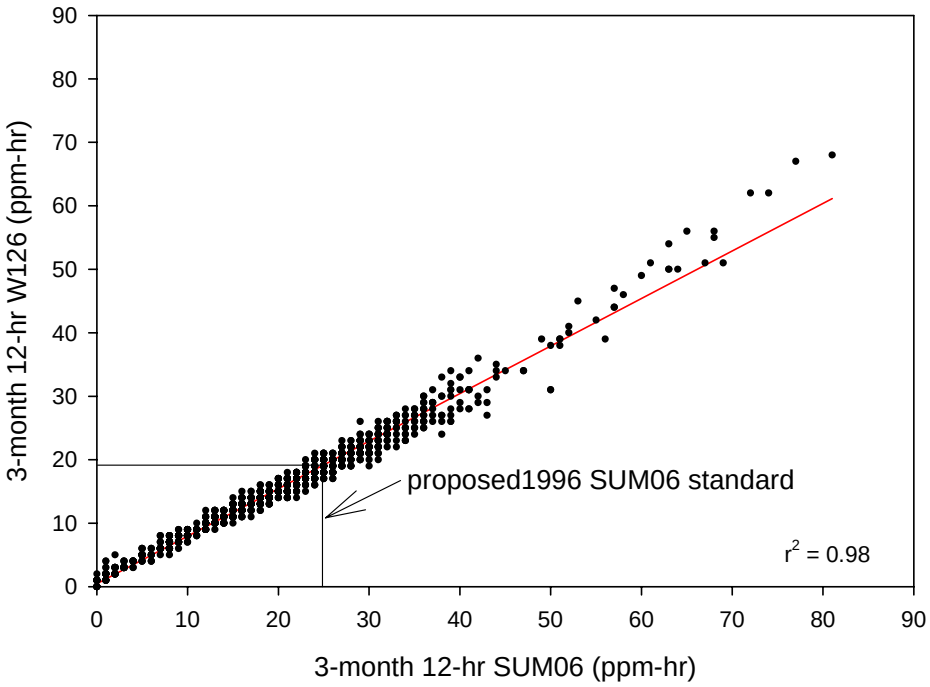


to estimate PRB levels. This model shows that PRB O_3 concentrations, which vary as a function of season, altitude and total surface O_3 concentration, are generally predicted to be in the range of 0.015 to 0.035 ppm at the surface in the afternoon, and they decline under conditions conducive to O_3 episodes. They are highest during spring and decline into summer. Higher values tend to occur at higher elevations during spring due to contributions from hemispheric pollution and stratospheric intrusions. The stratospheric contribution to surface O_3 is typically well below 0.020 ppm and only rarely elevates O_3 concentrations at low-altitude sites and only slightly more often elevates them at high-altitude sites (U.S. EPA, 2006, AX3-148).

The modeled range of 0.015 to 0.035 ppm in the 2006 CD is lower than the 0.03 to 0.05 ppm range used as background O_3 in the 1996 O_3 NAAQS review (U.S. EPA, 1996a, 2006). This is significant for the secondary standard review because the higher end of the range (0.05 ppm) provided an important policy consideration for staff in 1996 for selecting the cumulative SUM06 exposure index that did not weight concentrations below 0.06 ppm. Thus, SUM06 was not influenced by concentrations thought to be at background.

Partially on the basis of these lower estimates of PRB, as well as declining peak O_3 levels at some sites, staff has re-evaluated the usefulness of using the sigmoidally weighted W126 index to capture more of the vegetation relevant exposures below 0.06 ppm. Though the W126 index weights all concentrations, the concentrations below 0.04 ppm receive substantially smaller weights (3 percent or less) so as not to contribute significantly to the value of the index (Lefohn et al. 1988). Indeed, a constant concentration of the highest estimated PRB (0.035 ppm) would only add up to a 3-month 12-hr W126 of less than 1 ppm-hr. In addition, because the W126 form does not contain an absolute threshold like the SUM06 form, it is more in keeping with scientific consensus that there is no threshold for exposures that cause effects on vegetation (Heck and Cowling 1997, U.S. EPA 2006). Further, CASAC has indicated a preference for the 12-hr W126 metric (Henderson, 2006c). Therefore, staff has continued to include the 12-hr W126 in the vegetation risk analyses. Figure 7-4 shows the relationship between 12-hr W126 and SUM06 as measured at O_3 monitors in 2001. The metrics, as calculated at the monitors, are highly correlated. A similar correlation was seen with other years (2002-2004). Because the inflection point of W126 is approximately 0.06 ppm, the SUM06 metric is essentially a simple approximation of the sigmoidally weighted W126 form and it is not surprising that the two metrics measure O_3 exposures in a very similar way at most monitoring stations (Lee et al. 1989). Finally, the W126 metric should also be easier to model than SUM06 since small errors in prediction of hourly concentrations around 0.06 ppm could cause variations in the SUM06 metric. This issue is avoided in the continuous weighting of the W126.

Figure 7-4. Maximum 3-month 12-hr SUM06 plotted against maximum 3-month 12-hr W126. Data points are from the AQS and CASTNET O_3 monitors for the year 2001.

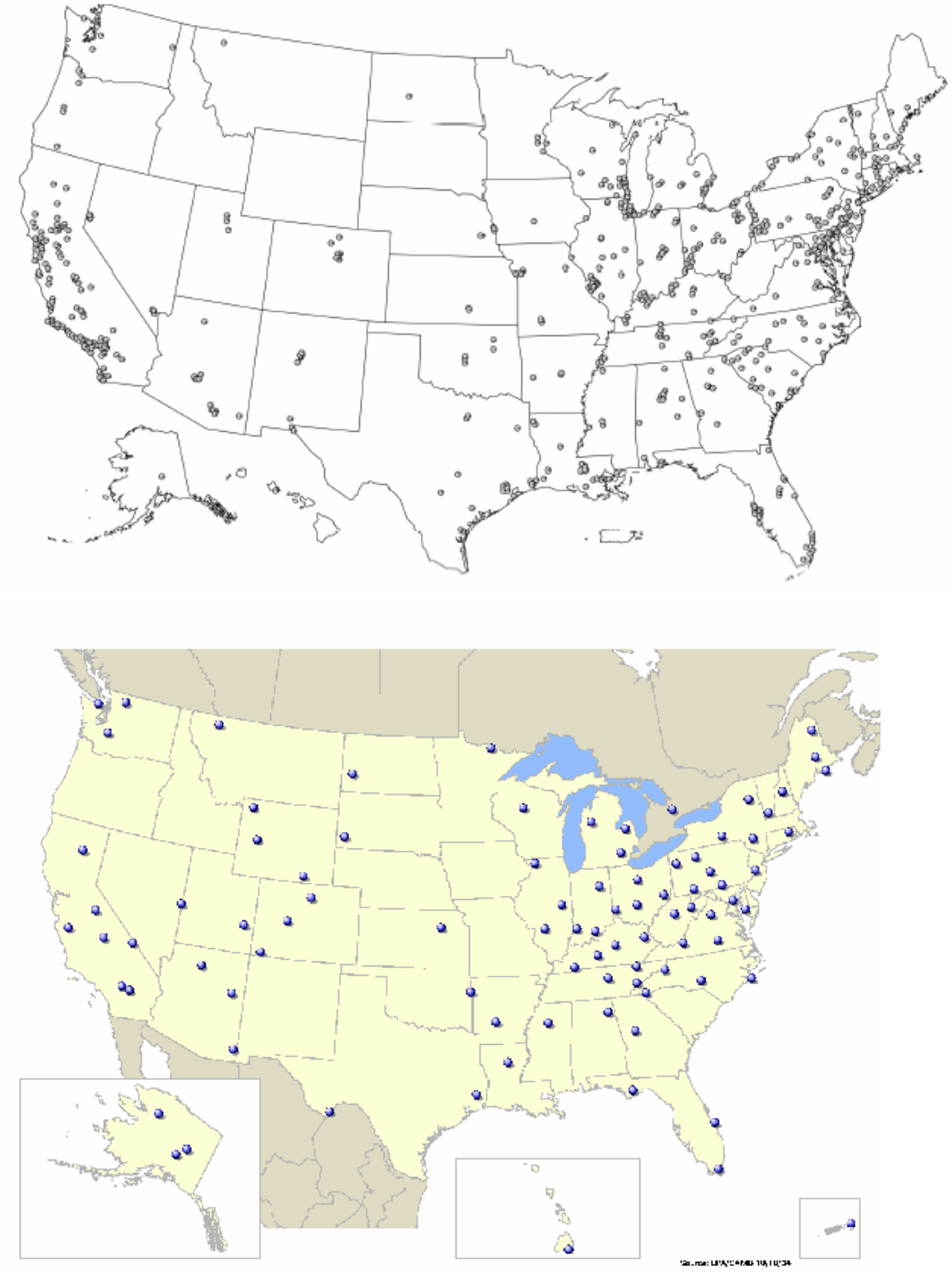


7.5.2 Monitor Networks: National Coverage

Hourly O_3 monitoring data are available from two national networks: (1) Air Quality System (AQS; http://www.epa.gov/ttn/airs/airsaqs) and (2) Clean Air Status and Trends Network (CASTNET; http://www.epa.gov/castnet/). The locations of these monitors are presented in Figure 7-5 and are described in sections 2.3.1 and 2.3.2 of this document. The AQS monitoring network currently has over 1100 active O_3 monitors which are generally sited near population centers. However, this network also includes approximately 36 monitors located in national parks. CASTNET is the nation's primary source for data on dry acidic deposition and rural, ground-level O_3. It consists of over 80 sites across the eastern and western U.S. and is cooperatively operated and funded with the National Park Service. Due to the overall stability in these monitoring networks and standardized, rigorous QA/QC and data handling protocols, they provide useful information regarding long-term trends in air quality across regions and at specific sites. For more on the AQS protocols, see section 2.3.1 of this Staff Paper or Code of Federal Regulations, Title 40, Part 58 (40 CFR Part 58). CASTNET, in terms of data quality, achieved 98% to 99% of all precision and accuracy audits being within the ±10% criteria for both precision and accuracy. Overall, CASTNET O_3 monitors are stable and show only very small variation (U.S. EPA 2003b, p.22). Both networks take O_3 measurements on an hourly time step which allows for quick comparisons between different air quality index forms and different averaging times.

In spite of the size and quality of these monitoring networks, however, vast rural areas of the U.S., where important crops and natural vegetation occur, still do not have O_3 monitor coverage (Figure 7-5). As was the case in the 1996 review, staff found it necessary to select a method that could be used to characterize O_3 air quality over broad geographical areas of concern (see sections 7.5.3 and 7.5.4 below) to support a national scale risk assessment of the effects of ambient O_3 exposures on vegetation and ecosystems. Staff's review of the monitoring data showed that within the five most recent years available (2000 to 2004), 2001 was a fairly moderate O_3 year. Based on this information, and because it coincided with the most recently available air quality model data (see section 7.5.3. below), 2001 was selected as the initial (base) air quality year for most of the quantitative vegetation risk analyses conducted in this review. In a few cases (e.g., visible foliar injury and tree growth modeling), monitoring data from other air quality years were used.

Figure 7-5. Locations of AQS monitors (top) and CASTNET monitoring stations (bottom)



7.5.3 Community Multi-scale Air Quality Model (CMAQ)

Staff investigated the appropriateness of using the O_3 outputs from the EPA/NOAA Community Multi-scale Air Quality (CMAQ) model system (http://www.epa.gov/asmdnerl/CMAQ, Byun and Ching, 1999; Arnold et al. 2003, Eder and Yu, 2005) to improve spatial interpolations based on the regionally limited and unevenly distributed O_3 monitoring network in the western U.S. (see section 7.5.2). The CMAQ model is a multi-pollutant, multiscale air quality model that contains state-of-the-science techniques for simulating all atmospheric and land processes that affect the transport, transformation, and deposition of atmospheric pollutants and/or their precursors on both regional and urban scales. It is designed as a science-based modeling tool for handling many major pollutants (including photochemical oxidants/O_3, particulate matter, and nutrient deposition) holistically. The CMAQ model can generate estimates of hourly O_3 concentrations for the contiguous U.S., making it possible to express model outputs in terms of a variety of exposure indices (e.g., W126, 8-hr average). Due to the significant resources required to run CMAQ, however, model outputs are only available for a limited number of years. For this review, 2001 outputs from CMAQ version 4.5 were the most recent available. This version of CMAQ utilizes the more refined 12 km x 12 km grid for the eastern U.S., while using the 36 km x 36 km grid for the western U.S. The 12 km x 12 km domain covers an area from roughly central Texas, north to North Dakota, east to Maine, and south to central Florida. More detailed information on CMAQ can be found in Appendix 7C. Section 7.5.4 below describes the very limited capacity in which staff used the CMAQ results. As explained below, in the final analysis, staff opted not to use O_3 values calculated from the CMAQ model, but instead only used model results to scale interpolations in the western U.S.

7.5.4 Generation of Potential Ozone Exposure Surfaces (POES)

Staff evaluated ten approaches for interpolating O_3 air quality across the U.S. which included (1) use of the CMAQ model alone; (2) use of only monitoring data with the Voronoi Neighbor Averaging (VNA) technique; and (3) use of a combination of monitoring data and scaling from CMAQ called enhanced Voronoi Neighbor Averaging (eVNA). The evaluations were based on how well the CMAQ model or interpolation techniques were able to predict the 12-hr SUM06, 12-hr W126 and the current 8-hr average form measured at each monitor. For VNA and eVNA evaluations, each monitor was dropped out sequentially and a value for that monitor was interpolated using the remaining monitors. At each monitor site, Normalized Mean Bias (NMB), Normalized Mean Error (NME), Absolute Mean Bias (AMB) and Absolute Mean Error (AME) were calculated (Table 7-1). For more details see discussions in section 7.5.5 below and in the Environmental Assessment TSD (Abt, 2007). From the results of these

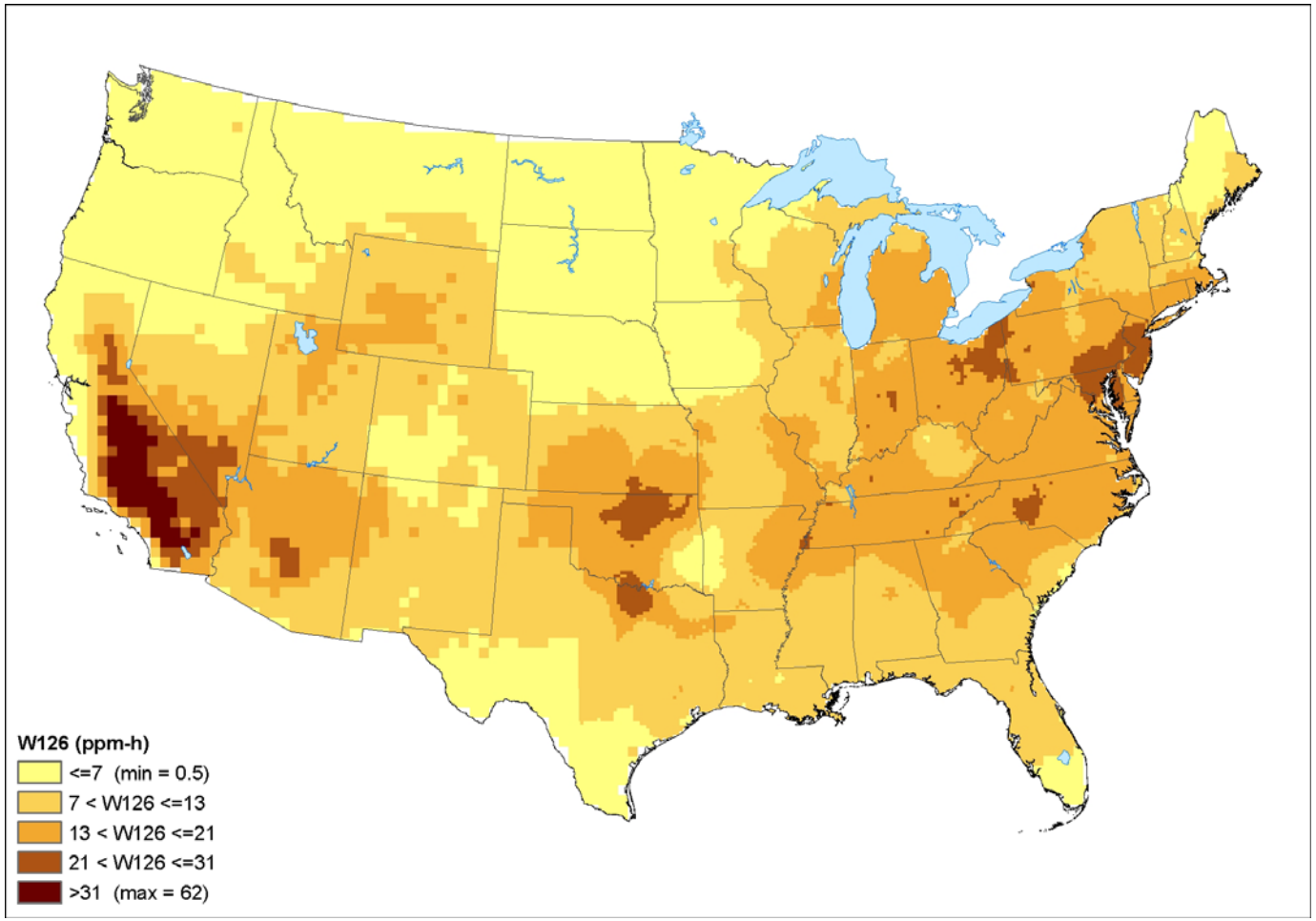
evaluations, the eVNA and VNA performed equally well in many cases. The CMAQ model alone did not perform as well as the VNA and eVNA methods. The staff chose to use separate interpolation techniques in the east and the west. The simpler VNA approach was chosen for the eastern U.S. since it was determined that enhancing the interpolation with CMAQ did not add much information to the eastern U.S. interpolation where the monitoring network has greater coverage than in the west (Figure 7-5). In the west, eVNA was chosen because of the sparse monitoring network in those states. Although the VNA and eVNA interpolation approaches are not as complex or sophisticated as some techniques (e.g., Bayesian methods), they have the advantages of relying on readily available data, being relatively inexpensive to run, and being able to quickly produce estimates of any exposure index, for multiple months or years, and for different air quality scenarios.

To generate the POES, a set of geographical locations for which O_3 data would be interpolated was needed. Ideally these locations would be regularly spaced, cover the continental US, and be close enough to each other to provide a good spatial resolution. Staff chose to use the regularly spaced grid structure of the CMAQ model as the basis for these locations. Specifically, the center of each grid cell was identified both for cells in the 12 km x 12 km grid (which covers only the Eastern U.S.), and the 36 km x 36 km grid (the Western US). This approach produced the densest possible non-redundant "composite" grid of 44432 regularly spaced grid cell center locations throughout the U.S. Using VNA in the eastern U.S. and eVNA in the West, O_3 values were interpolated for each grid cell center in the composite grid (see Environmental Assessment TSD for more details, Abt, 2007).

To support the vegetation exposure and risk assessments, ambient O_3 exposures were projected using seasonal O_3 air quality for the 2001 base year in terms of the 3-month 12-hr W126 (Figure 7-6) and 12-hr SUM06 exposure indices (Figure 7D-1 in Appendix 7D). The uncertainties of this interpolation are discussed below (section 7.5.5). Taking the uncertainties into account and given the absence of more complete monitoring data in rural areas, staff finds the POES serves as a useful tool for identifying areas across the country where O_3 exposure levels would be expected to exceed those known to produce yield loss or biomass loss at given levels for crops and trees, respectively.

Figure 7-6 suggests that under the base year (2001) air quality, a large portion of California had a 12-hr W126 above 31 ppm-hr which has been reported to produce 14% biomass loss in 50% of tree seedlings studies by National Health and Environmental Effects Research Lab, Western Ecology Division (NHEERL-WED). Broader multistate regions in the east and west were predicted to have 12-hr W126 above 21 ppm-hr, which is approximately equal to the secondary standard proposed in 1996. A 12-hr W126 ppm-hr of 21 is associated with a 9% biomass loss in 50% of tree seedlings studied (Lee and Hogsett, 1996). Much of the east and

Figure 7-6. Estimated 12-hr W126 Ozone Exposure – Max 3-months for 2001: "As Is" scenario



Arizona and California have seasonal W126 values above 13 ppm-, which has been associated with a 7% biomass loss in 25% of tree seedlings studied (Lee and Hogsett, 1996). This indicates that current air quality levels could result in significant impacts to vegetation in some areas. However, these exposures may be uncertain in some cases with respect to vegetation with canopy heights below monitor inlet heights, e.g., crops and tree seedlings. In the crop and tree seedling risk/benefit assessments, staff incorporated an adjustment of monitored O_3 to take into account the uncertainty associated with a potential vertical O_3 gradient from the height of the monitoring probe (~4 meters) to the approximate canopy height of crops and seedlings (see section 7.6.2.3).

To evaluate changing vegetation exposures and risks under changing air quality, maps were generated for selected "just meet" scenarios (Figures 7-7, 7-8, 7-9, 7-10) by analytically adjusting air quality distributions with the quadratic method to reflect "just meeting" the level of various alternative standards (see Horst and Duff, 1995; Rizzo, 2005 & 2006). This technique combines both linear and quadratic elements to reduce larger O_3 concentrations more than smaller ones. In this regard, the quadratic method attempts to account for reductions in emissions without greatly affecting lower concentrations near ambient background levels. The following "just meet" air quality scenarios were generated:

- 4^{th}-highest daily maximum 8-hr average of 0.084 ppm (current EPA standard) and 0.070 ppm
- 3-month, 12-hr SUM06 of 25 ppm-hr (alternate standard proposed in the 1996 NAAQS review) and 15 ppm-hr
- 3-month, 12-hr W126 of 21 ppm-hr and 13 ppm-hr

Maps generated for the SUM06 25 and 15 ppm-hr scenarios were nearly identical to the maps of 12-hr W126 levels of 21 and 13 ppm-hr and thus, only maps of the SUM06 25 and 15 ppm-hr scenarios are displayed. When 2001 air quality was rolled back to meeting the level of the current 8-hr standard (0.08 ppm), the overall seasonal 12-hr W126 exposures did not improve very much (Figure 7-7). Under this scenario, some areas in the east improve, but there are still many areas of the country that have seasonal O_3 levels above 12-hr W126 of 21 ppm-hr.

The exposure maps generated for the 0.070 ppm level of the 8-hr average form (Figure 7-8), 12-hr SUM06 of 25 and 15 ppm-hr alternatives (Figures 7-9 and 7-10) and 12-hr W126 of 21 and 15 ppm-hr showed a markedly improved picture of O_3 air quality compared to Figure 7-7. Thus, the staff observes that, except for just meeting the current form, all other alternative standards, when met at all locations, would be expected to provide improved protection of

Figure 7-7. Estimated 12-hr W126 Ozone Exposure – Max 3-months for 2001: Quadratic Rollback to just meet 4th-Highest 8-hr Maximum of >0.084

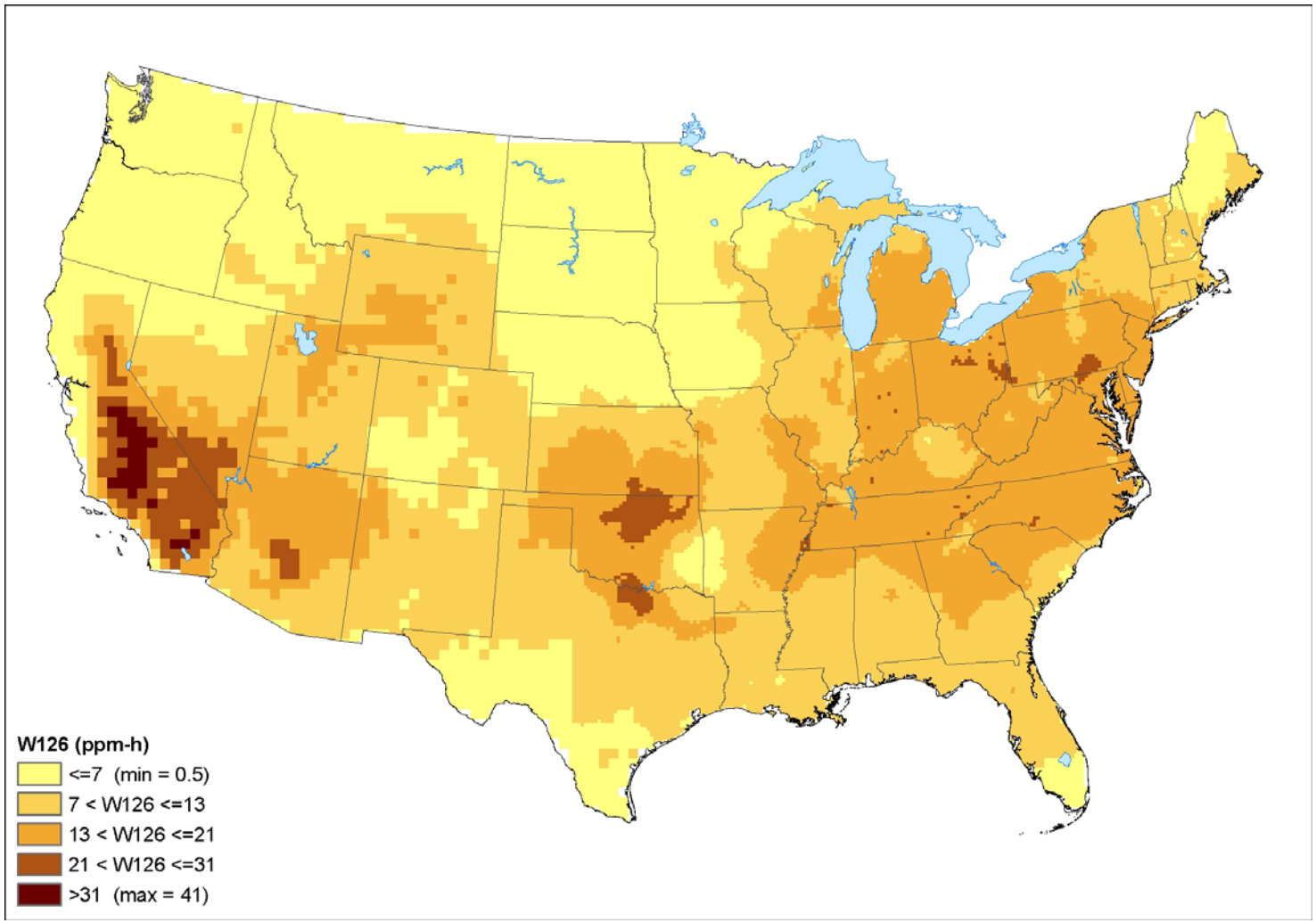


Figure 7-8. Estimated 12-hr W126 Ozone Exposure – Max 3-months for 2001: Quadratic Rollback to just meet 4th Highest 8-hr Maximum of >0.070

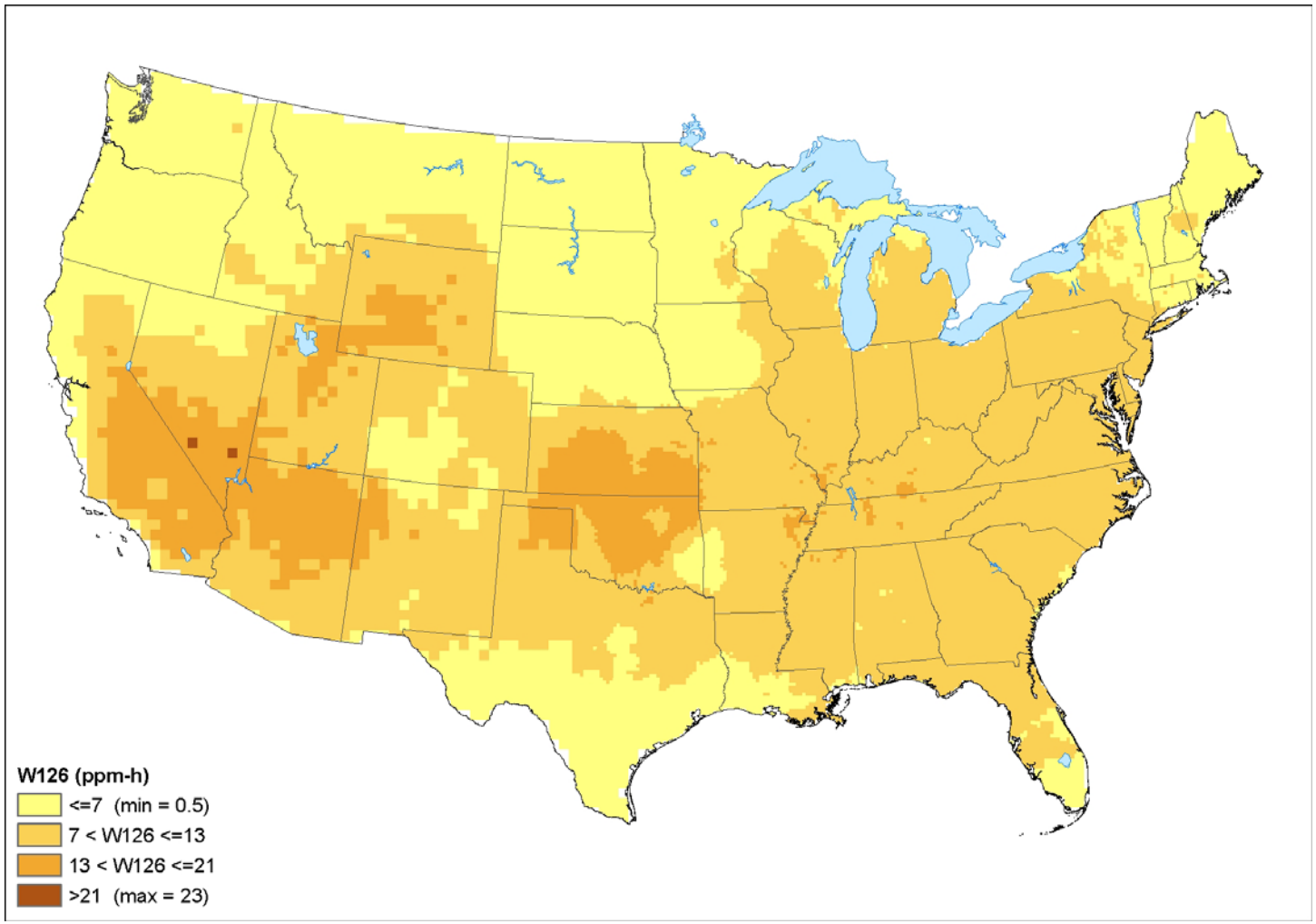


Figure 7-9. Estimated 12-hr W126 Ozone Exposure – Max 3-months for 2001: Quadratic Rollback to just meet 12-hr SUM06 of 25 ppm-hr, secondary standard proposed in 1996

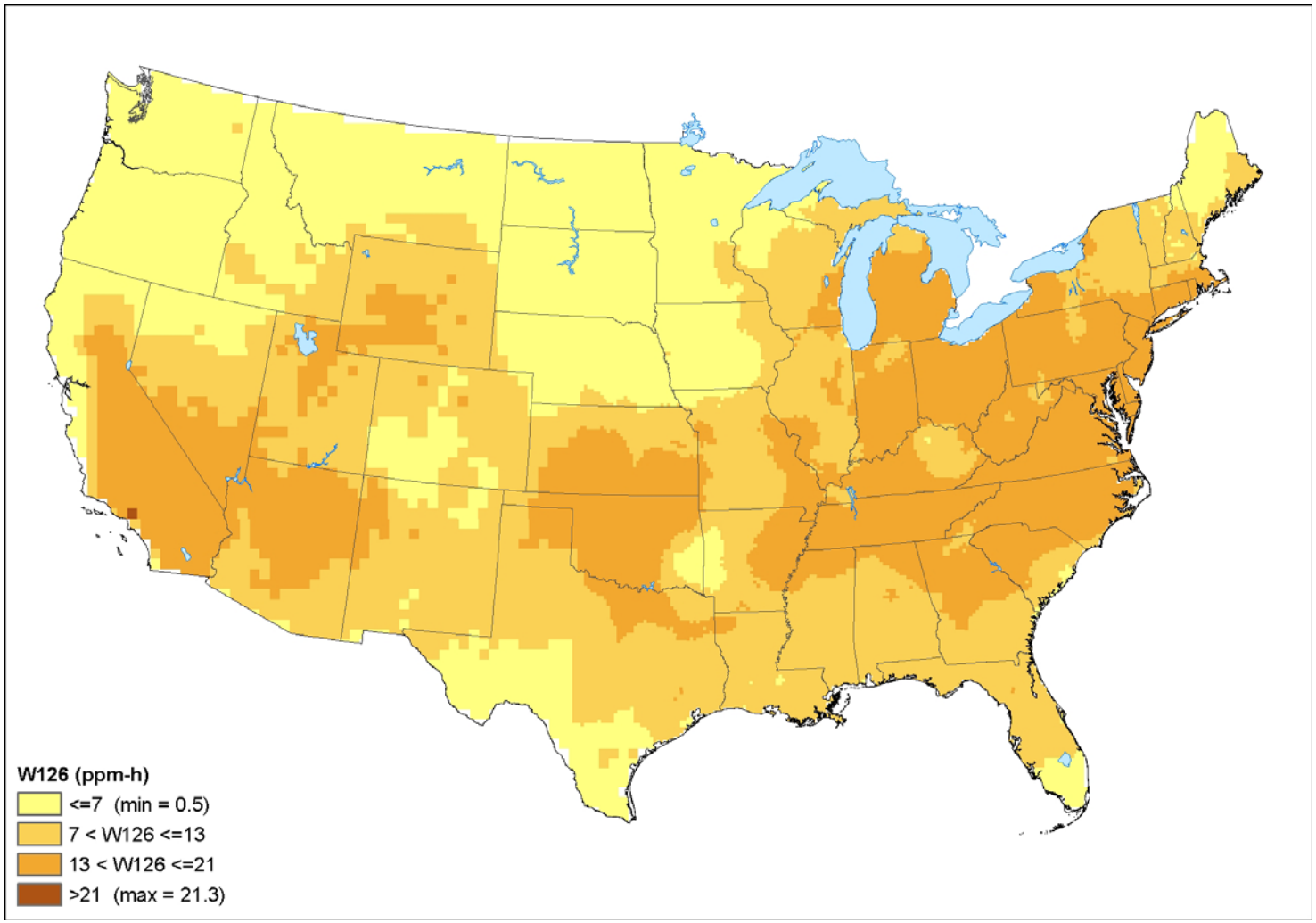
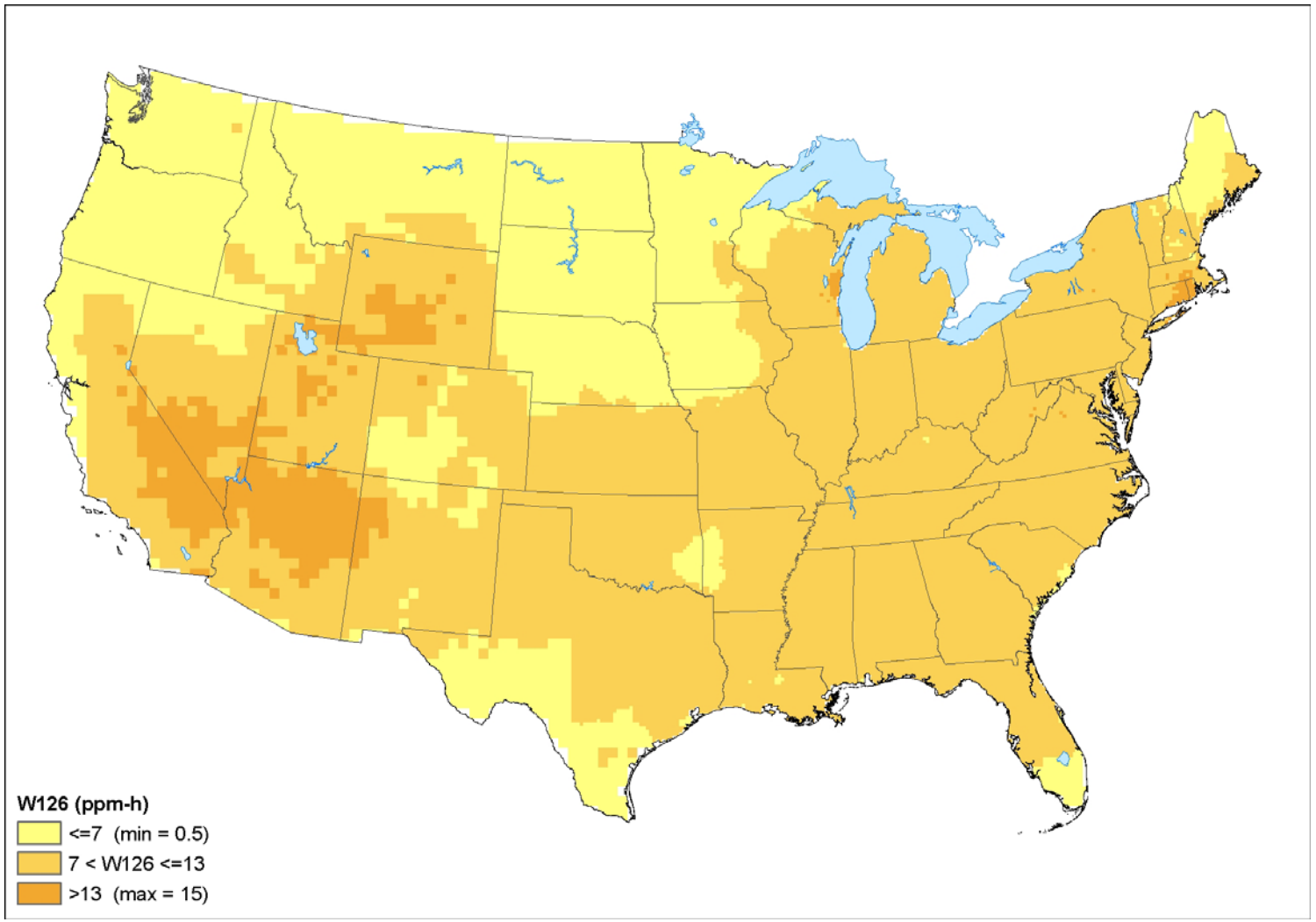


Figure 7-10. Estimated 12-hr W126 Ozone Exposure – Max 3-months for 2001: Quadratic Rollback to just meet 12-hr SUM06 of 15 ppm-hr



vegetation from seasonal O_3 exposures of concern over the current standard. As expected, however, the greatest improvements in air quality and estimated exposures to sensitive vegetation were observed when just meeting the lower 12-hr W126 of 13, 12-hr SUM06 of 15 and 0.07 ppm, 8-hr average scenarios.

7.5.5 Uncertainties in the O_3 Exposure Analysis

Staff recognizes there are inherent uncertainties in using an interpolation that must rely on sparse data that, for the most part, are representative of urban and near-urban areas. This network could bias the picture of the O_3 exposure estimate especially in the western U.S. where monitoring sites can be very far apart. Intuitively, it is expected that the eVNA approach with spatial scaling from CMAQ approach would be an improvement over a simple interpolation in the West. However, it is difficult to test for this because of the paucity of monitoring sites in the western U.S.

To quantify the uncertainty associated with the exposure surface, each monitoring site was sequentially dropped out of the interpolation and recalculated with the remaining monitoring sites. At each monitoring site, Normalized Mean Bias (NMB), Normalized Mean Error (NME), Absolute Mean Bias (AMB) and Absolute Mean Error (AME) were calculated. These statistics are defined below:

$$NMB = average_{i \in dropouts}\left(100 * \frac{predictedMETRIC_i - actualMETRIC_i}{actualMETRIC_i}\right)$$

$$NME = average_{i \in dropouts}\left(100 * \frac{\left|predictedMETRIC_i - actualMETRIC_i\right|}{actualMETRIC_i}\right)$$

$$AMB = average_{i \in dropouts}(predictedMETRIC_i - actualMETRIC_i)$$

$$AME = average_{i \in dropouts}(|predictedMETRIC_i - actualMETRIC_i|)$$

This method of evaluation may result in a slight overestimation of error and bias for the exposure surface since dropping out monitors loses information that the interpolation uses in that local area. Summary error and bias metrics are presented in Tables 7-1a and b. Using all the monitors, the Eastern U.S. interpolation had an NME of about 26% for the 12-h SUM06 metric (Table 7-1a). Western U.S. interpolation had a much higher NME of approximately 62%. However, since SUM06 and W126 values are often low numbers, NME can be calculated to be

large while the absolute difference is small. For example, if a monitor with a W126 of 4 ppm-hr is measured and the interpolation predicts a W126 of 6 ppm-hr then the NME would be 50%. Therefore, staff thought it was useful to also report the absolute mean bias and error. In absolute terms, the average bias for SUM06 was slightly low (-1.83 ppm-hr in the East and -2.62 ppm-hr in the West). CASTNET monitors are also presented to illustrate how well the interpolation techniques predicted air quality in that rural monitoring network. In general, the interpolations in the East and West under-predicted the 12-hr SUM06 values. This under-prediction is likely a result of the averaging inherent in the interpolation. Similar results are seen for the 12-hr W126 (Table 7-1b). However, in almost all cases, the interpolation was able to predict monitored W126 slightly better than monitored SUM06. The calculation of error and bias metrics for the interpolation represents a notable improvement over the 1996 assessment which did not have an evaluation of the error and bias associated with the exposure surface.

Figure 7-11 also depicts predicted W126 values, from the sequential drop-out exercise, against the actual W126 values measured at CASTNET monitors and AQS monitors designated as "rural." This figure gives a graphical representation of how well the O_3 exposures were predicted in the rural monitors away from urban areas. A perfect prediction would result in all points aligning on the black "one-to-one" line. In general, these graphs indicate that the interpolation technique slightly overestimated W126 exposure at the low levels and underestimated W126 exposure at the high levels. Biologically, the more significant error is at the high exposures, since vegetation responds more at high exposures. Figure 7-11 indicates, in general, that the most relevant high exposures were underestimated. This may have implications for the subsequent calculation of crop yield and tree seedling biomass loss, potentially resulting in an underestimation of risk in some areas. More detailed information from this analysis is presented in the Environmental Assessment TSD (Abt, 2007).

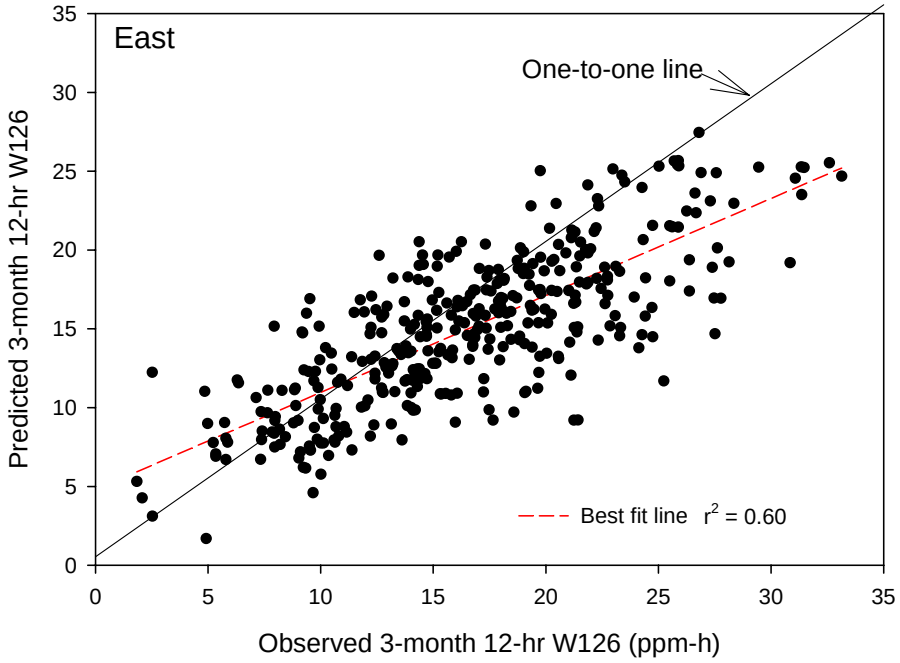
Table 7-1a. Evaluation statistics for the 3-month 12-hr SUM06 interpolations of the Eastern and Western U.S. domains. NMB is Normalized Mean Bias, NME is Normalized Mean Error, AMB is Absolute Mean Bias and AME is Absolute Mean Error. An explanation of these metrics is given in section 7.5.5.

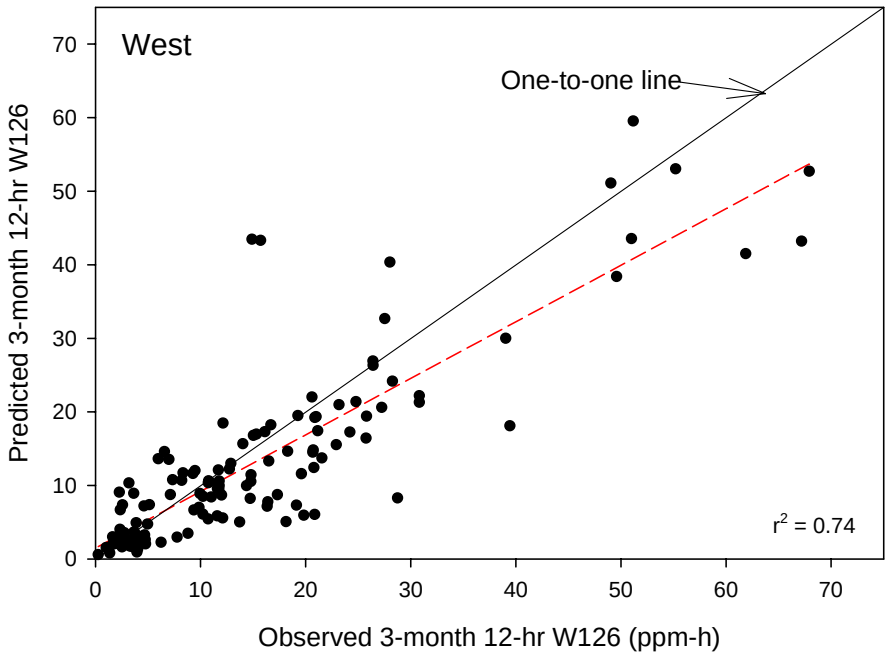
Region	Monitors	NMB (%)	NME (%)	AMB (ppm-hr)	AME (ppm-hr)
Eastern US	All monitors	-0.04	25.78	-1.83	4.07
Eastern US	CASTNET only	-8.84	20.76	-2.95	4.79
Western US	All monitors	16.46	62.39	-2.62	6.05
Western US	CASTNET only	-6.03	42.12	-2.15	7.98

Table 7-1b. Evaluation statistics for the 3-month 12-hr W126 interpolations of the Eastern and Western U.S. domains.

Region	Monitors	NMB (%)	NME (%)	AMB (ppm-hr)	AME (ppm-hr)
Eastern US	All monitors	-1.06	21.92	-1.21	2.97
Eastern US	CASTNET only	-8.43	17.44	-2.00	3.22
Western US	All monitors	14.57	48.38	-1.50	4.27
Western US	CASTNET only	0.67	41.47	-0.60	5.21

Figure 7-11. Comparison of predicted versus observed 12-hr W126 at CASTNET and "rural" AQS monitors. Monitor data was predicted by dropping out each monitor sequentially and interpolated with the all remaining monitors.



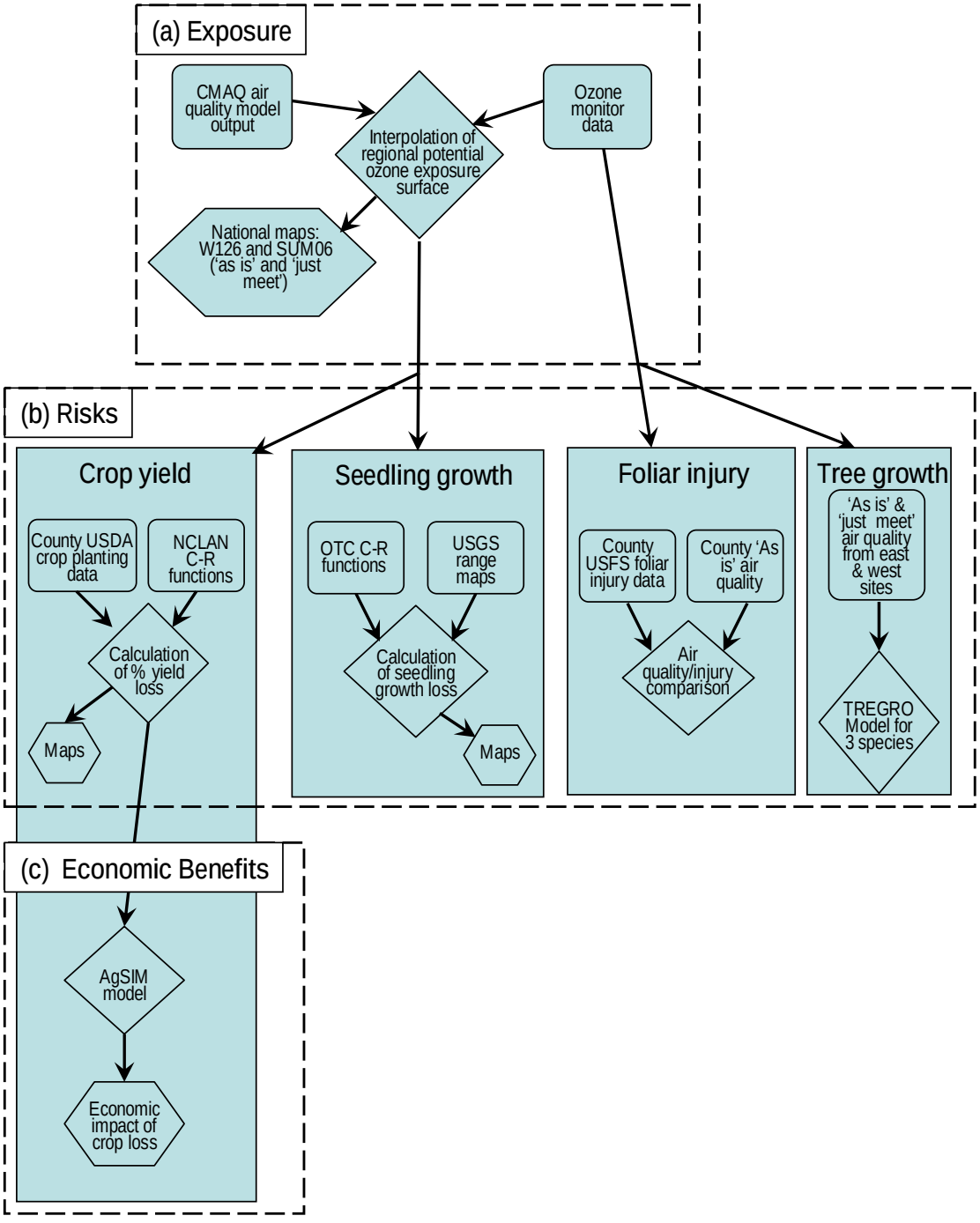


7.6 CHARACTERIZATION OF VEGETATION RISKS

7.6.1 Scope of Vegetation Risk Assessment

The vegetation impact assessment conducted for the current review (see Figure 7-12a-c), consists of exposure, risk and benefits analyses and improves and builds upon the similar analyses performed in support of the 1996 secondary NAAQS review (U.S. EPA 1996b). The vegetation exposure assessment was discussed above in section 7.5. The organization of this section reflects the remaining risk and benefit components of the assessment. The vegetation risk discussion which follows is divided between the crop and tree analyses. The crop analysis discussed in section 7.6.2 includes estimates of the risks to crop yields from current and alternative O_3 exposure conditions and the associated change in economic benefits expected to accrue in the agriculture sector upon meeting the levels of various alternative standards. The tree risk analysis described in section 7.6.3 includes three distinct lines of evidence: (1) estimates of seedling growth loss under current and alternative O_3 exposure conditions; (2) observations of visible foliar injury in the field linked to recent monitored O_3 air quality for the years 2001 - 2004; and (3) simulated mature tree growth reductions using the TREGRO model to simulate the effect of meeting alternative air quality standards on the predicted annual growth of a single western species (ponderosa pine) and two eastern species (red maple and tulip poplar). Both quantitative and qualitative discussions of known sources and ranges of uncertainties associated with the components of this assessment are also discussed.

Figure 7-12 (a-c). Major Components of Vegetation Risk Assessment



7.6.2 Characterization of Crop Risks and Associated Economic Benefits

7.6.2.1 Exposure Methodologies Used in Vegetation Research

In the 1996 review, O_3 exposure studies were dominated by the use of various versions of the open-top chamber (OTC), first described by Heagle et al. (1973) and Mandl et al. (1973). Hogsett et al. (1985, 1987) described in detail many of the subsequent modifications to the original OTC design. The OTC method continues to be a widely used technique in the U.S. and Europe for exposing plants to varying levels of O_3 (U.S. EPA, 2005b).

Chambered systems, including OTCs, have several advantages. For instance, they can provide a range of treatment levels including charcoal-filtered (CF), clean-air control, and above ambient concentrations for O_3 experiments. Depending on experimental intent, a replicated, clean-air control treatment is an essential component in many experimental designs. The OTC can provide a consistent, definable exposure because of the constant wind speed and delivery systems. From a policy perspective, the statistically robust concentration-response (C-R) functions developed using such systems are necessary for evaluating the implications of various alternative air quality scenarios on vegetation response.

Nonetheless, there are several characteristics of the OTC design and operation that can lead to exposures that might differ from those experienced by plants in the field. First, the OTC plants are subjected to constant turbulence, which, by lowering the boundary layer resistance to diffusion, which may result in increased uptake. This may lead to an overestimation of effects in areas with less turbulence (Krupa et al., 1995; Legge et al., 1995). As with all methods that expose vegetation to modified O_3 concentrations in chambers, OTCs create internal environments that differ from ambient air. This so-called "chamber effect" refers to the modification of microclimatic variables, including reduced and uneven light intensity, uneven rainfall, constant wind speed, reduced dew formation, and increased air temperatures (Fuhrer, 1994; Manning and Krupa, 1992). However, staff notes that the uncertainties associated with the influence of other modifying factors occurring in the field such as water and nutrient availability (see discussion above in section 7.4.2), are likely to be greater than the uncertainties in the data due to the influence of OTCs. Because of the standardized methodology and protocols used in National Crop Loss Assessment Network (NCLAN) and other programs, the database can be assumed to be internally consistent.

While it is clear that OTCs can alter some aspects of the microenvironment and plant growth, the question to be answered is whether or not these differences affect the relative response of a plant to O_3. As noted in the 1996 O_3 CD (U.S. EPA, 1996a), evidence from a number of comparative studies of OTCs and other exposure systems suggested that responses were essentially the same regardless of exposure system used and chamber effects did not

significantly affect response. For example, a study of chamber effects examined the responses of tolerant and sensitive white clover clones (*Trifolium repens*) to ambient O_3 in greenhouse, open-top, and ambient plots (Heagle et al., 1996). The response found in OTCs was the same as in ambient plots. The California Air Resources Board (CARB), during its recent O_3 standard review, came to a similar conclusion about the usefulness of OTC data. Its review states "there is little scientific justification for the categorical discounting of O_3 yield-response relationships obtained using the OTC technology" (CEPA, 2005).

In recent years, a few studies have employed a modified Free Air CO_2 Enrichment (FACE) methodology to expose vegetation to elevated O_3 without using chambers. This exposure method was originally developed to expose vegetation to elevated levels of CO_2, but has been modified to include O_3 exposure in Illinois (SoyFACE) and Wisconsin (AspenFACE) for soybean and deciduous trees, respectively (Dickson et al., 2000; Morgan et al., 2004). The FACE method releases gas (e.g., CO_2, O_3) from a series of orifices placed along the length of the vertical pipes surrounding a circular field plot and uses the prevailing wind to distribute it. This exposure method may more closely replicate conditions in the field and, more importantly for forest research, has the benefit of being able to expand vertically with the growth of the trees, allowing for exposure experiments to span numerous years.

The FACE methodology has a different set of limitations than those of the OTC. Most importantly, it is not possible with FACE to produce a number of replicated treatment levels, including O_3 concentrations below ambient levels that are needed to build the statistically robust C-R functions possible with OTCs. One also must recognize the potential for significant gradients of exposure gas concentrations throughout the FACE exposure rings. While the FACE protocols minimize exposure concentration gradients, plants near the gas emitters will be exposed to larger concentrations than centrally located plants near the air monitoring point. There is little information on within-plot O_3 concentrations in FACE-type exposures and this issue needs to be addressed more fully to understand O_3 exposure and response data from FACE studies. Despite the differences in these two exposure methods, recent evidence obtained using FACE and OTC systems appear to support the results observed in OTC studies used in the 1996 review. For example, a series of studies undertaken at AspenFACE (Isebrands et al., 2000, 2001) showed that O_3-symptom expression was generally similar in OTCs, FACE, and ambient-O_3 gradient sites, and supported the previously observed variation among trembling aspen clones (*Populus tremuloides* L.) using OTCs (Karnosky et al., 1999).

In the SoyFACE experiment in Illinois, soybean (Pioneer 93B15 cultivar) yield loss data from a two-year study was recently published (Morgan et al., 2006). This cultivar is a recent selection and, like most modern cultivars, has been selected with an already high current O_3. It was found to have average sensitivity to O_3 compared to 22 other cultivars tested at SoyFACE.

In this experiment, ambient hourly O_3 concentrations were increased by approximately 20% and measured yields were decreased by 15% in 2002, as a result of the increased O_3 exposure (Morgan et al., 2006). To compare these results to chamber studies, Morgan et al. (2006) calculated the expected yield loss from a linear relationship constructed from chamber data using 7-hr seasonal averages (Ashmore, 2002). They calculated an 8% expected yield loss from the 2002 O_3 exposure which was surprisingly less than the measured 15% yield loss. Staff believes that the expected yield loss may have been closer if the authors used C-R functions based on the W126 metric. Nonetheless, the results from this study suggest that C-R relationships developed from chambers are not overestimating response of recently developed soybean cultivars to elevated O_3 exposure. As more FACE data become available, a more quantitative comparison of findings from the SoyFACE and AspenFACE systems would be useful.

Other exposure methods described both in the 1996 and 2006 O_3 CDs (U.S. EPA, 1996a; U.S. EPA 2006) also provided useful information on plant responses to O_3 exposure. For example, Gregg et al. (2003), found significant effects of O_3 on the growth of cottonwood saplings along an ambient O_3 gradient in the New York City area, similar to those reported in OTCs (see section 7.6.3. Other exposure methods include but are not limited to chemical protectants (e.g., ethylenediurea [EDU]) and O_3 exclusion. Nonetheless, given a continued policy need for robust C-R functions to evaluate vegetation response under alternative air quality scenarios and the apparent consistency between plant responses using OTC and other methods, staff concludes that the robust C-R functions derived using the OTC methodology are currently the most useful in a policy context and we continue to rely on them in the following analyses.

7.6.2.2 Basis for C-R Functions

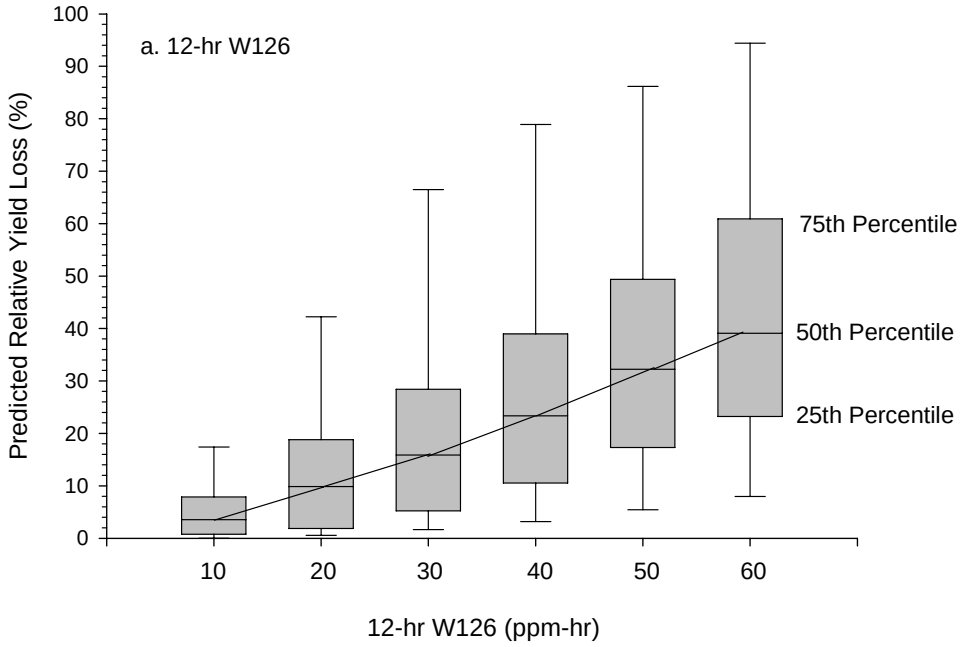
The 1996 crop assessment was built upon the NCLAN O_3 C-R functions. Since very few new studies have published C-R functions that would be useful in an updated assessment, C-R functions from NCLAN remain the best data available for a national assessment of crop loss under various O_3 air quality scenarios. The NCLAN protocol was designed to produce crop C-R functions representative of the areas in which the crops were typically grown. The U.S. was divided into 5 regions over which a network of field sites was established. In total, 15 crop species (corn, soybean, winter wheat, tobacco, sorghum, cotton, barley, peanuts, dry beans, potato, lettuce, turnip, and hay [alfalfa, clover, and fescue]), were studied. The first 12 of these 15 listed species were analyzed for the 1996 review and included 38 different cultivars studied under a variety of unique combinations of sites, water regimes, and exposure conditions, producing a total of 54 separate cases. Figure 7-13 uses the regression equations for each of the 54 cases to graph predicted relative yield loss at various exposure levels in terms of a 12-hr W126 (Figures 7E-1, 2, 3 present similar figures with the 8-hr average and 12-hr SUM06 forms).

Figures 7-14 (a-d) show composite graphs for some individual crops from NCLAN and the variations in sensitivity between important crops. According to the most recent USDA National Agricultural Statistical Survey (NASS) data, the 12 species analyzed in the last O_3 NAAQS review account for greater than 70% of principal crops acreage planted in the U.S. in 2004.[1] Corn, soybean, and winter wheat alone accounted for 62% of 2004 principal crop acreage planted. For the economic analysis described in section 7.6.2.4, a reduced list of 9 species (69% of 2004 principal crops) were included (e.g., cotton, field corn, grain sorghum, peanut, soybean, winter wheat, lettuce, kidney bean, potato), with tobacco, turnip and barley not evaluated.

Since the NCLAN studies were performed during the years 1980 to 1988, there is some uncertainty whether the crop cultivars tested in NCLAN are representative of crops grown today. In general, new crop varieties are not specifically bred for O_3 tolerance and the cultivars used today were bred from the same very narrow genetic stock available in the 1980's. Thus, it is not expected that there would be much difference in O_3 tolerance between cultivars used today and when the NCLAN studies were done. Since the last review, there has been no evidence that crops are becoming more tolerant of O_3 (U.S. EPA, 2006). For cotton, some newer varieties have been found to have higher yield loss due to O_3 compared to older varieties (Olszyk et al., 1993, Grantz and McCool, 1992). In a meta-analysis of 53 studies, Morgan et al. (2003) found consistent deleterious effects of O_3 exposures on soybean from studies published between 1973 and 2001. Further, early results from the SoyFACE experiment in Illinois indicate a lack of any apparent difference in the O_3 tolerance of old and recent cultivars of soybean in a study of 22 soybean varieties (Long et al., 2002).

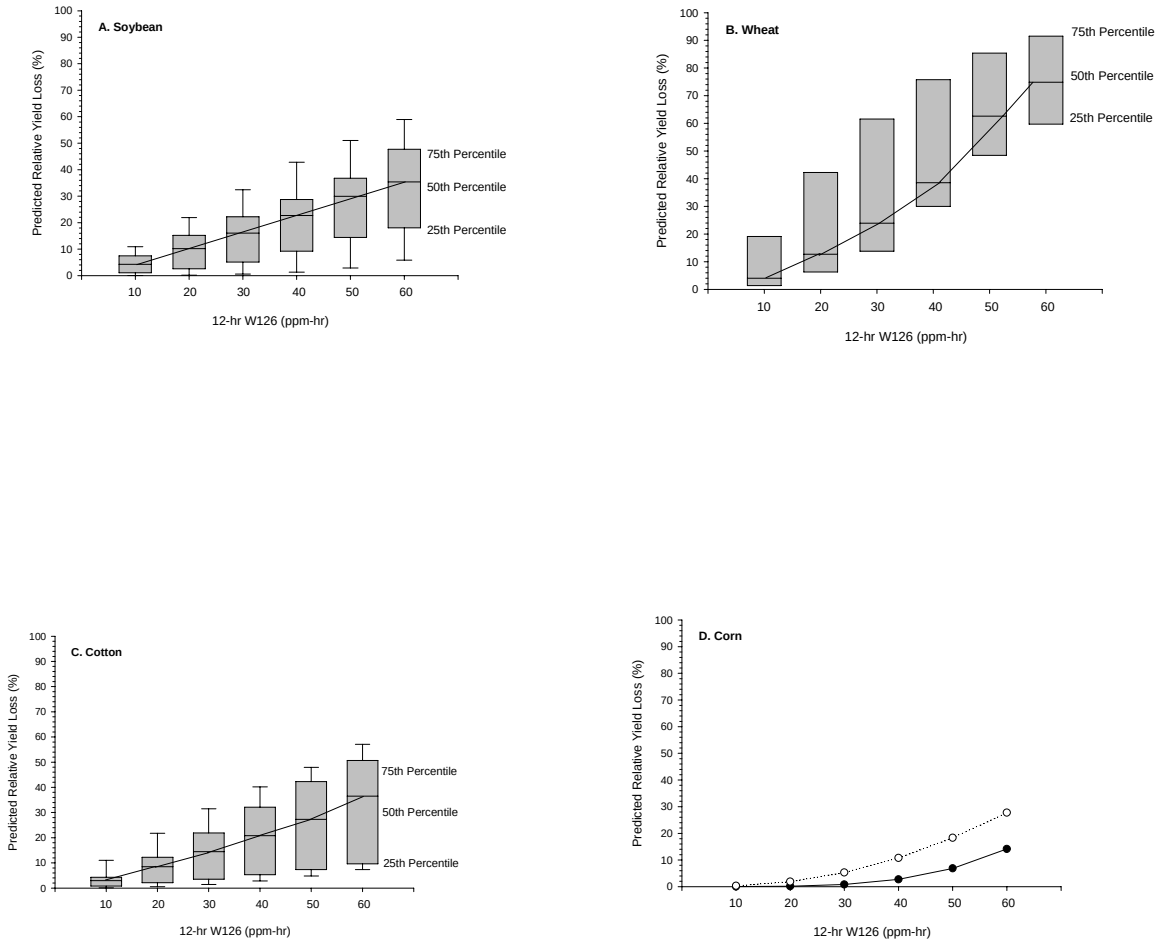
[1] Principal crops as defined by the USDA include corn, sorghum, oats, barley, winter wheat, rye, Durum wheat, other spring wheat, rice, soybeans, peanuts, sunflower, cotton, dry edible beans, potatoes, sugar beets, canola, proso millet, hay, tobacco, and sugarcane. Acreage data for the principal crops were taken from the USDA NASS 2005 Acreage Report (http://usda.mannlib.cornell.edu/reports/nassr/field/pcp-bba/acrg0605.pdf).

Figure 7-13. Median crop yield loss from NCLAN crops characterized with the 12-hr W126



Distribution of yield loss predictions from Weibull exposure-response models that relate yield to O_3 exposure characterized with the 12-hr W126 statistic using data from 31 crop studies from National Crop Loss Assessment Network (NCLAN). Separate regressions were calculated for studies with multiple harvests or cultivars, resulting in a total of 54 individual equations from the 31 NCLAN studies. Each equation was used to calculate the predicted relative yield or biomass loss at a 12-hr W126 of 10, 20, 30, 40, 50, and 60 ppm-hr, and the distributions of the resulting loss were plotted. The solid line represents the 50th percentile. Source: U.S. EPA, 1996a; Lee and Hogsett 1995.

Figure 7-14 (A-D). Median soybean (A), wheat (B), cotton (C) and corn (D) yield loss from NCLAN crops characterized with the 12-hr W126



Distribution of yield loss predictions from Weibull exposure-response models that relate yield to O_3 exposure characterized with the 12-hr W126 statistic using data from 22 soybean, 7 wheat, 9 cotton and 2 corn studies from National Crop Loss Assessment Network (NCLAN). Separate regressions were calculated for studies with multiple harvests or cultivars. Each equation was used to calculate the predicted relative yield loss at a 12-hr W126 of 10, 20, 30, 40, 50, and 60 ppm-hr, and the distributions of the resulting loss were plotted. Source: U.S. EPA, 1996a; Lee and Hogsett 1995.

7.6.2.3 Considerations for Exposures at Crop Canopy Height

An important consideration when predicting crop yield and/or tree seedling biomass loss using monitored O_3 exposure levels is the potential positive exposure bias associated with the height at which the measurement is taken. Ambient monitor inlets are typically at heights of 3 to 5 meters, and thus are located in the inner part of the planetary boundary layer (U.S. EPA, 2005b). It is well known that within this layer O_3 reacts with vegetation, other surfaces and volatile compounds and can create a vertical gradient of decreasing O_3 concentration from the inlet height of the monitors to the canopies of short vegetation. The magnitude of the gradient is determined in large part by the intensity of turbulent mixing in the surface layer. During daytime hours, the vertical O_3 gradient is relatively small because turbulent mixing maintains the downward flux of O_3. For example, Horvath et al. (1995) calculated a 7% decrease in O_3 going from a height of 4 meters down to 0.5 meters above the surface during unstable (or turbulent) conditions in a study over low vegetation in Hungary [see section AX3.3.2. of the 2006 CD (U.S. EPA, 2006)]. This is compared to a 20% decrease during stable conditions, which usually occur during the night. The average decrease for all times measured was 10%. The daytime versus nighttime bias is an important distinction since the assessments outlined below rely heavily on daytime metrics, such as the12-hr SUM06 and W126. Thus, staff selected 10% as a daytime downward adjustment factor to apply to hourly monitor-derived exposures (including interpolated values) prior to estimating crop yield and tree seedling biomass loss values. We consider this 10% adjustment at the upper-end of the differences between the monitor height and top of the canopy of low vegetation in the daytime.

Staff recognizes that a 10% adjustment to hourly monitoring data across the country is a very simple method to deal with a complicated issue. The exchange of O_3 between the atmosphere and vegetation is controlled by complex interactions of meteorological and biological processes. Ideally one should account for the exact height of each monitor, canopy roughness for each vegetation type and the seasonal and diurnal nature of turbulence. This was not possible in our analyses. To bound the uncertainty associated with applying a 10% adjustment to all monitors and short canopies, staff performed a sensitivity analysis by also calculating crop and tree seedling assessments without an adjustment. Staff agrees with CASAC comments that these calculations will provide a bracket of responses within which the reality probably lies for the true exposure of O_3 to short vegetation (Henderson, 2006c). For brevity, staff has presented the 10% adjusted figures in the main body of the Staff Paper and have placed companion figures without the 10% adjustment in the Appendices 7-G and 7-H. However, both sets of results are discussed in this chapter.

The inclusion of a 10% hourly adjustment had a substantial effect on the predicted 12-hr SUM06 and W126 exposures. Reducing each hourly value by 10% over the entire interpolated surface resulted in an average reduction of the 3-month 12-hr SUM06 by 53% and an average reduction of 42% in the 3-month 12-hr W126. These large reductions in the SUM06 and W126 exposures are most likely a result of many monitored hourly concentrations occurring near the SUM06 threshold and the inflection point for W126 (approximately 0.06 ppm). When these "mid-level" hourly O_3 values are reduced by 10%, many fall below 0.06 ppm, decreasing the amount of hourly values counted in (SUM06) or contributing to (W126) these metrics.

Given the somewhat lesser impact of the 10% adjustment on exposures using the W126 and the lack of evidence for a biological threshold for effects at 0.06 ppm, staff considered the W126 index form more appropriate for conducting the crop yield and tree seedling biomass loss risk assessment. Other information that supports this decision includes: 1) studies that document effects on crops and other sensitive vegetation at O_3 concentrations below 0.06 ppm [e.g., exposures as low as a 0.04 ppm 7-hr seasonal average (U.S. EPA, 2006)]; and 2) the high degree of correlation between both forms when describing ambient exposures (see Figure 7-4) and their similar predictive power of NCLAN crop data results in retrospective analyses (Lee et al., 1989; U.S. EPA, 1996a, 2006).

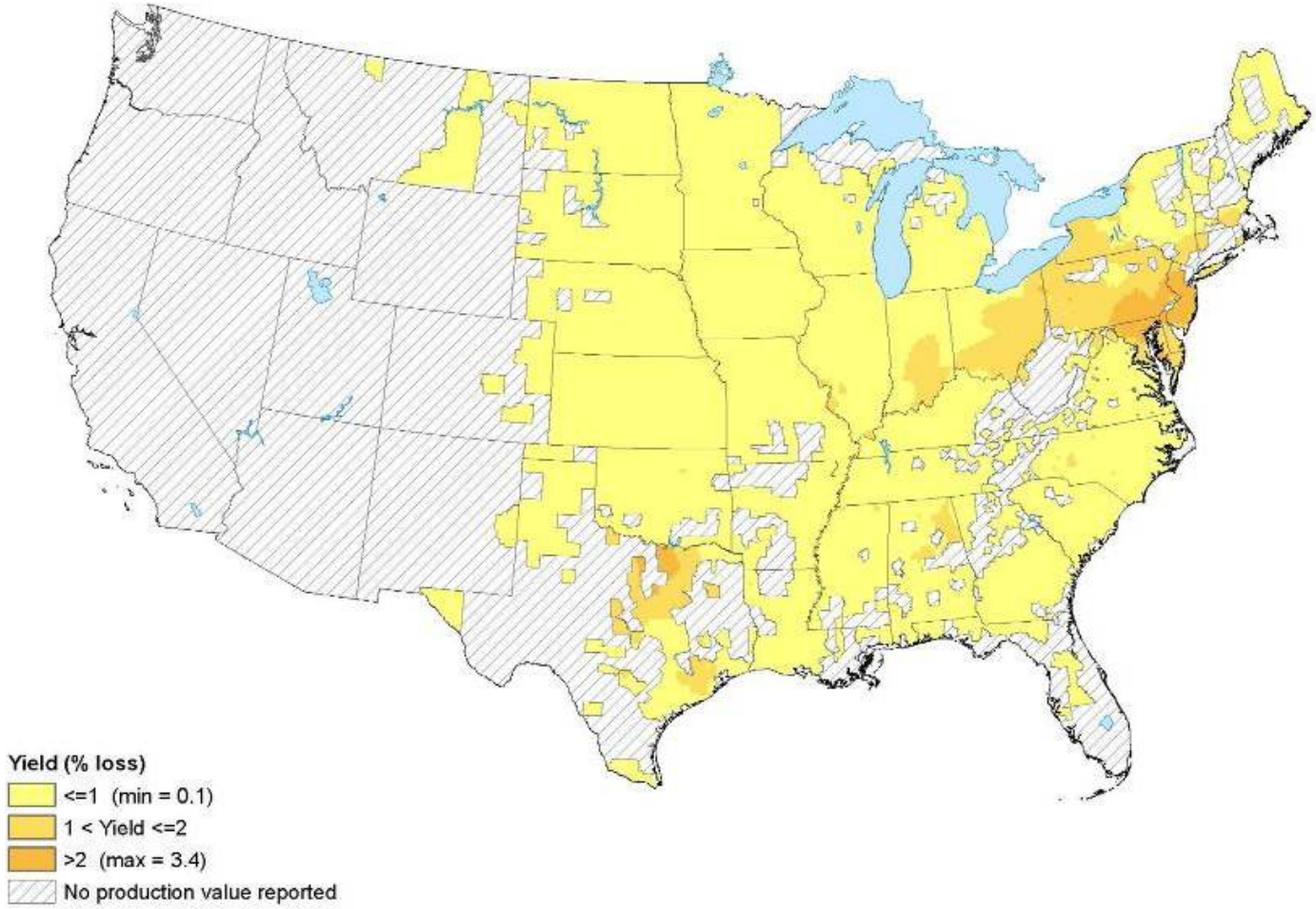
7.6.2.4 Quantifiable Risk of Yield Loss In Select Commodity, Fruit and Vegetable Crops

The 2001 county-level crop planting data were obtained for the 9 commodity crops (corn, soybean, winter wheat, sorghum, cotton, peanuts, kidney bean, potato & lettuce) from USDA-NASS (National Agricultural Statistics Service; http://www.usda.gov/nass). The appropriate NCLAN C-R functions (available in the 12-hr W126 format) for each of the nine commodity crops were identified from the analysis done for the 1996 Staff Paper (U.S. EPA 1996b, Table 7F-1). The C-R functions for six fruit and vegetable species (tomatoes-processing, grapes, onions, rice, cantaloupes, Valencia oranges) were identified from the 1996 California fruit and vegetable analysis (Table 7F-2). Staff notes that fruit and vegetable studies were not part of the NCLAN program and C-R functions were available only in terms of seasonal 7 hr or 12-hr mean index. This index form is considered less effective in predicting plant response for a given change in air quality than the cumulative form used with other crops. Therefore, staff considers the fruit and vegetable C-R functions more uncertain than those for commodity crops. Staff combined the C-R functions with the crop planting information and with projections of 2001 O_3 exposure based on a 12-hr W126 calculated for the 3 months prior to the harvest date for each commodity crop and the appropriate growing season 7-hr or 12-hr average used for some fruits and vegetables. Calendar periods used for computing W126, 7-hr and 12-hr exposure statistics,

are based on the harvest date and are done on a state-specific basis. This allows for geographic variation and better reflects actual O_3 exposure during the true growing period of the crop so that calculated expected yield change for each crop, fruit and vegetable is specific to where they were planted (Abt, 2007).

Some of the results of this risk assessment are presented in Appendix 7F in Table 7F-4. This table depicts the maximum county-level relative change in crop yield loss under air quality scenarios of just meeting various alternative standard options under consideration using the median C-R functions. Maps of predicted yield loss for selected major crops are presented in Appendix 7G. Figure 7-15 shows a map of predicted yield loss for soybean from 2001 using the 10% adjusted "as is" estimated O_3 exposure scenario. Soybean is predicted to have the largest yield loss in southwestern Pennsylvania, southern New Jersey and east Texas. However, these areas are not places of high soybean production. In a high soybean producing state, such as Illinois, yield loss was predicted to reach a maximum range of 1-2% with a 10% adjusted O_3 exposure (Figure 7-15) and 3-4% without a 10% adjusted O_3 exposure (Figure 7G-1 in Appendix 7G). Corn, another major commodity crop, was not predicted to have any loss in 2001. This is because the two corn cultivars studied in NCLAN were not sensitive to O_3. In contrast, cotton, a more sensitive crop, had predicted yield loss above 10% in southern California (see Appendix 7G, Figures 7G-3 & 4).

Figure 7-15. Estimated soybean yield loss based on interpolated 2001 3-month 12-hr W126 with a 10% downward adjustment of hourly O_3 concentrations.



7.6.2.5 Economic Benefits Assessment – AGSIM

This section presents results of the quantitative economic benefits analysis associated with just meeting alternate standards. Adequate data are currently available to assess economic benefits for 9 of the commodity crops studied in the NCLAN project and 6 fruit and vegetable species. Fruits and vegetables were evaluated in the 1996 review using a separate regional benefits model separate from the national commodity crop model (U.S. EPA 1996b). This was due to the fact that only regional planting data were available at the time for those fruits and vegetables. In the current benefits assessment, both commodity crops and fruits and vegetables were evaluated together in the same national scale model. Fruit and vegetables are a large part of the U.S. agricultural sector and may be especially susceptible to O_3 pollution because much of the production is located in the San Joaquin Valley region of California, which has very high levels of O_3 exposure (CEPA, 2005). Because 6 of fruits and vegetables were not a part of the NCLAN program and the uncertainties inherent in those experiments are less well known, information on fruits and vegetables is presented separately in this document. Nonetheless, fruits and vegetables are large portion of the U.S. agricultural economy. For example, in 2004, cash income from California fruit and nut production was worth more than 9.6 billion dollars and over 7.2 billion dollars for vegetable crops (California Agricultural Resource Directory, 2005, http://www.cdfa.ca.gov/).

The Agriculture Simulation Model (AGSIM) (Taylor, 1994; Taylor, 1993) has been utilized recently in many major policy evaluations.[2] AGSIM is an econometric-simulation model used to calculate agricultural benefits of changes in O_3 exposure and is based on a large set of statistically estimated demand and supply equations for agricultural commodities produced in the U.S. A number of updates to AGSIM were performed before running this analysis: (1) an update of the commodity data for 2001, (2) incorporation of the most recent version of the official USDA baseline model, (3) an econometric component added to AGSIM to compute total farm program payments for different levels of farm program parameters, and (4) a farm payment program component was added to the economic surplus module. The AGSIM model was run to provide benefit estimates for nine major commodity crops (soybeans, corn, winter wheat, cotton, peanuts, sorghum, potato, lettuce, kidney bean) and six fruits and vegetables mainly grown in California (tomatoes-processing, grapes, onions, rice, cantaloupes, Valencia oranges). As

[2] For example, AGSIM© has been used in EPA's prospective study of the benefits derived from the Clean Air Act Amendments of 1990 required by section 812-B of the Clean Air Act, non-road, land-based diesel engine rule, and proposed Clear Skies legislation.

described earlier, hourly O_3 exposures were adjusted downward by 10% before calculating the W126, 7-hr or 12-hr exposure metrics.

Percent relative yield losses (PRYL) calculated as the change in yield occurring between just meeting 'as is' air quality and various alternative standard scenarios were the relevant input parameters to the AGSIM model. The AGSIM model predicted acreage, production, supply and price parameters for each crop for each year, as well as yield-per-harvested acre, based on calculated new yield-per-planted acre values, as well as on lagged price data, ending stocks from the previous year and other variables. From these results and demand relationships embedded in the model, AGSIM calculated the utilization of each crop (i.e., exports, feed use, other domestic use, etc), as well as change in consumer surplus, net crop income, deficiency payments and other government support payments. Total undiscounted economic surplus was calculated as the crop income plus consumer surplus. For more detail on the AGSIM model see Appendix I of the Environmental TSD (Abt, 2007). The AGSIM model was run for 14 years for each scenario in order for the model parameters to adjust to the initial change in yield. Annual changes in total undiscounted economic surplus were calculated for each of the 14 years. The annual averages for the 14 years are reported in Tables 7-3A-B.

The results from applying the AGSIM model to determine commodity crop and fruit and vegetable benefits based on meeting the level of the current 8-hr average standard and five alternative standards are presented in Tables 7-3A-B. Note that Table 7-3A presents results with the 10% downward hourly adjustment and Table 7-3B presents results without the adjustment. In summary, this analysis estimated a range of benefits using both the available minimum and maximum yield loss equations for each crop. Results are presented in annual 2000 dollars for the commodity crops, fruits and vegetables and total agricultural sector. Overall, benefits from the fruit and vegetable species in this analysis accounted for a relatively large portion of the total agricultural benefits compared with the commodity crops. This is likely because many of the fruits and vegetables are grown in parts of California with high O_3 exposures and any rolling back of air quality produced greater changes in O_3 levels, resulting in higher changes in yield. All of the alternative standards analyzed showed positive incremental benefits greater than those associated with just meeting the level of the current 8-hr average standard. Including a 10% downward adjustment the hourly monitoring did not have a large effect on the overall benefits calculated for each standard. Not surprisingly, not adjusting the hourly monitoring data downward resulted in slightly higher benefits. Meeting the SUM06 of 25 ppm-hr proposed in the last review and the approximate equivalent W126 of 21 ppm-hr produced benefits of approximately $140-$260 million for the total agricultural sector. Of all the scenarios, W126 of 13 ppm-hr, SUM06 of 15 ppm-hr and 8-hr average of 0.07 ppm had the largest economic benefit. Meeting the alternative W126 of 13 ppm-hr and approximate equivalent of SUM06 of 15 ppm-hr

produced benefits of approximately $290-$630 million for the total agricultural sector. It is important to note that these results represent a macro-analysis of the U.S. agricultural economy. Farmers in areas that have higher O_3 levels are more adversely affected than farmers that are in areas with low O_3 levels. These important effects are difficult to quantify in a macro-analysis.

The current CD reports very few new studies conducted on the economic effect of O_3 on U.S. agriculture (U.S. EPA, 2006). A study by Murphy et al. (1999) confirmed the general magnitude of economic effects reported by two key studies performed a decade earlier (Adams, 1986; Adams et al., 1985). Specifically, Murphy et al. (1999) evaluated benefits to eight major crops associated with several scenarios concerning the reduction or elimination of O_3 precursor emissions from motor vehicles in the U.S. Their analysis reported a $2.8 to 5.8 billion (1990 dollars) benefit from complete elimination of O_3 exposures from all sources, i.e., ambient O_3 reduced to a background level assumed to be 0.025 to 0.027 ppm. In comparison, AGSIM calculates $300 million to 2.5 billion (2000 dollars) in economic benefit for 9 major commodity crops when O_3 levels are reduced to near background. These AGSIM results are without any downward adjustment to the O_3 monitoring data and without subtracting out farm payments. With a 10% adjustment and subtracting farm payments, AGSIM calculates substantially lower benefits ($200-800 million) for the same 9 major commodity crops. The Murphy et al. (1999) analysis and the current AGSIM analysis are quite difficult to compare for many reasons: different economic models, different air quality years, different treatment of government farm payment programs, dollar value unadjusted for inflation, different assumptions, etc. However, these comparisons point out that initial assumptions about O_3 exposure and crop payments have large implications when calculating agricultural benefits for reducing O_3 to background levels.

7.6.2.6 Uncertainties In the Crop Risk and Benefit Analyses

The crop risk assessment utilized the C-R relationships developed in OTC experiments performed between 1980 and1988 in the NCLAN program and in other experiments done on fruits and vegetables. As discussed earlier, fruit and vegetable studies were not part of the NCLAN program and C-R functions were available only in terms of a seasonal 7-hr or 12-hr mean index. This mean index form is considered less effective in predicting plant response for a given change in air quality than the cumulative form used with other crops. Two of the uncertainties using the OTC C-R functions in the crop risk assessment are chamber effects (see section 7.6.2.1) and sensitivity of current crops (see section 7.6.2.2). Staff qualitatively addressed these uncertainties citing studies with recent cultivars and studies not using chambers. However, it was not possible to perform a quantitative assessment of these uncertainties. Therefore, despite support in the scientific literature for the magnitude of yield effects of O_3 exposure on crops from OTCs, staff cannot quantify how these uncertainties would affect

estimated crop risks and benefits. Staff notes that the predicted yield losses calculated from the OTC studies represent losses for crops that were not being affected by other stressors. Stressors such as drought may decrease the yield response to O_3 exposure while insect or disease damage to crops may be exacerbated by O_3 exposure.

An additional source of uncertainty not described or accounted for in the last review is that associated with the presence of a decreasing O_3 gradient from the height of the monitor probe down to the canopy heights for most crops. The presence of this gradient makes less certain the predictions of current crop exposures and the associated yield losses based on ambient monitor data. Staff selected a 10% reduction factor to represent the maximum gradient believed to occur for daylight hours. However, recognizing that the actual downward adjustment value varies depending on interactions between numerous plant and site-specific factors, staff chose to present estimates of yield loss for each crop as a range, with non-adjusted and 10%-adjusted air quality as the upper and lower bounds (see section 7.6.2.3 for a detailed discussion).

It is important to restate the uncertainties associated with the results of the AGSIM economic analysis presented in section 7.6.2.5. Uncertainties are introduced by: (1) the interpolation of air quality monitoring data to estimate 2001 national O_3 exposures; (2) the use of C-R functions from OTC studies to estimate relative yield losses from 2001 exposures; (3) the use of a quadratic rollback method to project the "just meet" air quality scenarios without a direct link to an emissions control strategy; and (4) the inherent uncertainties associated with use of an economic model such as AGSIM. It is also important to note that the range of results from this analysis represents impacts associated only with available NCLAN experimental data and a limited number of fruits and vegetable studies. Not all crops have been subjected to exposure-response experiments and effects on those crops would be missed. Despite the amount of uncertainty, staff concludes that this analysis provides useful insights for comparing the relative benefits obtained as a result of meeting alternative regulatory scenarios.

Table 7-3 A-B. Agricultural model results with (A) and without (B) a 10% adjustment of hourly O_3 exposures.

A.

Average Annual Changes in Total Undiscounted Economic Surplus for the Current 8hr Standard (0.08) and Alternative Standards (millions $; 2000)			
Standard	**Commodity Crops**	**Fruits & Vegetables**	**Total Ag.**
0.08 4th-highest	$10 - 20	$60 - 80	$70 - 100
0.07 4th-highest	$50 - 200	$310 - 360	$360 - 560
W126 = 21	$10 - 40	$130 - 140	$140 - 180
W126 =13	$30 - 140	$260 - 300	$290 - 440
SUM06 = 25	$10 - 50	$160 - 180	$170 - 230
SUM06 = 15	$60 - 200	$290 - 330	$350 - 530

B.

Average Annual Changes in Total Undiscounted Economic Surplus for the Current 8hr Standard (0.08) and Alternative Standards (millions $; 2000)			
Standard	**Commodity Crops**	**Fruits & Vegetables**	**Total Ag.**
0.08 4th-highest	$10 - 30	$70 - 80	$80 - 110
0.07 4th-highest	$70 - 280	$350 - 410	$420 - 690
W126 = 21	$20 – 40	$140 - 160	$160 - 200
W126 =13	$60 - 190	$280 - 340	$340 - 530
SUM06 = 25	$20 - 60	$180 - 200	$200 - 260
SUM06 = 15	$70 - 260	$320 - 370	$390 - 630

7.6.3 Tree Risk Assessments

In the last review (U.S. EPA, 1996b), analyses of the effects of O_3 on trees were limited to 11 tree species for which C-R functions for the seedling growth stage had been developed from OTC studies conducted by the National Health and Environmental Effects Research Lab, Western Ecology Division (NHEERL-WED). Figure 7-16A uses the regression equations for each of the 26 studies (49 cases) to graph predicted relative yield loss at various exposure levels in terms of a 12-hr W126. Figures 7-16B-C show composite graphs for the intensively studied quaking aspen and ponderosa pine. Work done since the 1996 review at the AspenFACE site in Wisconsin (see section 7.6.2.1) on quaking aspen has confirmed the detrimental effects of O_3 exposure on tree growth in a field study without chambers (Karnosky et al., 2005). Since the 1996 review, only a few new studies have developed C-R functions for additional tree seedling species (U.S. EPA, 2006). One such study of eastern cottonwood (*Populus deltoides*) saplings was done without chambers or O_3 FACE-type fumigation (Gregg et al., 2003). Eastern cottonwood is a fast growing tree that is important ecologically along streams and commercially for pulpwood, manufacturing furniture and a possible source for energy biomass (Burns and Hankola, 1990). Gregg et al. (2003) found that the cottonwood saplings grown in urban New York City grew faster than saplings grown in more rural areas where O_3 was higher. The secondary nature of the reactions of O_3 formation and NOx titration reactions within the city center resulted in significantly higher cumulative O_3 exposures in the rural sites. After carefully considering many factors, they concluded the major explanation for the difference in growth was the gradient of O_3 exposure between urban and rural sites. This explanation was also confirmed with an OTC study (Gregg et al., 2003). Figure 7-17 shows the biomass growth of cottonwood plotted against the monitored 12-hr W126 at the sites the trees were planted (Gregg et al., 2003). Staff notes that the responses of natural populations of cottonwood to O_3 may vary because of precipitation patterns and differences in native soils. The Gregg et al. (2003) study is important because it demonstrated that growth effects of O_3 exposure could be documented in field without chambers or fumigation and the growth decreases were as great as to seen in previous OTC studies. The evidence from the AspenFACE results and Gregg et al. (2003) provide support for the continued use of NHEER-WED OTC studies to estimate risk to seedlings in the U.S. Section 7.6.3.1 describes how staff updated the tree seedling risk analysis performed in the last review.

Section 7.6.3.2 discusses the approach for assessing O_3 effects on vegetation in natural settings using visible foliar injury data. Section 7.6.3.3 discusses the analysis and results for modeling O_3 impacts on mature trees in the Eastern and Western U.S. The tree and/or forest analyses outlined below will enable staff to begin to assess important long-term effects of various secondary standard levels on forest ecosystem health and services.

Figure 7-16. Median tree seedling biomass loss for all 49 cases (A), quaking aspen (B), and ponderosa pine (C) characterized with the 12-hr W126

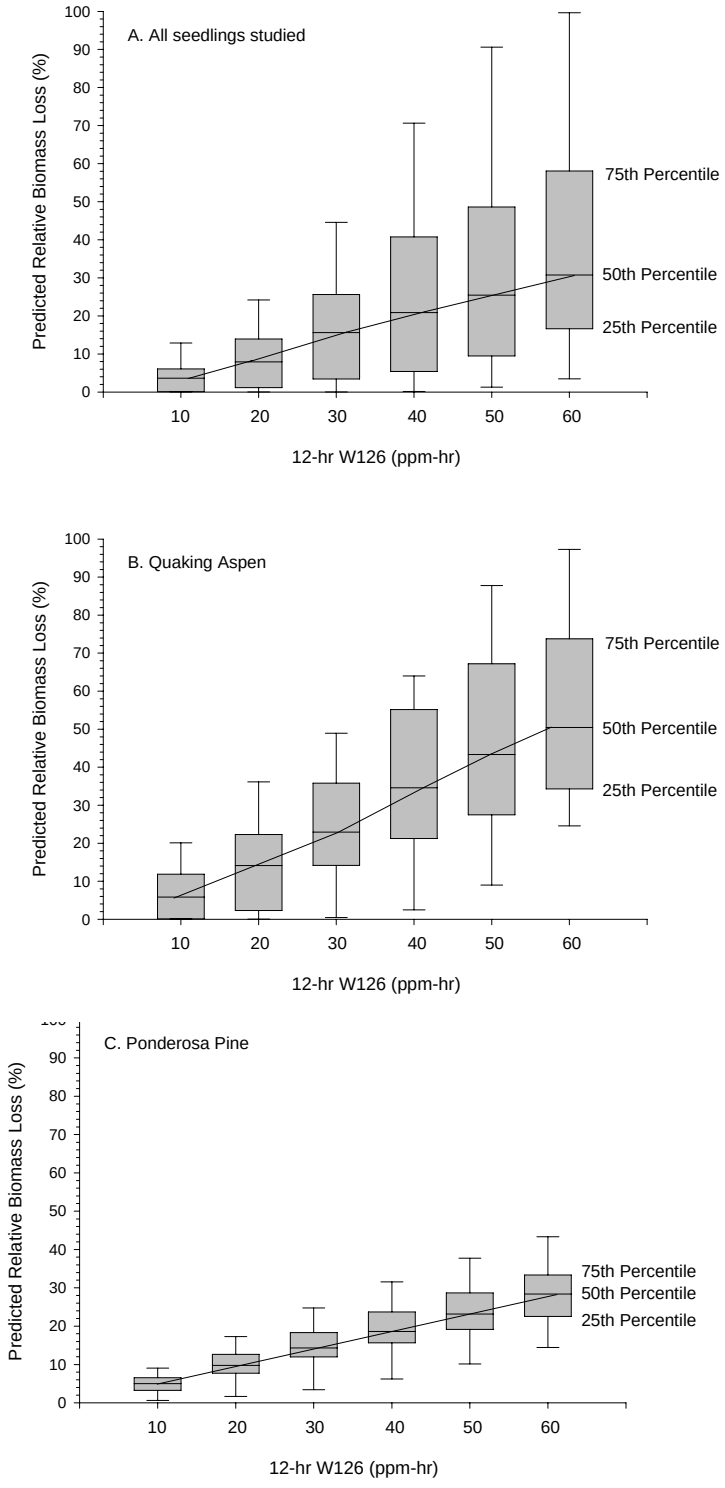
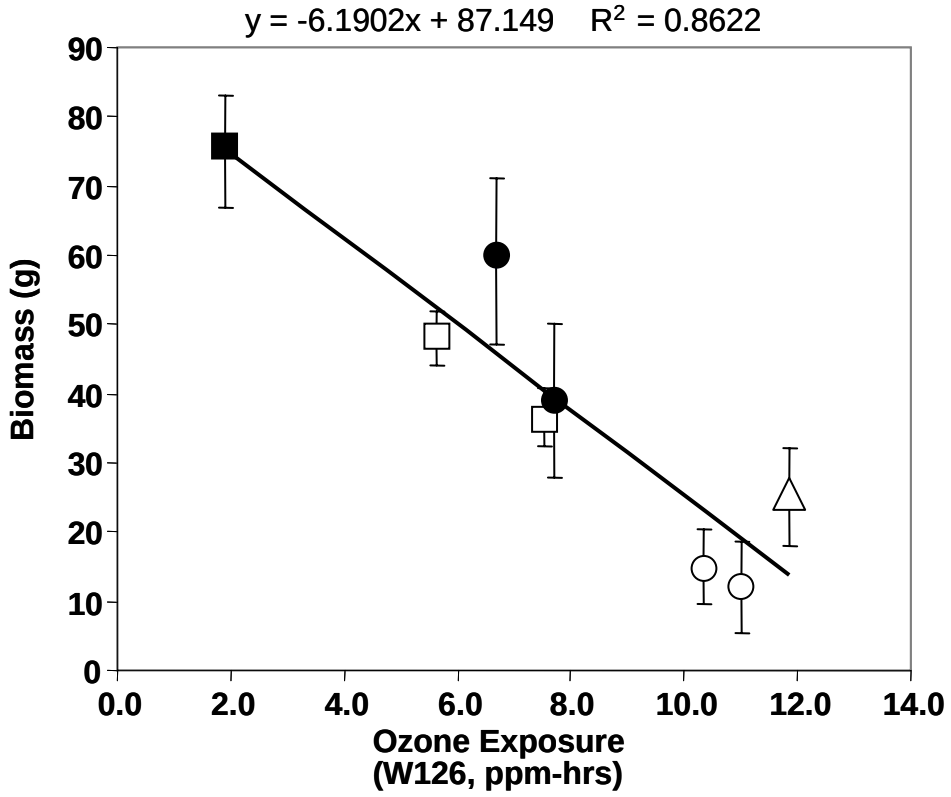


Figure 7-17. Cottonwood (*Populus deltoides*) shoot biomass (mean ± s.e.) at urban (filled) and rural (open) sites in the vicinity of New York City versus ambient O_3 exposure (growing period 12-hr W126, July 7 – Sept. 20). Squares, circles and triangles represent responses in 1992, 1993 and 1994, respectively. Cottonwood saplings were grown in potting soil under well watered conditions. (Modified from Gregg et al., 2003)

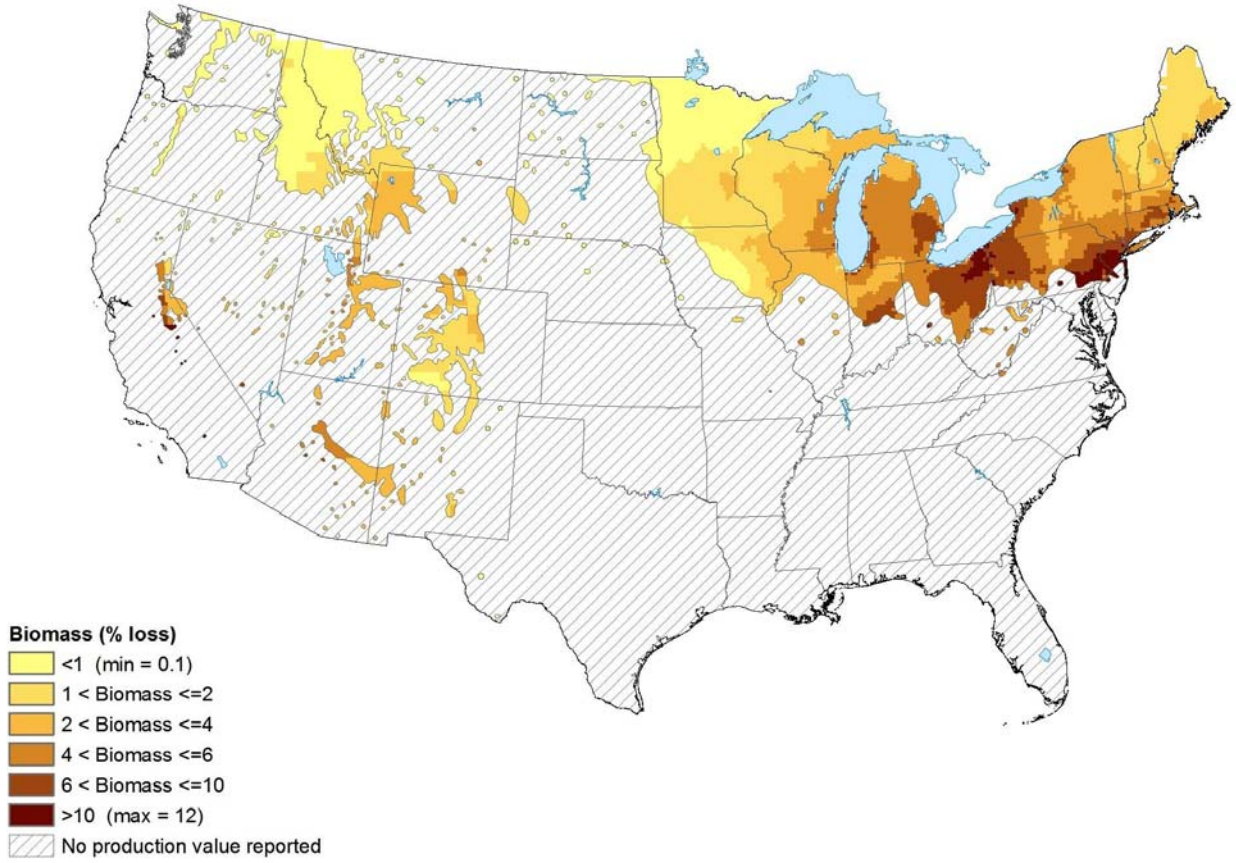


7.6.3.1 Quantifiable Risk of Biomass Loss In Select Tree Seedling Species

In a process similar to that used for crops above (7.6.2.4), C-R functions for biomass loss for a subset of seedling tree species taken from the CD (Table 7F-3) and information on tree growing regions derived from the U.S. Department of Agriculture's Atlas of United States Trees (Little, 1971) were combined with projections of air quality based on 2001 POES, to produce estimated biomass loss for each of the seedling tree species individually. Some of the results for the highest areas of risk to tree seedlings are presented in Table 7F-5 in Appendix 7F. In addition, maps depicting these results for selected tree seedling species are found in Appendix 7H.

Figure 7-18 shows an example of the quaking aspen seedling biomass loss with an hourly O_3 exposures adjusted down by 10%. Figure H-1 in Appendix H shows the quaking aspen without the 10% hourly adjustment. The quaking aspen maps show significant variability in projected seedling biomass loss across its range for 2001. Quaking aspen seedling biomass loss (with the10% adjustment) was projected to be greater than 4% over much of its geographic range, though it can reach as high as 12% in some areas. In Appendix 7H, there are additional maps of ponderosa pine and black cherry along with maps of seedling biomass loss with and without a 10% adjustment of the monitoring data. Further, in Chapter 5 of the Environmental TSD, a series of maps are presented showing seedling biomass gain when various standard levels are met. These biomass gain maps indicate that substantial improvements in seedling growth may be achieved when the alternative standards are met, especially the 0.07 ppm 4^{th}-highest max., SUM06 of 15 ppm-hr and W126 of 13 ppm-hr. It should be noted that the species mapped are generally sensitive and they are also important tree species in ecosystems across vast areas of the U.S. Though each map shows the geographical range for a species, it does not indicate that an individual of that species will be found at every point within its range. It should also be recognized that the production of these maps incorporates several separate sources of uncertainty, beginning with the C-R functions produced for seedlings in OTCs to the uncertainties associated with the inputs used to generate the POES. Furthermore, percent biomass loss in tree seedlings is not intended to be a surrogate for expected biomass loss in mature trees of the same species (see section 7.6.3.3 for modeling of mature tree growth). Studies indicate that mature trees can be more or less sensitive than seedlings depending on the species. Further, seedling biomass loss cannot be considered comparable to percent yield loss in agricultural crops. This is because a small biomass loss per year in a perennial tree species, if compounded over multiple years of exposure could have a large effect on the growth of that tree, while yield loss in annual crops is only affected by the O_3 exposure for that year. In summary,

Figure 7-18. Estimated aspen seedling annual biomass loss based on interpolated 2001 maximum 3-month 12-hr W126 with a 10% downward adjustment of hourly O_3 concentrations. This map indicates the geographic range for quaking aspen (*Populus tremoloides)*, but it does not necessarily indicate that aspen will be found at every point within its range.



this analysis indicates that current air quality can produce significant seedling biomass loss in the areas which those trees grow. Meeting the level of alternative standards is expected to improve biomass growth in the seedlings analyzed.

7.6.3.2 Visible Foliar Injury Incidence

The use of sensitive plants as biological indicators to detect phytotoxic levels of O_3 is a longstanding and effective methodology (Chappelka and Samuelson, 1998; Manning and Krupa, 1992). Some well defined bioindicators for ambient O_3 include blackberry, black cherry, green ash, milkweed, quaking aspen, sassafras, yellow poplar, and white ash. Each of these bioindicators exhibits typical O_3 injury symptoms when exposed under appropriate conditions. These symptoms are considered diagnostic as they have been verified in exposure-response studies under experimental conditions. Typical visible injury symptoms on broad leaved plants include: 1) acute exposure (pigmented lesions (stippling), flecking, surface bleaching, and/or bifacial necrosis); 2) chronic exposure (pigmentation (bronzing), chlorosis or premature senescence). Typical visible injury symptoms for conifers include: 1) chlorotic banding or tipburn (acute exposure); 2) flecking or chlorotic mottling, premature senescence of needles (chronic exposure). Though common patterns of injury develop within a species, these foliar lesions can vary considerably between and within taxonomic groups. Furthermore, the degree and extent of visible foliar injury development varies from year to year and site to site, even among co-members of a population exposed to similar O_3 levels, due to the influence of co-occurring environmental and genetic factors. It is important to note that the visible foliar injury occurs only when sensitive plants are exposed to elevated O_3 concentrations in a predisposing environment. Thus, great care must be taken when assessing the response of bioindicators to ambient O_3 (Flagler, 1998).

The Unites States Forest Service (USFS) through the Forest Health Monitoring Program (FHM) (1990 - 2001) and currently the Forest Inventory and Analysis (FIA) Program has been collecting data regarding the incidence and severity of visible foliar injury on a variety of O_3 sensitive plant species throughout the U.S. (Coulston et al., 2003, 2004; Smith et al., 2003). The plots where these data are taken are known as biosites. These biosites are located throughout the country and analysis of visible foliar injury within these sites follows a set of established protocols (for more details see http://fiaozone.net/). Since the conclusion of the 1996 NAAQS review, the FIA monitoring program network and database has continued to expand. The visible foliar injury indicator has been identified as a means to track O_3 exposure stress trends in the nation's natural plant communities as highlighted in EPA's most recent Report on the Environment (U.S. EPA, 2003a; http://www.epa.gov/indicators/roe). EPA staff also considers it important to assess the degree to which O_3-induced visible foliar injury observed *in situ*,

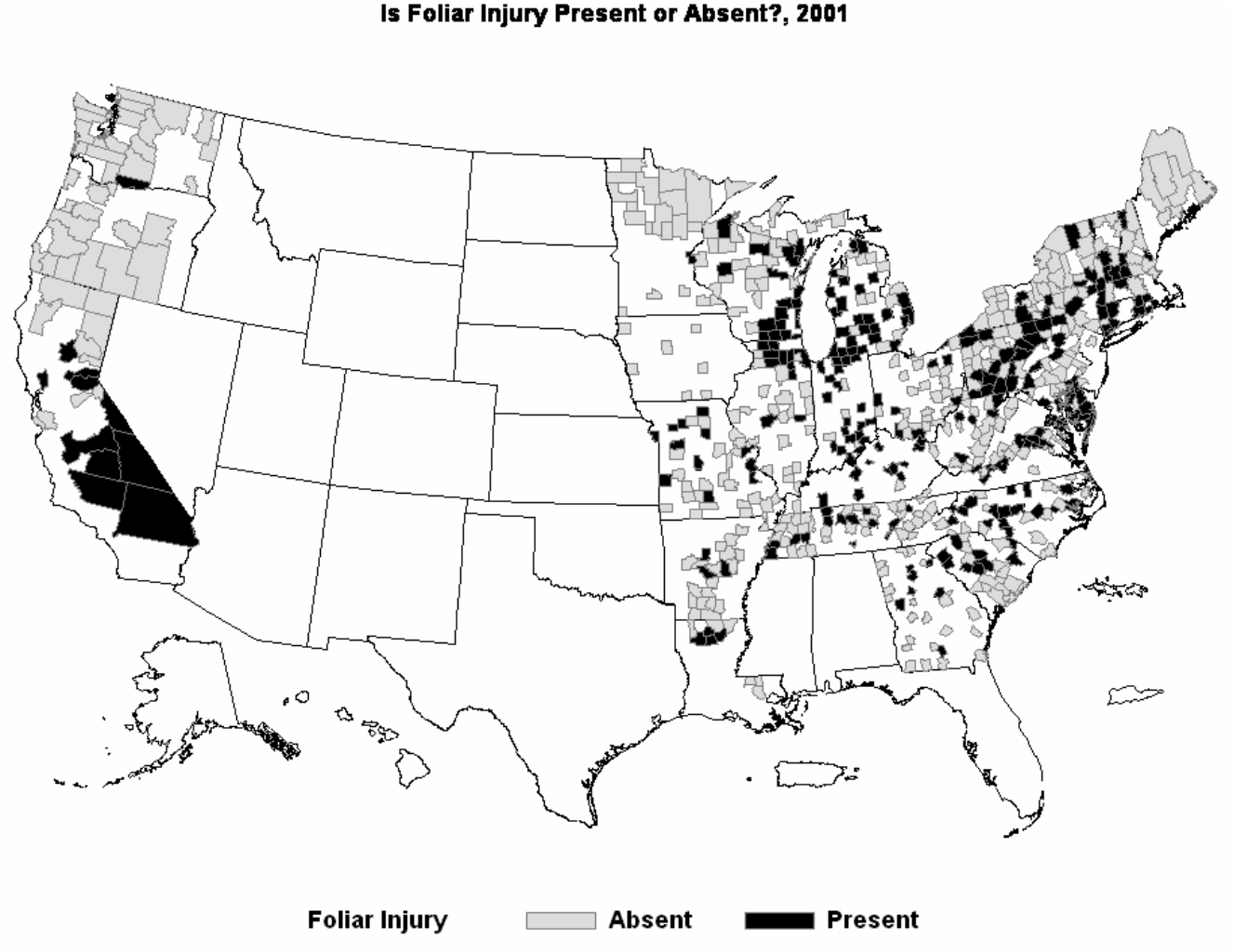
corresponds with monitored O_3 air quality in recent years. In a collaborative effort with FIA staff, EPA staff conducted an analysis to compare the incidence of visible foliar injury at different levels of air quality (e.g., the current standard and alternative levels under consideration) by county throughout the U.S. This analysis potentially provides a measure of the effectiveness and degree of protection provided by the current form/level of the secondary standard for this welfare effect.

The major confounding effect for O_3 induced visible foliar injury is the amount of soil moisture (local rainfall) available to a plant during the year that the visible foliar injury is being assessed. This is because lack of soil moisture decreases stomatal conductance of plants and, therefore, limits the amount of O_3 entering the leaf that can cause injury. Many researchers have shown that dry periods in local areas tend to decrease the incidence and severity of visible foliar injury caused by O_3 in plants measured by the USFS (Smith et al., 2002). Therefore, the incidence of visible foliar injury is not always higher in years with higher O_3, especially when there is drought in areas where visible foliar injury is assessed.

Due to a congressional requirement that the USFS protect landowner privacy, FIA cannot publicize the exact locations of the biosites. As a result, all data in this analysis are reported on a county-level. County-level visible foliar injury data were available for the years 2001 to 2004 for all areas of the U.S. except the Mountain West region. However, according to the FIA staff, no O_3 injury was reported at any site in that region. Figure 7-19, shows that the incidence of visible foliar injury in 2001 was widespread across the eastern and western U.S. The 2001 data are indicative of the incidence of visible foliar injury in the years 2001 to 2004. (see Appendix 7I for 2002). This indicates that O_3 levels are above phytotoxic levels sufficient to cause adverse effects in natural plant populations in many areas. It is important to note that direct links between O_3 induced visible foliar injury symptoms and other adverse effects (e.g., biomass loss), are not always found. However, in some cases, visible foliar symptoms have been correlated with decreased vegetative growth (Karnosky et al., 1996; Peterson et al., 1987; Somers et al., 1998) and with impaired reproductive function (Black et al., 2000; Chappelka, 2002). Though visible injury is a valuable indicator of the presence of phytotoxic concentrations of O_3 in ambient air it is not always a reliable indicator of damage or other injury endpoints. The lack of visible injury does not indicate a lack of phytotoxic concentrations of O_3 or a lack of non-visible O_3 effects.

In an attempt to assess how meeting various O_3 standard levels affected the incidence of visible foliar injury, staff matched up county-level O_3 monitoring data with counties that had US Forest Service biosites. In counties containing multiple O_3 monitors, staff used the monitor measuring the highest O_3 to characterize county air quality. Because visible foliar injury symptoms reflect the O_3 stress of the year in which they are observed, staff looked at yearly

Figure 7-19. 2001 County-level incidence of visible foliar injury in the eastern and western U.S. as measured by the US Forest Service FIA program.



snapshots of county-level air quality data. Between 235 and 286 counties had EPA O_3 monitoring and at least one USFS FIA biosite surveyed for visible foliar injury in the years 2001 – 2004 (see Table 7-4). However, because the specific locations of the USFS biosite are not publicly available, staff was unable to determine how close the biosites within each county were to the O_3 monitor selected to represent that county. Air quality was evaluated in terms of both the current 8-hr. average and 12-hr SUM06 forms, using a number of different levels. Table 7-4 shows the percentage and number of counties with and without visible foliar injury at or below various standard levels for the 2001-2004 period. Because the FIA program reorganized the locations of biosites in 2002 and expanded the number of biosites in 2003 and 2004, the total number of counties containing both an O_3 monitor and an FIA biosite changed each year and it is difficult to interpret changes in the number of counties in different categories between years. Therefore, staff found it more informative to present results in terms of percent of total counties with or without injury under different levels of air quality. First, this table illustrates that visible foliar injury is occurring in areas that are meeting the current 8-hr average O_3 standard (0.084 ppm). Second, the table illustrates that the secondary standard option of a SUM06 of 25 ppm-hr proposed in 1996 did not appear to offer more protection from visible foliar injury than the current 8-hr average standard form. By comparison, the SUM06 of 15 ppm-hr and the 8-hr average of 0.074 ppm provided more protection across all years than either the 0.084 ppm 8-hr average or SUM06 of 25 ppm-hr standards. At the 0.084 ppm, 8-hr average, the percent of counties showing injury ranged from 21% to 39%. Under a SUM06 of 25 ppm-hr, the percent of counties with injury was 26% to 49%. For the two lower air quality alternatives (0.074 ppm 8-hr average and SUM06 of 15 ppm-hr), values ranged from 12% injured to 30% and 35%, respectively.

In summary, this analysis indicates that incidence of O_3 induced visible foliar injury is widespread across the eastern and western U.S. Visible foliar injury was observed in counties that are meeting the current level of the 8-hr standard and an alternative secondary standard option of a SUM06 of 25 ppm-hr proposed in 1996. Lower standards in the 8-hr average and SUM06 forms would be expected to have lower incidences of visible foliar injury. However, the level of protection would depend heavily on local environmental variable such as soil moisture. Finally, in the consensus workshop held on the secondary O_3 standard, researchers were in agreement that a 3 month 12-hr SUM06 value of 8 to 12 ppm-hr should be considered for protection from visible foliar injury to natural ecosystems (Heck and Cowling, 1997). The analysis above supports this recommendation that these levels would reduce the incidence of visible foliar injury to natural ecosystems.

Table 7-4. Percentage and number of counties with visible foliar injury (injured) and without injury (not injured) below various standard levels for the years 2001-2004. Each county had an O_3 monitor and a USDA forest service FIA plot tracking visible foliar injury due to O_3 exposure.

Year		≤0.084* (ppm)	≤0.074* (ppm)	≤SUM06 25 (ppm-hr)	≤SUM06 15 (ppm-hr)	Total Counties with O_3 monitoring & FIA biosites
2001	# of counties	99	36	134	48	235
	injured	39% (39)	25% (9)	49% (65)	23% (11)	
	not injured	61% (60)	75% (27)	51% (69)	77% (37)	
2002	# of counties	89	43	129	59	270
	injured	21% (19)	12% (5)	26% (33)	12% (7)	
	not injured	79% (70)	88% (38)	74% (96)	88% (52)	
2003	# of counties	185	61	236	135	285
	injured	28% (52)	11% (7)	34% (81)	25% (34)	
	not injured	72% (133)	89% (54)	66% (155)	75% (101)	
2004	# of counties	260	159	249	220	286
	injured	35% (91)	30% (47)	37% (91)	35% (76)	
	not injured	65% (169)	70% (112)	63% (158)	65% (144)	

*These standard levels represent the annual 4th-highest 8-hr maximum average

7.6.3.3 Modeled Mature Tree Growth Response: Eastern and Western Species Case Studies

In the 1996 O_3 Staff Paper, evaluations of O_3 impacts on tree growth were limited to the seedling growth stage (U.S. EPA, 1996b). At that time, robust C-R functions were available only for 11 tree seedlings developed from OTC data. Few studies had been done comparing seedling sensitivity to that of a mature tree of the same species. Recent experiments using the FACE methodology have been able to expose 3 tree species to O_3 beyond the seedling growth stage. However, this methodology has not yielded C-R functions at this time, due to the limited number of exposure regimes used. Findings from FACE publications, however, do show decreased biomass growth under elevated O_3 in trees beyond the seedling stage (King et al., 2005). In order to better characterize the potential O_3 effects on mature tree growth, staff used a tree growth model (TREGRO) as a tool to evaluate the effect of changing O_3 air quality scenarios from just meeting alternative O_3 standards on the growth of mature trees.

TREGRO is a process-based, individual tree growth simulation model (Weinstein et al, 1991) and has been used to evaluate the effects of a variety of O_3 scenarios and linked with concurrent climate data to account for O_3 and climate/meteorology interactions on several species of trees in different regions of the U.S. (Tingey et al., 2001; Weinstein et al., 1991; Retzlaff et al., 2000; Laurence et al., 1993; Laurence et al., 2001; Weinstein et al., 2005). The model provides an analytical framework that accounts for the nonlinear relationship between O_3 exposure and response. The interactions between O_3 exposure, precipitation and temperature are integrated as they affect vegetation, thus providing an internal consistency for comparing effects in trees under different exposure scenarios and climatic conditions (see the Environmental Assessment TSD for more details on TREGRO). An earlier assessment of the effectiveness of national ambient air quality standards in place since the early 1970s took advantage of 40 years of air quality and climate data for the Crestline site in the San Bernardino Mountains of California to simulate Ponderosa pine growth over time with the improving air quality using TREGRO (Tingey et al., 2004).

Staff collaborated with the EPA NHEERL-WED laboratory to use the TREGRO model to assess growth of Ponderosa pine (*Pinus ponderosa*) in the San Bernardino Mountains of California (Crestline) and the growth of yellow poplar (*Liriodendron tulipifera*) and red maple (*Acer rubrum*) in the Appalachian mountains of Virginia and North Carolina, Shenandoah National Park (Big Meadows) and Linville Gorge Wilderness Area (Cranberry), respectively. Total tree growth associated with 'as is' air quality, and air quality adjusted to just meet alternative O_3 standards was assessed (Table 7-5). Ponderosa pine is one of the most widely distributed pines in western North America, a major source of timber, important as wildlife

Table 7-5. Relative increase in total annual tree biomass growth, simulated with the TREGRO model, if the level of the current (0.08 ppm) and alternative standards are met.

Species	red maple	red maple	yellow poplar	yellow poplar	ponderosa pine
Site	Big Meadows, VA (1993-1995)	Cranberry, NC (1993-1995)	Big Meadows, VA (1993-1995)	Cranberry, NC (1993-1995)	Crestline, CA (1995-2000)
0.08 4th-highest	0.41%	*no rollback*[1]	0.03%	*no rollback*[1]	8.63%
0.07 1st-highest	2.71%	2.31%	0.38%	6.54%	10.81%
0.07 4th-highest	2.24%	1.38%	0.34%	3.91%	*n.a.*[2]
SUM06 = 25[3]	0.34%	*no rollback*[1]	0.07%	*no rollback*[1]	10.33%
SUM06 = 15[4]	4.49%	2.99%	0.60%	8.26%	*n.a.*[2]

[1]A rollback was not necessary for the Cranberry site for the 0.08 ppm 4th-highest and SUM06 = 25 ppm-hr scenarios since air quality was at or below the levels of those scenarios.

[2] TREGRO was not run for ponderosa pine for the 0.07 ppm 4th-highest scenario.

[3]The roll-back to a SUM06 of 25 ppm-hr was a W126 of approximately 18 ppm-hr at Cranberry, Big Meadows and Crestline.

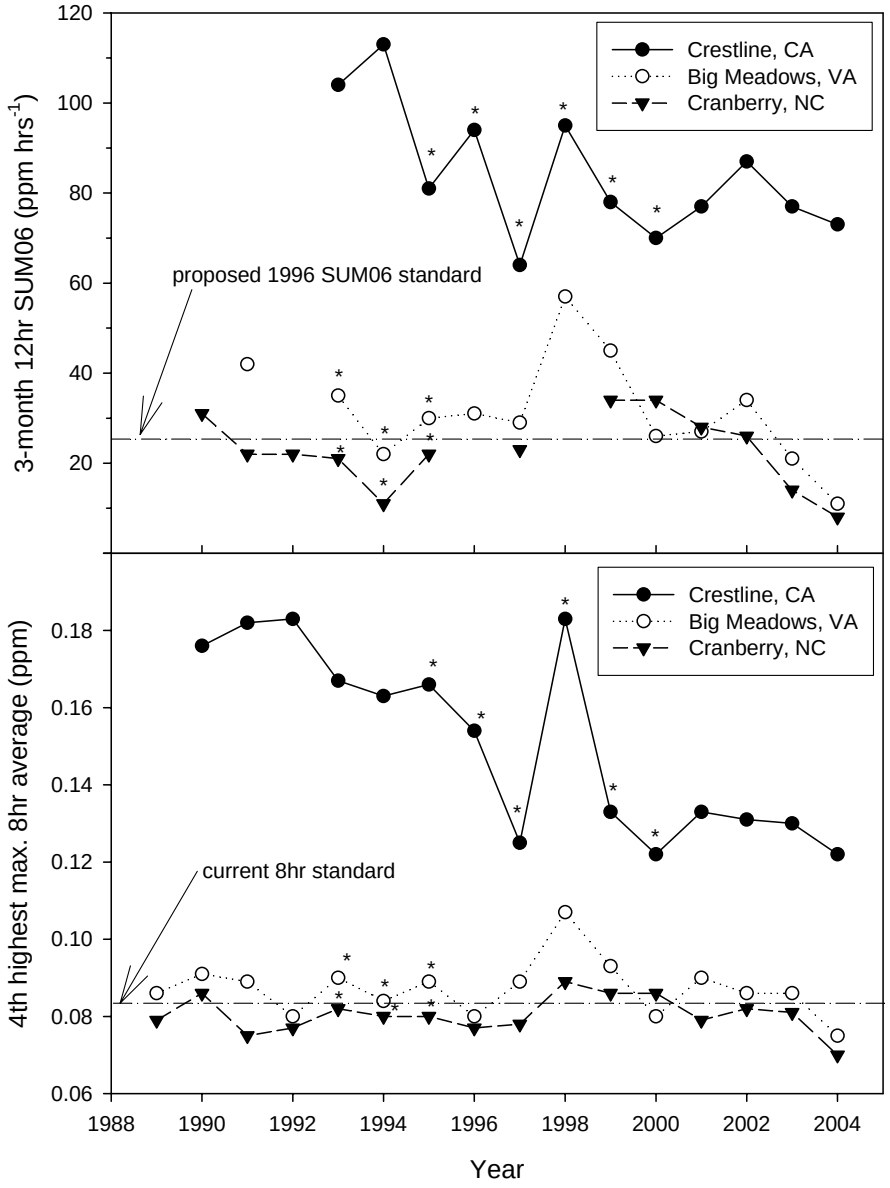
[4]The roll-back to a SUM06 of 15 ppm-hr was a W126 of approximately 13 ppm-hr at Cranberry and Big Meadows

habitat, and valued for aesthetics (Burns and Honkala, 1990). Red maple is one of the most abundant species in the eastern U.S. and is important for its brilliant fall foliage and highly desirable wildlife browse food (Burns and Honkala, 1990). Yellow poplar is an abundant species in the southern Appalachian forest. It is 10% of the cove hardwood stands in southern Appalachians which are widely viewed as some of the country's most treasured forests because the protected, rich, moist set of conditions permit trees to grow the largest in the eastern U.S. The wood has high commercial value because of its versatility and as a substitute for increasingly scarce softwoods in furniture and framing construction. Yellow poplar is also valued as a honey tree, a source of wildlife food, and a shade tree for large areas (Burns and Honkala, 1990).

At the western site, staff and NHEERL-WED scientists used Crestline, CA air quality and climate data from the years 1995 to 2000 to run TREGRO, while at the eastern sites, staff used Big Meadows, VA and Cranberry, NC air quality and climate data from the years 1993 to 1995. These three years were the only years in the east with readily available O_3 and climate data that could be used in TREGRO. The years chosen to run the TREGRO at each site appear to have annual O_3 exposures typical of the last 15 years (Figure 7-20). Air quality from each site and year was adjusted using the quadratic roll-back method to 'just meet' the current 8-hr average secondary standard (0.084 ppm), a 12-hr SUM06 of 25 ppm-hr, and 1st highest max average of 0.07 ppm. Staff also tested the 4th-highest 0.07 ppm level on the Cranberry and Big Meadows sites. For the ponderosa pine at Crestline, TREGRO was run for "as is" and "just meet" air quality conditions in four 3 year increments to increase the accountability of climate variability and the annual average biomass determined from these 4 simulations to yield an annual average biomass change over the 6 years of O_3 exposure. For the yellow poplar and red maple, two sites (Big Meadows, VA and Cranberry, NC) were chosen to run TREGRO to increase the variability in climate since there were only 3 years of data available at each site. The differences between growth under "just meet" and "as is" air quality conditions were compared to evaluate the effectiveness of the current secondary standard and alternative standards in protecting these three tree species.

Results of the TREGRO simulations are presented in Table 7-5. Clearly, the greatest simulated growth benefits in the scenarios are seen in ponderosa pine at the Crestline site in California. As shown in Figure 7-20, O_3 levels are much higher at Crestline than the sites in the eastern US. Meeting the level of the current standard was simulated to result in an 8.63% increase annual growth and a SUM06 of 25 ppm-hr is expected increase growth 10.33% in ponderosa pine. In the eastern sites (Cranberry and Big Meadows), O_3 levels are much lower (Figure 7-20) and had less of an effect on the simulated growth of red maple and yellow poplar. In fact, the Cranberry, NC site was below the level of the current 8-hr average standard and the

Figure 7-20. Historical O_3 data as measured in the 3-month 12-hr SUM06 and 4th highest 8-hr metrics for the 3 sites used to run the TREGRO model. For Big Meadows, VA and Cranberry, NC, climate and O_3 data from 1993 to 1995 was used to run TREGRO and for Crestline, CA, 1995 to 2000 was used. Missing data points in the top panel indicate incomplete data to calculate a SUM06. * indicates which years of data were used in the TREGRO model at each site.



SUM06 of 25 ppm-hr scenarios and, therefore, no benefits were calculated for those levels. At Big Meadows, VA, the current 8-hr average standard and SUM06 scenarios resulted in relatively small growth increases for yellow poplar (0.03-0.07%) and red maple (0.34-0.41%). This was mostly because the Big Meadows site was close to meeting those levels in 1993-1995 (Figure 7-20). Red maple was simulated to have a similar response (~2%) to the 0.07 ppm 1st and 4th-highest 8-hr max in Big Meadows and Cranberry. For the same scenarios, yellow poplar had a very different response to O_3 reduction at Big Meadows (0.34-0.38%) compared to Cranberry (3.91-6.54%). The climate at Cranberry is much more ideal for yellow poplar than under the cool temperatures of Big Meadows, making it much more likely that its growth would be suppressed by O_3 and that, conversely, it would respond much more to O_3 reductions. Red maple has a much larger geographical distribution, so that the temperature differences between Big Meadows and Cranberry are less likely to affect the growth response. This phenomenon was reflected in the simulations.

The effect of O_3 on an individual tree may be quite different than the predicted effect on a forest stand after many years. Some researchers have used the ZELIG model, a forest stand simulator, to predict stand growth using growth rates of individual species from TREGRO scenarios (Laurence et al., 2001; Weinstein et al., 2005). Small changes in growth of an individual tree over a short period of time have sometimes been simulated to have large changes in basal area as it develops over a long time period. For example, Weinstein et al. (2005) found a simulated O_3 effect on an individual ponderosa pine at Crestline to reduce growth by 6.7% in three years under normal precipitation, yet stand basal area was calculated to be reduced by 29% after 100 years. Similarly, Laurence et al. (2001) found individual yellow poplar in NC with an O_3 induced growth loss of 1.7% which was then calculated to reduce basal area by 14% after 100 years. This suggests that small effects on individual tree growth may result in substantial effects on forest stand growth after many years.

7.6.3.4 Uncertainties In the Tree Risk Analyses

It should be recognized the seedling risk assessment incorporates several sources of uncertainty that have been previously discussed. Specifically, uncertainties associated with the development C-R functions using OTCs and uncertainties associated with the inputs used to generate the POES (see sections 7.6.2.1 and 7.5.5). As with crops, the potential differences between exposures measured above seedling canopies and actual exposure at the top of the canopy is an important uncertainty. As explained in section 7.6.2.3, it is impossible to fully account for these potential differences throughout the U.S. Therefore, staff calculated risks using a 10% adjustment of hourly exposures and no adjustment of hourly exposures. These

calculations provide a bracket of responses within which the reality probably lies for the actual O_3 exposures tree seedling canopies.

The visible foliar injury risk assessment contains several sources of uncertainty. First, due to the major confounding effect of soil moisture (local rainfall) in determining the level of observed symptom expression, the incidence and degree of visible foliar injury is not always higher in years with higher O_3, especially when there is drought in areas where foliar injury is assessed. Second, the lack of visible injury does not indicate a lack of phytotoxic concentrations of O_3 or a lack of non-visible O_3-induced effects, since it is not always a reliable indicator of other O_3-related injury and damage endpoints. Finally, due to the change in FIA protocols in 2002 and the unavailability of specific biosite locations, staff was unable to determine the degree to which county level monitored O_3 values reflect the actual O_3 exposure conditions at the biosites within those counties.

As with every model, TREGRO has many known and unknown sources of uncertainty. Because TREGRO only models individual trees, the effects of competition are not factored in. There is genetic variability within species so that the values produced for an individual tree may not reflect the variability within the species as a whole. Only a few species have been parameterized in TREGRO. Due to the limited number of species tested and included in this assessment, it is unclear to what degree these results apply to O_3 impacts on mature trees in general. For further discussion of uncertainties see Appendix J in the Environmental Assessment TSD (Abt, 2007).

7.7 QUALITATIVE RISK: ECOSYSTEM CONDITION, FUNCTION AND SERVICES

Ecosystems are comprised of complex assemblages of organisms that provide distinct ecological attributes, many of which may be adversely affected by O_3 (U.S. EPA, 2006). A new effort has been initiated within the Agency to identify indicators of ecological condition whose responses can be clearly linked to changes in air quality and be used to track improvements in environmental protection attributable to environmental program actions/implementation. Moreover, a recent critique of the secondary NAAQS review process published in the report by the National Academy of Sciences on Air Quality Management in the United States (NRC, 2004) stated that "EPA's current practice for setting secondary standards for most criteria pollutants does not appear to be sufficiently protective of sensitive crops and ecosystems..." This report made several specific recommendations for improving the secondary NAAQS process and concluded that "There is growing evidence that tighter standards to protect sensitive ecosystems in the United States are needed..." However, the vast majority of information regarding the effects of O_3 involves the sensitivity of individual species. Therefore, this section lays out some

examples of our current understanding of how O_3 may be affecting ecosystems and identifies areas of research needed to address this issue.

An ecosystem is defined as comprising all of the organisms in a given area interacting with the physical environment, so that a flow of energy leads to a clearly defined trophic structure, biotic diversity, and cycling of materials between living and nonliving parts (Odum, 1963). Individuals within a species and populations of species are the building blocks from which communities and ecosystems are constructed. Classes of natural ecosystems, e.g., tundra, wetland, deciduous forest, and conifer forest, are distinguished by their dominant vegetation forms. Ecosystem boundaries are delineated when an integral unit is formed by their physical and biological parts. Defined pathways for material transport and cycling and for the flow of energy are contained within a given integrated unit.

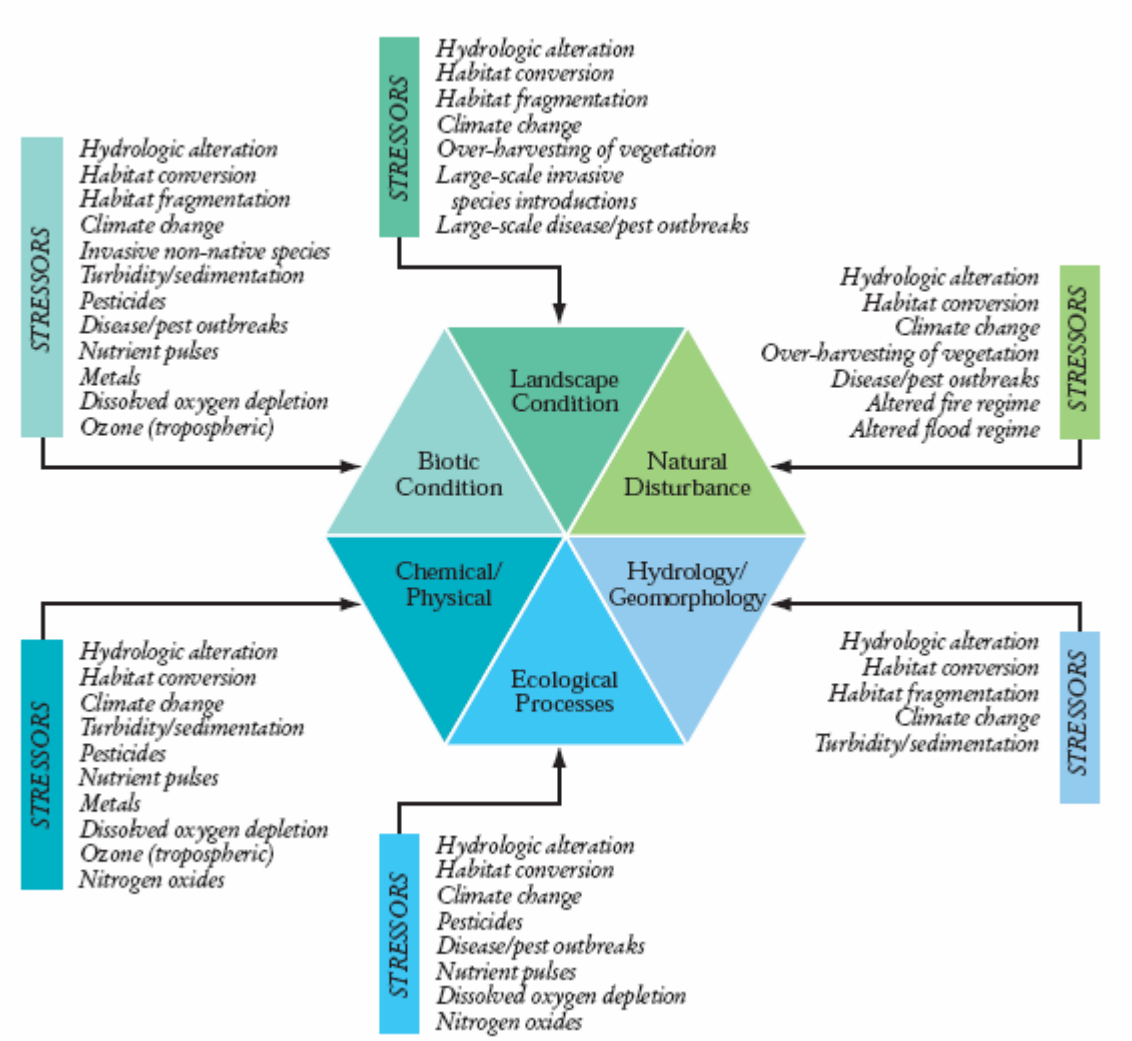
Each level of organization within an ecosystem has functional and structural characteristics. At the ecosystem level, functional characteristics include, but are not limited to, energy flow; nutrient, hydrologic, and biogeochemical cycling; and maintenance of food chains. The sum of the functions carried out by ecosystem components provides many benefits to humankind, as in the case of forest ecosystems (Smith, 1992). Some of these benefits include food, fiber production, aesthetics, genetic diversity, and energy exchange.

A conceptual framework for discussing the effects of O_3 on ecosystems was developed by the EPA Science Advisory Board (Young and Sanzone, 2002). Their six essential ecological attributes (EEAs) include landscape condition, biotic condition, organism condition, ecological processes, hydrological and geomorphological processes, and natural disturbance regimes. Figure 7-21 outlines how common anthropogenic stressors, including tropospheric O_3, might affect the essential ecological attributes outlined by the SAB.

There is evidence that tropospheric O_3 is an important stressor of ecosystems, with documented impacts on the biotic condition, ecological processes, and chemical/physical nature of natural ecosystems (U.S. EPA, 2006). Most of the effects on ecosystems must be inferred from O_3 exposure to individual plants and processes that are scaled up through the ecosystem affecting processes such as energy and material flow, inter- and intraspecies competition, and net primary productivity (NPP). Thus, effects on individual keystone species and their associated microflora and fauna, which have been shown experimentally, may cascade through the ecosystem to the landscape level. By affecting water balance, cold hardiness, tolerance to wind and by predisposing plants to insect and disease pests, O_3 may even impact the occurrence and impact of natural disturbance (e.g., fire, erosion).

Another approach to assessing O_3 effects on ecosystems is the identification and use of indicators. For example, the main indicators of phytotoxic O_3 exposures used for forest ecosystems are visible foliar injury (as described in section 7.6.3.2 above) and radial growth of

Figure 7-21. Common anthropogenic stressors and the essential ecological attributes they affect. Modified from Young and Sanzone (2002)



trees. Systematic injury surveys demonstrate that foliar injury occurs on O_3-sensitive species in many regions of the United States. However, there is not always a direct relationship between the severity of the visible foliar symptoms and growth. This essentially means it is difficult to quantify or characterize the degree which EEAs may be impacted when visible foliar injury is found in the field. Investigations of the relationship between changes in radial growth of mature trees and ambient O_3, in combination with data from many controlled studies with seedlings, suggest that ambient O_3 is reducing the growth of mature trees in some locations. However, definitively attributing growth losses in the field to O_3 in a wide array of ecosystems is often difficult because of confounding factors with other pollutants, climate, insect damage and disease.

The CD (U.S. EPA, 2006) outlines seven case studies where O_3 effects on ecosystems have either been documented or are suspected. However, in most cases, only a few components in each of these ecosystems have been examined and characterized for O_3 effects and, therefore, the full extent of ecosystem changes in these example ecosystems is not fully understood. Clearly, there is a need for highly integrated ecosystem studies that specifically investigate the effect of O_3 on ecosystem structure and function in order to fully determine the extent to which O_3 is altering ecosystem services.

7.7.1 Evidence of Potential Ozone Alteration of Ecosystem Structure and Function

The seven case studies listed in the 2006 CD demonstrate the potential for O_3 to alter ecosystem structure and function (U.S. EPA, 2006). The oldest and clearest example involves the San Bernardino Mountain forest ecosystem. In this example, O_3 appeared to be a predisposing factor leading to increased drought stress, windthrow, root diseases, and insect infestation (Takemoto et al., 2001). Increased mortality of susceptible tree species, including ponderosa and Jeffrey pine, resulting from these combined stresses has shifted community composition towards white fir and incense cedar and has altered forest stand structure (Miller et al., 1989). A shift of community composition towards white fir may make this ecosystem more susceptible to fire. Although the role of O_3 was extremely difficult to separate from other confounding factors, such as high N deposition, there is evidence that this shift in species composition has altered trophic structure and food web dynamics (Pronos et al., 1999) and C and N cycling (Arbaugh et al., 2003). Ongoing research in this important ecosystem will reveal the extent to which ecosystem services have been affected.

One of the best-documented studies of population and community response to O_3 effects are the long-term studies of common plantain (*Plantago major*) in native plant communities in the United Kingdom (Davison and Reiling, 1995; Lyons et al., 1997; Reiling and Davison,

1992c). Elevated O_3 significantly decreased the growth of sensitive populations of common plantain (Pearson et al., 1996; Reiling and Davison, 1992a, b; Whitfield et al., 1997) and reduced fitness as determined by decreased reproductive success (Pearson et al., 1996; Reiling and Davison, 1992a). While spatial comparisons of population responses to O_3 are complicated by other environmental factors, rapid changes in O_3 resistance were imposed by ambient levels and variations in O_3 exposure (Davison and Reiling, 1995). At the site of plantain seed collection, the highest correlations occurred between O_3 resistance and ambient O_3 concentrations (Lyons et al., 1997). In this case study, it appeared that O_3-sensitive individuals are being removed by O_3 stress and the genetic variation represented in the population could be declining. If genetic diversity and variation is lost in ecosystems, there may be increased vulnerability of the system to other biotic and abiotic stressors, and ultimately a change in the services provided by those ecosystems.

Reconstructed ecosystems in artificial exposure experiments have also provided new insight into how O_3 may be altering ecosystem structure and function (Karnosky et al., 2005). For example, the Aspen Free-Air CO_2 Enrichment facility was designed to examine the effects of both elevated CO_2 and O_3 on aspen (*Populus tremuloides*), birch (*Betula papyrifera*), and sugar maple (*Acer saccharum*) in a simple reconstructed plantation characteristic of Great Lakes aspen-dominated forests (Karnosky et al., 2003; Karnosky et al., 1999). They found evidence that the effects on above- and below-ground growth and physiological processes have cascaded through the ecosystem, even affecting microbial communities (Larson et al., 2002; Phillips et al., 2002). This study also confirmed earlier observations of O_3-induced changes in trophic interactions involving keystone tree species, as well as important insect pests and their natural enemies (Awmack et al., 2004; Holton et al., 2003; Percy et al., 2002).

Collectively these examples suggest that O_3 is an important stressor in natural ecosystems, but it is difficult to quantify the contribution of O_3 due to the combination of stresses present in ecosystems. Continued research, employing new approaches, will be necessary to fully understand the extent to which O_3 is affecting ecosystem services.

7.7.2 Effects on Ecosystem Services and Carbon Sequestration

Since it has been established that O_3 affects photosynthesis and growth of plants, O_3 is most likely affecting the productivity of forest ecosystems. Therefore, it is desirable to link effects on growth and productivity to essential ecosystem services. However, it is very difficult to quantify ecosystem-level productivity losses because of the amount of complexity in scaling from the leaf-level or individual plant to the ecosystem level, and because not all organisms in an ecosystem are equally affected by O_3.

Terrestrial ecosystems are important in the Earth's carbon (C) balance and could help offset emissions of CO_2 by humans if anthropogenic C is sequestered in vegetation and soils. The annual increase in atmospheric CO_2 is less than the total inputs from fossil fuel burning and land use changes (Prentice et al., 2001) and much of this discrepancy is thought to be attributable to CO_2 uptake by plant photosynthesis (Tans & White, 1998). Temperate forests of the northern hemisphere have been estimated to be a net sink of 0.6 to 0.7 Pg of C per year (Goodale et al. 2002). Ozone interferes with photosynthesis, causes some plants to senesce leaves prematurely and in some cases, reduces allocation to stem and root tissue. Thus, O_3 decreases the potential for C sequestration. For the purposes of this discussion, we define C sequestration as the net exchange of carbon by the terrestrial biosphere. However, long-term storage in the soil organic matter is considered to be the most stable form of C storage in ecosystems.

In a study including all ecosystem types, Felzer et al. (2004), estimated that U.S. net primary production (net flux of C into an ecosystem) was decreased by 2.6-6.8% due to O_3 pollution in the late 1980's to early 1990's. Ozone not only reduces C sequestration in existing forests, it can also affect reforestation projects (Beedlow et al. 2004). This effect, in turn, has been found to ultimately inhibit C sequestration in forest soils which act as long-term C storage (Loya et al., 2003; Beedlow et al. 2004). The interaction of rising O_3 pollution and rising CO_2 concentrations in the coming decades complicates predictions of future sequestration potential. Models generally predict that, in the future, C sequestration will increase with increasing CO_2, but often do not account for the decrease in productivity due to the local effects of tropospheric O_3. In the presence of high O_3 levels, the stimulatory effect of rising CO_2 concentrations on forest productivity has been estimated to be reduced by more that 20% (Tingey et al., 2001; Ollinger et al. 2002; Karnosky et al., 2003).

In summary, it would be anticipated that meeting lower O_3 standards would increase the amount of CO_2 uptake by many ecosystems in the U.S. However, the amount of this improvement would be heavily dependent on the species composition of those ecosystems. Many ecosystems in the U.S. do have O_3 sensitive plants. For, example forest ecosystems with dominant species such as aspen or ponderosa pine would be expected to increase CO_2 uptake more with lower O_3 than forests with more O_3 tolerant species.

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8. STAFF CONCLUSIONS AND RECOMMENDATIONS ON THE SECONDARY O_3 NAAQS

8.1 INTRODUCTION

This chapter presents staff conclusions and recommendations regarding an appropriate range of options for the Administrator to consider in selecting a pollutant indicator, averaging time, form, and level for the secondary O_3 standard. In so doing, this chapter describes the results and conclusions of staff assessments of scientific evidence presented in the CD and of air quality, exposure, and risk analyses presented in Chapters 2 and 7 herein. Comments and advice received from CASAC in their review of earlier drafts of this document, as well as comments on earlier drafts submitted by interested parties, that have significantly informed the development of staff's views, are also discussed.

In presenting policy options for the Administrator's consideration, we note that the final decision on retaining or revising the current secondary O_3 standard is largely a public welfare policy judgment. A final decision should draw upon scientific information and analyses about welfare effects, exposure and risks, as well as judgments about the appropriate response to the range of uncertainties that are inherent in the scientific evidence and analyses. Our approach to informing these judgments, discussed more fully below, is consistent with the requirements of the NAAQS provisions of the Act and with how EPA and the courts have historically interpreted the Act. These provisions require the Administrator to establish secondary standards that, in the Administrator's judgment, are requisite to protect public welfare from any known or anticipated adverse environmental effect. In so doing, the Administrator seeks to establish standards that are neither more nor less stringent than necessary for this purpose.

8.2 APPROACH

Welfare effects, as defined in section 302(h) (42 U.S.C. 7602(h)) of the Clean Air Act include, but are not limited to, ‘‘effects on soils, water, crops, vegetation, manmade 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.” As in the last review, this review has focused on vegetation and crops, since effects on these public welfare categories are well-studied and currently known to be of most concern at O_3 concentrations typically occurring in the U.S. Further, by adversely affecting natural vegetation and commercial crops, O_3 may also indirectly adversely affect natural ecosystems and their components (e.g., soils, water, animals, and wildlife). Therefore, these important but less well-studied indirect effects will be qualitatively discussed. As discussed above in Chapter 7, for

other welfare effects categories, insufficient new information was available to inform the selection of an indicator, form, averaging time or level for a distinct secondary standard and they are not discussed further, except in terms of research needs.

In evaluating whether the current secondary standard is adequate or whether consideration of revisions is appropriate, we adopted an approach in this review that builds upon the general approach used in the last review and reflects the broader body of evidence now available. In developing conclusions and recommendations for the Administrator to consider in this review, staff presents effects-, exposure- and risk-based considerations. We have expanded and modified the exposure and risk assessments to reflect the availability of new tools, assessment methods, and a larger and more diverse body of evidence. We have taken a weight of evidence approach that evaluates information across the variety of vegetation-related research areas described in the CD (e.g., seedling and mature forest tree species and commodity, fruit, vegetable and forage crop species), and includes assessments of air quality, exposures, and qualitative and quantitative risks associated with alternative air quality scenarios.

With respect to vegetation effects information, we have evaluated the conclusions drawn at the end of the last review in light of more recent evidence from chamber, free air, gradient, model and field-based observation studies for a variety of vegetation effects endpoints. We place greater weight on U.S. studies due to the often species-, site-, and climate-specific nature of O_3-related vegetation response. With respect to quantitative exposure- and risk-based considerations, we have relied on both monitored and interpolated O_3 exposures as described in section 7.5. of Chapter 7. Several alternative air quality scenarios were selected for evaluation to reflect a range of alternative standards under consideration. These scenarios include current "as is" air quality (2001), as well as six "just meet" scenarios for which interpolated O_3 air quality is adjusted using a rollback method to simulate just meeting a range of alternative standards. Uncertainties associated with the exposure and risk assessments are also discussed, including, where possible, some sense of the direction and/or magnitude of the uncertainties that should be taken into account as one considers these estimates. With regard to the use of the TREGRO model for estimating mature tree risks, staff acknowledges the presence of unknown and unquantifiable sources of uncertainty, as is typical with all such models.

In this review, a series of general questions frames our approach to informing conclusions and the identification of an appropriate range of policy options for consideration by the Administrator regarding the current secondary O_3 standard. Our consideration of the adequacy of the current standard begins in section 8.3.1 by addressing questions such as the following:

- To what extent does newly available information reinforce or call into question evidence of associations with effects identified in the last review?

- To what extent does newly available information reinforce or call into question any of the basic elements of the current standards?

- To what extent have important uncertainties identified in the last review been reduced and have new uncertainties emerged?

To the extent that the available information suggests that revision of the current standards may be appropriate to consider, we explore whether the currently available information supports consideration of a standard that is either more or less protective by addressing the following questions:

- Is there evidence that vegetation effects extend to ambient O_3 concentration levels that are as low as or lower than had previously been observed, and what are the important uncertainties associated with that evidence?

- Are exposures and vegetation risks of concern estimated to occur in areas that meet the current standard; are they important from a public welfare perspective; and what are the important uncertainties associated with the estimated risks?

To the extent that there is support for consideration of revised standards, we then identify a range of alternative standards (in terms of an indicator for photochemical oxidants, averaging time, level, and form in sections 8.3.2 through 8.3.5 below, respectively) that staff feels are appropriate for the Administrator to consider and that reflect staff conclusions and recommendations on the science, taking into account other public welfare policy considerations. In so doing, staff addresses the following questions:

- Does the evidence provide support for considering a different O_3 indicator?

- Does the evidence provide support for considering different averaging times?

- What ranges of levels and forms of alternative standards are supported by the evidence, and what are the uncertainties and limitations in that evidence?

- To what extent do specific levels and forms of alternative standards reduce the estimated exposures of concern and risks attributable to O_3 and other photochemical oxidants, and what are the uncertainties in the estimated exposure and risk reductions?

A summary of staff conclusions and recommendations regarding a range of policy options identified for the Administrator's consideration, as well as key CASAC and public commenter views concerning whether, and if so how, to revise the current secondary O_3 standard is presented in section 8.3.6 below. This chapter concludes with a discussion of key uncertainties and recommendations for additional research related to setting a secondary O_3 NAAQS in section 8.4.

8.3 SECONDARY O_3 STANDARD

The current secondary standard is a 3-year average of the annual 4th-highest maximum 8-hr average set at a level of 0.08 ppm. This standard was selected to provide protection to the public welfare, especially agricultural crops and other at-risk sensitive plant species, against a wide range of O_3-induced effects. As an introduction to our discussion in this section of the adequacy of the current O_3 standard, it is useful to summarize the key factors that formed the basis of the decision in the last review to revise the averaging time, level and form of the then current 1-hr secondary standard.

In the 1996 proposal notice (61 FR 65716), the Administrator proposed to replace the then existing 1-hr O_3 secondary NAAQS with one of two alternative new standards: a standard identical to the proposed 0.08 ppm, 8-hr primary standard (described above), or alternatively, a new seasonal standard expressed as a sum of hourly concentrations greater than or equal to 0.06 ppm, cumulated over 12 hours per day during the maximum 3-month period during the O_3 monitoring season (SUM06), set at a level of 25 ppm-hr. At the time, this latter standard was considered to be an annual standard. This proposal was based on a thorough review of the latest scientific information available and described in the 1996 O_3 CD, as well as (1) staff assessments of the policy-relevant information in the 1996 O_3 CD presented in the 1996 O_3 Staff Paper including air quality, vegetation exposure and risk, and economic values; (2) consideration of the degree of protection to vegetation potentially afforded by the proposed 0.08 ppm, 8-hr primary standard; (3) CASAC advice and recommendations; and (4) public comments.

In the final rule for the O_3 NAAQS published in July 1997 (62 FR 38877), the Administrator decided to replace the then current 1-hr, 0.12-ppm secondary NAAQS with a standard that was identical in every way to the new revised primary standard of an 0.08 ppm annual 4th-highest maximum 8-hr average standard averaged over 3 years. Her decision was based on her judgment that: (1) the then existing secondary standard did not provide adequate protection for vegetation against the adverse welfare effects of O_3; (2) reflected CASAC advice "that a secondary NAAQS, more stringent than the present primary standard, was necessary to protect vegetation from O_3" (Wolff, 1996); (3) the new 8-hr average standard would provide substantially improved protection for vegetation from O_3-related adverse effects as compared to the level of protection provided by the then current 1-hr, 0.12-ppm secondary standard; (4) significant uncertainties remained with respect to exposure dynamics, air quality relationships, and the exposure, risk, and monetized valuation analyses presented in the proposal, resulting in only rough estimates of the increased public welfare likely to be afforded by each of the proposed alternative standards; (5) there was value in allowing more time to obtain additional information to better characterize O_3-related vegetation effects under field conditions from additional research and to develop a more complete rural monitoring network and air quality

database from which to evaluate the elements of an appropriate seasonal secondary standard; and (6) there was value in allowing more time to evaluate more specifically the improvement in rural air quality and in O_3–related vegetation effects resulting from measures designed to attain the new primary standard (62 FR 38877-78).

The Administrator further concluded (62 FR 38877-78) that continued research on the effects of O_3 on vegetation under field conditions and on better characterizing the relationship between O_3 exposure dynamics and plant response would be important in the next review because:

- The available biological database highlighted the importance of cumulative, seasonal exposures as a primary determinant of plant responses.
- The association between daily maximum 8-hr O_3 concentrations and plant responses had not been specifically examined in field tests.
- The impacts of attaining an 8-hr, 0.08 ppm primary standard in upwind urban areas on rural air quality distributions could not be characterized with confidence due to limited monitoring data and air quality modeling in rural and remote areas.

8.3.1 Adequacy of Current O_3 Standard

The new evidence available in this review continues to support and strengthen key policy-relevant conclusions drawn in the previous review (U.S. EPA, 2006). Based on this new evidence, the current CD once more concludes that: (1) a plant's response to O_3 depends upon the cumulative nature of ambient exposure (e.g., concentration times duration) as well as the temporal dynamics of those concentrations; (2) current ambient concentrations in many areas of the country are sufficient to impair growth of numerous common and economically valuable plant and tree species; (3) the entrance of O_3 into the leaf through the stomata is the critical step in O_3 effects; (4) effects can occur with only a few hourly concentrations above 80 ppb; (5) other environmental biotic and abiotic factors are also influential to the overall impact of O_3 on plants and trees; and (6) a high degree of uncertainty remains in our ability to assess the impact of O_3 on ecosystem services. The effects-based evidence described in the CD underlying the reaffirmation of these conclusions will be discussed in more detail in the sections that follow. Based on the above policy-relevant findings from the CD, and while recognizing that important uncertainties and research questions remain, we also conclude that progress has been made since the last review and thus, we generally find support in the available effects-based evidence for consideration of an O_3 standard that is at least as protective as the current standard and do not find support for consideration of an O_3 standard that is less protective than the current standard. This general conclusion is consistent with the advice and recommendations of CASAC and with

the views expressed by all the interested parties who provided comment on the previous draft of this document.

Having reached this general conclusion, we then evaluated the adequacy of the current standard by considering to what degree risks to vegetation and ecosystems would be expected to occur after just meeting the current as well as a range of alternative secondary standards. As discussed in Chapter 7 and in greater detail below, staff conclusions regarding the adequacy of the current standard are based on the available vegetation effects, exposure and risk-based evidence (section 8.3.1.1) and CASAC and public commenter views (section 8.3.1.2) in conjunction with the additional policy-relevant considerations presented under the discussions on indicator, averaging time, form, and level (sections 8.3.2 through 8.3.5). In evaluating the strength of this information, staff has taken into account the uncertainties and limitations in the scientific evidence and analyses as well as considered the views of CASAC and other interested parties provided on the second draft of this document.

8.3.1.1 Vegetation Evidence-, Exposure- and Risk-Based Considerations

In the last review, crop yield and seedling biomass loss open-top chamber (OTC) data provided the basis for staff analyses, conclusions, and recommendations (U.S. EPA, 1996b). Since then, several additional lines of evidence have progressed sufficiently to provide staff with a more complete and coherent picture of the scope of O_3-related vegetation risks, especially those currently faced by seedling, sapling and mature tree species growing in field settings, and indirectly, forested ecosystems. Specifically, new research reflects an increased emphasis on field-based exposure methods (e.g., free air exposure and ambient gradient), improved field survey biomonitoring techniques, and mechanistic tree process models. Findings from each of these research areas are discussed separately below. However, in reaching conclusions regarding the adequacy of the current standard, staff has considered the combined information from all these areas together, using an integrated, weight of evidence approach.

In evaluating the degree to which the current standard is adequate in protecting vegetation at the national scale, staff has relied on both measured and modeled air quality information. For some effects, like visible foliar injury and modeled mature tree growth response, only monitored air quality information was used. For other effects categories (e.g., crop yield and tree seedling growth), staff relied on interpolated O_3 exposures. Staff recognizes that exposures predicted by this interpolation method are more uncertain. The uncertainties associated with this approach are discussed under the exposure assessment discussion below. Additional sources of uncertainty associated with the risk assessment are described in the section preceding the discussion of seedling and mature tree biomass-loss risk results.

Visible Foliar Injury Evidence

Recent systematic injury surveys continue to document visible foliar injury symptoms diagnostic of phytotoxic O_3 exposures on sensitive bioindicator plants. These surveys produce more expansive evidence than that available at the time of the last review that visible foliar injury is occurring in many areas of the U.S. under current ambient conditions. Staff performed an assessment combining recent U.S. Forest Service Forest Inventory and Analysis (FIA) biomonitoring site data with the county level air quality data for those counties containing the FIA biomonitoring sites. This assessment showed that incidence of visible foliar injury ranged from 21 to 39% during the four-year period (2001-2004) across all counties with air quality levels at or below that of the current 8-hr standard. The magnitude of these percentages suggests that phytotoxic exposures sufficient to induce visible foliar injury would still occur in many areas after meeting the level of the current secondary standard. Additionally, the data show that visible foliar injury occurrence is geographically widespread and is occurring on a variety of plant species in forested and other natural systems (see Figure 7-19 in section 7.6.3.2). Linking visible foliar injury to other plant effects is still problematic. However, its presence indicates that other O_3–related vegetation effects could also be present.

The presence of visible foliar injury can adversely impact the public welfare. For example, visible foliar injury in national parks and wilderness areas can impact the aesthetic experience for both outdoor enthusiasts and the occasional park visitor. In addition, because these areas are afforded a higher degree of protection, the presence of O_3-induced vegetation effects, including visible foliar injury, can take on increased significance. Specifically, federal land managers (FLMs) "…have determined that given the high ecological, aesthetic, and intrinsic value of federal lands, all native species are significant and warrant protection" (NPS, 2000). As a result, FLMs have identified visible foliar injury, along with other O_3-induced vegetation effects, as air quality related values (AQRV) of concern (NPS, 2000). As shown in Appendix 7J, numerous O_3 sensitive species are found on Class I federal lands. In addition, the presence of visible foliar injury also has the potential to economically impact for those who rely on healthy looking vegetation for their livelihood (e.g., horticulturalists, farmers of leafy crops, landscapers, Christmas tree growers). Many ornamental species have been listed as sensitive to O_3 (Abt, 1993). Similarly, early senescence of fall foliage could also diminish the time available for viewing fall foliage, important in some regions of the country in drawing tourists. Although data are not available to allow the quantification of these impacts, the potential for their existence should not be overlooked.

Exposure-Based Considerations

As described in Chapter 7, due to the paucity of rural O_3 monitoring data, it was necessary to select an interpolation method that could be used to characterize O_3 air quality over

broad geographic areas. Staff recognizes there are inherent uncertainties in the interpolation that must rely on sparse data that, for the most part, are representative of urban and near-urban areas. The interpolation method used for the western U.S. contains additional sources of uncertainty associated with the use of CMAQ model outputs to develop scaling factors for the interpolation. See section 7.5 of chapter 7 for details on how the interpolation was constructed and how staff quantified the uncertainties (error and bias) associated with the interpolation. This quantification of exposure uncertainty for the interpolation represents a notable improvement over the 1996 assessment which did not have an evaluation of the exposure surface. In general, this interpolation method under-predicts higher 12-hr W126 exposures. Due to the important influence of higher exposures in determining risks to plants, this feature of the interpolated surface could result in an under-estimation of risks to vegetation in some areas. Taking these uncertainties into account, and given the absence of more complete rural monitoring data, staff judged that this approach was appropriate to use in developing national vegetation exposure and risk assessments that estimate relative changes in risk for the various alternative standards analyzed.

To evaluate changing vegetation exposures and risks under selected "just meet" scenarios, staff analytically adjusted 2001 base year air quality distributions with a rollback method (Horst and Duff, 1995; Rizzo, 2005 & 2006) to reflect "just meeting" the current and alternative secondary standard options. This technique combines both linear and quadratic elements to reduce higher O_3 concentrations more than lower ones. In this regard, the rollback method attempts to account for reductions in emissions without greatly affecting lower concentrations near ambient background levels. The following "just meet" air quality scenarios were generated along with maps for several scenarios (see Figures 7-7, 7-8, 7-9, 7-10):

- 4^{th}-highest daily maximum 8-hr average: 0.084 ppm (the effective level of the current standard) and 0.070 ppm levels
- 3-month, 12-hr. SUM06: 25 ppm-hr (proposed in the 1996 review) and 15 ppm-hr levels
- 3-month, 12-hr. W126: 21 ppm-hr and 13 ppm-hr levels

Staff's rationale for selecting these six alternative standards for evaluation is presented here and in section 7.5.1 of Chapter 7. The two 8-hr average levels were chosen as possible alternatives of the current form for comparison with the cumulative seasonal alternative forms. For both the SUM06 and W126 forms, the two levels were selected on the basis of the associated levels of crop yield loss protection described in the last review. Specifically, both the upper levels of SUM06 (25 ppm-hr) and W126 (21 ppm-hr) were associated with a level of crop protection of approximately no more than 10% yield loss in 50% of crop cases studied in the

National Crop Loss Assessment Network (NCLAN) experiments (section 7.6.2.2.). Alternatively, the lower levels of both SUM06 (15 ppm-hr) and W126 (13 ppm-hr) were associated with a level of crop protection of approximately no more than 10% yield loss in 75% of NCLAN cases. Another level to note is the upper level benchmark of W126 of 31 ppm-hr that approximates the upper end of the SUM06 range analyzed in the last review (U.S. EPA, 1996b) and which was associated with no more than 17% yield loss in 50% of crop cases as described in the last review. The above levels have also been associated with varying levels of tree seedling biomass loss protection based on a similar set of tree seedling studies performed by scientists in the National Health and Environmental Effects Research Lab, Western Ecology Division (NHEERL-WED).

Under the base year (2001) air quality, a large portion of California had 12-hr W126 above 31 ppm-hr, which has been reported to produce approximately 14% biomass loss in 50% of NHEERL-WED tree seedling studies. Broader multi-state regions in the east and west are predicted to have levels of air quality above the W126 level of 21 ppm-hr, which is approximately equal to the secondary standard proposed in 1996 and is associated with approximately 9% biomass loss in 50% of tree seedlings studied. Much of the east and Arizona and California have 12-hr W126 values above 13 ppm-hr which has been reported to allow approximately 7% biomass loss in 25% of tree seedlings studied. Although there is appreciable uncertainty associated with these exposure estimates, the results of the exposure assessment indicates that current air quality levels could result in significant impacts to vegetation in some areas.

When 2001 air quality is rolled back to just meet the current 8-hr secondary standard, the overall 3-month 12-hr W126 exposures do not improve by much (Figure 7-7). Under this scenario, there are still many areas of the country that have seasonal O_3 levels above the 12-hr W126 level of 21 ppm-hr. The exposure maps generated for the 0.070 ppm, 4^{th}-highest maximum 8-hr average alternative standard, the SUM06 alternatives of 25 and 15 ppm-hr, and the W126 alternatives of 21 and 13 ppm-hr (Figures 7-8, 7-9, 7-10), all showed a markedly improved picture of O_3 air quality compared to the current standard (Figure 7-7). Thus, the staff observes that all other alternative standards, when met at all locations, would be expected to provide improved protection of vegetation from seasonal O_3 exposures of concern over the current standard. As expected, however, the greatest improvements in air quality and estimated exposures to sensitive vegetation were observed when just meeting the lower W126 alternative of 13 ppm-hr, the SUM06 alternative of 15 ppm-hr, and the 0.07 ppm, 8-hr alternative standard.

Risk-Based Considerations

This review continues to rely upon the concentration-response (C-R) functions developed from OTC exposure systems (also relied upon in the 1996 review). Due to what has been

described in the scientific literature as the "chamber effect," some continue to express concern as to the appropriateness of applying OTC generated C-R functions to non-chambered environments. A shift toward the use of more field-based approaches (e.g., free air exposure and ambient gradient) in recent research has occurred, providing information in the peer-reviewed literature that at least qualitatively informs how one might weigh this concern. These new field-based studies, conducted on a limited number of crop and tree seedling species to date, demonstrate plant growth and visible foliar injury responses similar in nature and magnitude to those observed previously under OTC exposure conditions. These findings lend qualitative support to the conclusion that OTC conditions do not fundamentally alter the nature of the O_3-plant response. A related concern with respect to the use of the OTC C-R functions for crops is the concern that the crop varieties grown today may have O_3 sensitivities significantly different than those used to derive the NCLAN crop and OTC tree seedling C-R functions which are relied upon in this review. Nothing in the recent literature, however, suggests that the O_3 sensitivity of crop or tree species studied in the last review and for which C-R functions were developed has changed significantly in the intervening period. Indeed, in the few recent studies where this is examined, O_3 sensitivities are found to be as great or greater than those observed in the last review. As a result, staff continues to rely on the C-R functions available in the last review for predicting relative crop yield and tree seedling biomass loss potentials across a range of possible ambient O_3 exposures.

An additional source of uncertainty not described or accounted for in the last review is that associated with the presence of a decreasing O_3 gradient from the height of the monitor probe down to the lower plant canopy heights for most crop and seedling trees. The presence of this gradient makes less certain the predictions of current crop and tree seedling exposures and the associated yield and biomass losses, respectively, based on ambient monitor data. Staff selected a 10% reduction factor to represent the maximum gradient believed to occur for daylight hours. However, recognizing that the actual downward adjustment value varies depending on interactions between numerous plant and site-specific factors, staff chose to present estimates of yield and biomass loss for each crop and tree seedling species, respectively, as a range, with non-adjusted and 10%-adjusted air quality as the upper and lower bounds (See Chapter 7, section 7.6.2.3 for a detailed discussion).

Seedling and Mature Tree Biomass Loss. Biomass loss in sensitive tree seedlings is predicted to occur under O_3 exposures that just meet the level of the current secondary standard (see Table 7F-5 in Appendix 7F). For instance, black cherry, ponderosa pine, eastern white pine, and aspen had estimated median seedling biomass losses as high as 24, 11, 6, and 6%, respectively, when air quality was rolled back to just meet the current 8-hr standard with the 10% adjustment applied. Staff notes that these results are for tree seedlings and that mature trees of

the same species may have more or less of a response to O_3 exposure. Due to the potential for compounding effects over multiple years, a consensus workshop on O_3 effects reported that a biomass loss greater than 2% annually can be significant (Heck and Cowling, 1997). Decreased seedling root growth and survivability could affect overall stand health and composition in the long term.

Our analysis using modeled mature tree growth response under different air quality scenarios for a western species (ponderosa pine) and two eastern species (red maple and tulip poplar) projected that just meeting the current standard would likely continue to allow O_3-related reductions in annual net biomass gain in these species (see Table 7-5 in Chapter 7). This judgment is based, in part, on model outputs that estimate that as O_3 levels are reduced below those of the current standard, significant improvements in growth would occur. For instance, estimated growth in red maple increased by 4% and 3% at Big Meadows and Cranberry sites, respectively, when air quality was rolled back to just met a SUM06 value of 15 ppm-hr (approximately equivalent to a W126 value of 13 ppm-hr). Yellow poplar was projected to have a growth increase between 0.6 and 8% under the same scenarios at the two sites.

Though there is significant uncertainty associated with this analysis, we judge that this information should be given careful consideration in light of several other pieces of evidence. Specifically, limited evidence from experimental studies that goes beyond the seedling growth stage continues to show decreased growth under elevated O_3 (King et al. 2005). Some mature trees such as red oak have shown an even greater sensitivity of photosynthesis to O_3 than seedlings of the same species (Hanson et al., 1994). As indicated above, smaller growth loss increments may be significant for perennial species. The potential for cumulative "carry over" effects as well as compounding must be considered. The accumulation of such "carry-over" effects over time may affect long-term survival and reproduction of individuals and ultimately the abundance of sensitive tree species in forest stands.

Crop Yield Loss. Staff exposure and risk assessments estimate that just meeting the current 8-hr standard would still allow O_3–related yield loss to occur in several fruit and vegetable species and major commodity crop species currently grown in the U.S. (see Table 7F-4 in Appendix 7F). These estimates are substantially lower than those estimated in the last review as a result of several factors. First, O_3 air quality has improved in many areas of the country since the last review. Secondly, staff has factored in an O_3 adjustment for the height gradient, as described above, and will present results for both non-adjusted and adjusted exposure levels to approximate upper and lower bounds of predicted yield loss.

Several sources of uncertainty should be taken into account when evaluating the significance of these findings. First, yield loss estimates were generated using the median C-R function when more than one function was available for a given species. For some species,

however, only one C-R function was available. In this latter case, there is more uncertainty regarding the range of variability in O_3 sensitivity within each crop. Secondly, six of the fruit and vegetable species were not part of the NCLAN program and C-R functions were available only in terms of seasonal 7-hr or 12-hr mean indices. These indices are considered less effective in predicting plant response for a given change in air quality than cumulative forms used with other crops. Therefore, staff places less weight on the fruit and vegetable yield loss numbers than those for commodity crops, even though the magnitude of the fruit and vegetable effect was much greater. Finally, staff recognize that agricultural systems are heavily managed and vulnerable to adverse impacts from a variety of other factors (e.g., weather, insects, disease), which can overshadow the magnitude of yield impacts predicted for a given O_3 exposure. However, it should also be recognized that, in some experimental cases, exposure to O_3 has made plants more sensitive or vulnerable to other important stressors such as disease, insect pests, and harsh weather (U.S. EPA, 2006). Due to the significant impact these other stressors can have on crop production in some areas, staff recommends that additional research be done to better understand the nature and significance of these interactive effects of O_3 with other plant stressors.

Keeping these uncertainties in mind, the results of the risk assessment show that when air quality is rolled back to just meet the current standard, yield loss is still estimated to occur in several fruit and vegetable species and major commodity crop species currently grown in the U.S. (see 7.6.2.4 of Chapter 7). For example, based on median C-R function response, in counties with the highest O_3 levels, potatoes and cotton had estimated yield losses of 9-15% and 5-10%, respectively, when air quality just met the level of the current standard. Estimated yield improved in these counties when the alternative SUM06 and W126 standard levels were met. The very important soybean crop had generally small yield losses throughout the country under current air quality (0-6%) and just meeting the current standard (0-4%).

Another group of crops, multiple year forage crops, have also received additional study since the last review. Based on these new studies, the yields and quality of multiple-year forage crops have also been shown to be sufficiently reduced as to have nutritional and possibly economic implications for their use as ruminant animal feed at O_3 exposures that occur in some years over large areas of the U.S. However, it is not clear at this time to what degree they are impacted at lower levels of air quality, since the studies were not designed to address this question.

Summary

In summary, O_3 levels that would be expected to remain after meeting the level of the current secondary standard are sufficient to cause visible foliar injury, seedling and mature tree growth, and reduce crop yields. Other O_3-induced effects described in the literature include an

impaired ability of many sensitive species and genotypes within species to adapt to or withstand other environmental stresses such as freezing temperatures, pest infestations and/or root disease, and reduced ability to compete for available resources. In the long run, the result of these impairments (e.g., loss in vigor) may be premature plant death. Though effects on other ecosystem components have not been examined, except in isolated cases, effects such as those described above could have significant implications for plant community and associated species biodiversity and the structure and function of whole ecosystems (Young and Sanzone, 2002).

8.3.1.2 CASAC and Public Commenter Views on the Adequacy of the Current Standard

Staff recognizes that the exposure-and risk-based information can be considered both in terms of whether the risks estimated to remain upon attaining the current standard are important from a public welfare perspective and/or whether additional reductions in risk estimated to be associated with alternative, more protective standards are also important from a public welfare perspective. Judgments about the importance of the estimates of exposure and risks need to take into account the important uncertainties associated with such estimates.

There is general recognition among staff, CASAC, and all interested parties that public welfare policy judgments, including the weight to place on various types of evidence and how to weigh the importance of estimated risks in a public welfare perspective, are ultimately decisions left to the Administrator. To help inform those judgments with regard to the adequacy of the current secondary O_3 standard, the views expressed by CASAC as well as the views of other interested parties who have commented on earlier drafts of this document are summarized here. The range of views generally reflects differing judgments as to the relative weight to place on various types of exposure- and risk-based information, and the associated uncertainties, as well as differing judgments about the importance of various O_3-related vegetation effects from a public welfare perspective.

In a letter to the Administrator (Henderson, 2006c), the CASAC O_3 Panel, with full endorsement of the chartered CASAC, unanimously concluded that "despite limited recent research, it has become clear since the last review that adverse effects on a wide range of vegetation including visible foliar injury are to be expected and have been observed in areas that are below the level of the current 8-hour primary and secondary ozone standards…" Therefore, "based on the Ozone Panel's review of Chapters 7 and 8, the CASAC unanimously agrees that it is not appropriate to try to protect vegetation from the substantial, known or anticipated, direct and/or indirect, adverse effects of ambient ozone by continuing to promulgate identical primary and secondary standards for ozone. Moreover, the members of the Committee and a substantial

majority of the Ozone Panel agrees with EPA staff conclusions and encourages the Administrator to establish an alternative cumulative secondary standard for ozone and related photochemical oxidants that is distinctly different in averaging time, form and level from the currently existing or potentially revised 8-hour primary standard" (Henderson, 2006c).

In contrast to the views of CASAC discussed above, others submitted comments that supported retaining the current standards.[1] In considering the available evidence as a basis for their views, these commenters identified a number of key concerns that, in their view, make it inappropriate to revise the secondary standard at this time. For example, they assert: 1) The key uncertainties cited by the Administrator in the 1997 review as reasons for deciding it was not appropriate to move forward with a seasonal secondary, (e.g., uncertainties in the exposure, risk and valuation analyses and the lack of air quality data in rural and remote areas), have not been materially reduced in this current review; and 2) The exposure assessment is inaccurate and too uncertain due to the use of low estimates of policy-relevant background (PRB), an arbitrary rollback method that is uninformed by atmospheric chemistry from photochemical models, and the use of the CMAQ model in the west, whose biases and uncertainties are insufficiently characterized and evaluated.

8.3.1.3 Staff Conclusions on the Adequacy of the Current Standard

On the basis of the vegetation effects that have been observed to still occur under current ambient exposure conditions and those predicted to occur under the scenario of just meeting the current secondary NAAQS, staff concludes that the current secondary NAAQS is inadequate to protect the public welfare from known and anticipated adverse welfare effects. As discussed above, this conclusion derives from several lines of evidence.

First, visible foliar injury observations for the years 2001 to 2004 at USDA FIA biomonitoring sites show widespread O_3–induced effects occurring in the field, including in forested ecosystems. For a few studied species, it has been further shown that the presence of visible foliar injury is linked to the presence of other vegetation effects (e.g., reduced plant growth and impaired below ground root development) (U.S. EPA, 2006), though for most species, making this linkage remains problematic. Nevertheless, when visible foliar injury is present, the possibility that other O_3-induced vegetation effects could also be present should be considered. Staff recognizes that it is not possible at this time to quantitatively assess the degree of visible foliar injury that should be judged adverse in all settings and across all species, and that other environmental factors can mitigate or exacerbate the degree of O_3-induced visible foliar injury expressed at any given concentration of O_3. However, recognizing that the presence

[1] This group of commenters included industry associations, corporations, and individuals.

of visible foliar injury alone can be adverse to the public welfare (see foliar injury discussion in 8.3.1.1), and on the basis of the above considerations, staff concludes that the current standard continues to allow levels of visible foliar injury that could reasonably be considered to be adverse from a public welfare perspective.

Second, a recent ambient gradient study and a free air O_3 enrichment (FACE) experiment have supported earlier findings from O_3 experiments conducted in OTC. Studies conducted at the AspenFACE site in Wisconsin (see section 7.6.2.1 of chapter 7) on quaking aspen has confirmed the detrimental effects of O_3 exposure on tree growth in a field study without chambers (Isebrands et al., 2000, 2001). The recent ambient gradient study (Gregg et al, 2003) evaluated biomass loss in cottonwood along an urban-to-rural gradient at several locations. Study results found that conditions in the field were sufficient to produce substantial biomass loss in cottonwood, with larger impacts observed in downwind rural areas due to the presence of higher O_3 concentrations (See Section 7.6.3). Staff's inclusion and emphasis on these two field-based lines of evidence is consistent with the Administrator's conclusion at the end of the last review (62 FR 38877-78), that continued research on the effects of O_3 on vegetation under field conditions would be important in this next review. Staff feels that the expanded field-based evidence provides qualitative support for the continued usefulness of findings obtained from chamber studies.

Staff's conclusion is further strengthened by evidence of remaining impacts on tree seedling biomass loss when the current 8-hr standard is met. Staff estimated annual biomass loss up to 6-24% for some sensitive species in areas of high O_3 exposure. Because of the potential for indirect effects on plant vigor from even small incremental biomass or growth reductions in the field, staff observes that these levels of tree seedling growth reduction are well above the 1-2% range of concern identified by the 1997 consensus workshop (Heck and Cowling, 1997). Staff also took into account modeled mature tree growth loss estimates and commodity crop and fruit and vegetable yield loss in arriving at these conclusions. Linkages across ecosystem hierarchies (Young and Sanzone, 2002) make indirect impacts to ecosystems another welfare effects category of concern even after attaining the current secondary standard.

8.3.2 Pollutant Indicator

The staff concludes that O_3 remains the appropriate pollutant indicator for use in a secondary NAAQS that provides protection for public welfare from exposure to all photochemical oxidants. This conclusion is based on the same rationale presented in the previous Staff Paper (U.S. EPA, 1996b), which recognizes that among the other photochemical oxidants, the database for vegetation effects only raises concern at levels found in the ambient air

for O_3 and, therefore, control of ambient O_3 levels provides the best means of controlling other photochemical oxidants of potential welfare concern.

8.3.3 Averaging Times

Plants, unlike people, are exposed to ambient air 24 hours a day, every day for their entire life. For annual species, this is for only a period within one year; for perennials, exposures are for multiple years, decades or centuries. Regardless of plant type, it has been well established in the literature that O_3 effects are cumulative, and that longer exposure durations have a greater impact than shorter durations, all else being equal (U.S. EPA, 2006). Air quality indices that account for the exposure duration overall do a better job predicting plant response than short- or long-term averages. However, O_3 levels are not continuously elevated and plants are not equally sensitive to O_3 over the course of a day, season or lifetime. Thus, it becomes necessary to identify periods of exposure that have the most relevance for plant response.

8.3.3.1 Seasonal Window

Many recent studies described in the 2006 CD have specifically selected exposure indices that take into account the cumulative, concentration-weighted impact of O_3-induced effects throughout the growing season when measuring growth and yield impacts and have substantiated the 1996 CD and 1996 Staff Paper conclusions on the importance of cumulative, seasonal exposures (U.S. EPA, 2006). Annual crops are typically grown for periods of two to three months before being harvested. In contrast, perennial species may be photosynthetically active longer (up to 12 months each year for a few species) depending on the species and where it is grown. In general, the period of maximum physiological activity and thus, potential O_3 uptake for annual crops, herbaceous species, and deciduous trees and shrubs coincides with some or all of the intra-annual period defined as the O_3 season, which varies on a state-by-state basis. This is because the high temperature and high light conditions that promote the formation of tropospheric O_3 also promote physiological activity in vegetation.

In the 1996 Staff Paper and proposal notice, we noted that the selection of any single averaging time for a national standard would represent a compromise, given the significant variability in growth patterns and lengths of growing seasons among the wide range of vegetation species that may experience adverse effects associated with O_3 exposure. However, we concluded, based on the information available at that time, that selection of the maximum consecutive 3-month period within the O_3 season was reasonable, and in most cases, would most likely coincide with the periods of greatest plant sensitivity on an annual basis. Based on the information assessed in the current CD (U.S. EPA, 2006) and Chapter 7 above, we again conclude the maximum consecutive 3-month period within the O_3 season is a reasonable averaging time for vegetation.

8.3.3.2 Diurnal Window

Stomata are the entry points for O_3 into plant leaves. Over the course of a day, plant stomatal conductance varies along with light level, soil moisture and other factors. In general, stomata are most open during daylight hours in order to allow sufficient CO_2 uptake for use in carbohydrate production through the light driven process of photosynthesis. At most locations, O_3 concentrations are also highest during the daytime, potentially coinciding with maximum stomatal uptake. Ozone uptake during daylight hours impairs the light-driven process of photosynthesis, which can then lead to impacts on carbohydrate production, plant growth, reproduction (yield) and root function. Thus, in the last review, staff selected the 12-hr daylight window (8 am to 8 pm) to capture the diurnal window with most relevance to the photosynthetic process. Since that time, some limited work has been done by Musselman and Minnick (2000) to more fully characterize O_3 uptake at night and its potential contribution to total plant uptake and response**.** This work reports that some species do take up O_3 at night, but that the degree of nocturnal stomatal conductance varies widely between species and its relevance to overall O_3-induced vegetation effects remain unclear. We conclude that such information continues to be preliminary and not generalizable at this time (see also Appendix 7A of Chapter 7). Staff, therefore, again concludes that the daytime 12-hr window is the most appropriate period over which to cumulate diurnal O_3 exposures, specifically those most relevant to plant growth and yield responses.

8.3.3.3 Alternative Views and Staff Conclusions

The CASAC expressed views in agreement with staff with respect to both seasonal and diurnal averaging times. Specifically, CASAC states "the suggested approach to the secondary standard is a cumulative seasonal growing standard such as the indices SUM06 or W126 aggregated over at least the three summer months exhibiting the highest cumulative ozone levels and includes the ozone exposures from at least 12 daylight hours."

In contrast, some commenters pointed to new information on nocturnal conductance as evidence for the need for a 24-hour diurnal window. Specifically, they state "an extensive review of the literature reported that a large number of species had varying degrees of nocturnal stomatal conductance." Based on this review, Musselman and Minnick (2000) recommend that any O_3 exposure index used to relate air quality to plant response should use the 24-hour cumulative exposure period. No commenters addressed the adequacy of the three month seasonal window. In examining the available information on nocturnal conductance (See Appendix 7A), staff concludes that it remains unclear to what extent nocturnal uptake contributes to the vegetation effects of yield loss, biomass loss or visible foliar injury. Due to the many species- and site-specific variables that influence the potential for and significance of nocturnal

uptake, staff concludes that additional research needs to be done before considering whether this component of vegetation exposure should be addressed with a different averaging time.

Based on these considerations, as well as information assessed in the current CD (U.S. EPA, 2006) and Chapter 7 above, we again conclude that a 12-hr (8:00 am to 8:00 pm) diurnal window remains appropriate for a secondary NAAQS designed to protect a wide range of vegetation growing in environmental conditions found across the U.S**.**

8.3.4 Form of the Standard

The 2006 O_3 CD states, "In the 1996 O_3 CD…, it was concluded, based on the best available data, that those O_3 exposure indices that cumulated differentially weighted hourly concentrations were the best candidates for relating exposure to plant growth response…. The few studies that have been published since the 1996 O_3 CD continue to support the earlier conclusions…" (U.S. EPA, 2006, pg. 9-12). The following selections taken from the 1996 CD (see U.S. EPA, 1996a, pgs. 5-88/89, 5-95/96), further elucidate the depth and strength of these conclusions. "When O_3 effects are the primary cause of variation in plant response, plants from replicate studies of varying duration showed greater reductions in yield or growth when exposed for the longer duration." "The mean exposure index of unspecified duration could not account for the year-to-year variation in response." "Because the mean exposure index treats all concentrations equally and does not specifically include an exposure duration component, the use of a mean exposure index for characterizing plant exposures appears inappropriate for relating exposure with vegetation effects"

Though the scientific justification for a cumulative, seasonal form was generally accepted in the last review, an analysis undertaken by EPA at that time had showed that there was considerable overlap between areas that would be expected not to meet the range of alternative 8-hr standards being considered for the primary NAAQS and those expected not to meet the range of values (expressed in terms of the seasonal SUM06 index) of concern for vegetation. This result suggested that improvements in national air quality expected to result from attaining an 8-hr primary standard within the recommended range of levels would also be expected to reduce levels below those of concern for vegetation in those same areas. Thus, in the proposal notice, the Administrator proposed two alternatives for consideration: one alternative was to make the secondary standard equal in every way to the proposed 8-hr, 0.08 ppm primary standard; and the second was to establish a 3-month, 12-hr SUM06 seasonal secondary standard (set at a level of 25 ppm-hr) as also appropriate to protect public welfare from known or anticipated adverse effects given the available scientific knowledge and that such a seasonal standard "…is more biologically relevant…" (61 FR 65716). In the 1997 final rule, the decision was made to make the secondary identical to the primary standard. It acknowledged, however,

that "it remained uncertain as to the extent to which air quality improvements designed to reduce 8-hr average O_3 concentrations averaged over a 3-year period would reduce O_3 exposures measured by a seasonal SUM06 index." (62 FR 38876) In other words, it was uncertain as to whether the 8-hr average form would, in practice, provide sufficient protection for vegetation from the seasonal, cumulative and concentration-weighted exposures described in the scientific literature as of concern.

On the basis of that history, Chapters 2 and 7 of this Staff Paper revisited the issue of the appropriateness of using an 8-hr average standard form to provide the requisite protection required for vegetation.

8.3.4.1 Comparison of 8-Hour Average and Cumulative Seasonal Forms

Staff performed an analysis to evaluate the extent to which there appears to be a relationship between county level air quality measured in terms of the current 8-hr average form and that measured in terms of an alternative cumulative, seasonal form (e.g. 12-hr W126). Staff determined it was most useful to begin by comparing the 3-year averages of each form, since the current 8-hr average secondary form is a 3-year average. However, in recognition that some vegetation effects (e.g. crop yield and foliar injury) are driven solely by annual O_3 exposures, and that typically the cumulative forms are defined in terms of the annual growing season, staff also performed a comparison of the current 8-hr form to the annual W126 air quality values for both 2002 and 2004 (see Appendix 7B).

Staff performed this analysis using recent (2002-2004) county-level air quality data from AQS sites and the subset of CASTNET sites having the highest O_3 levels for the counties in which they are located. Due to the lack of more complete monitor coverage in many rural areas, staff acknowledges that this analysis may not be an accurate reflection of the situation in non-monitored, rural counties. Results of the 3-year average comparison showed that after meeting the current 3-year average form of the 0.08-ppm, 8-hr average standard, only a few counties showed 3-year average W126 values above the upper level (21 ppm-hr) evaluated (see Figure 7-1). This result, taken alone, might suggest that areas that met the current level and form would typically overlap with the areas that met the analyzed alternative cumulative level and form. However, at the lower W126 level of 13 ppm-hr (see discussion on level in section 8.3.5 below), many more counties that meet the current 8-hr standard level and form no longer meet the alternative W126 form at this level. When individual years are compared, this lack of a relationship becomes clearer. For example, the relatively high 2002 air quality year, showed a greater degree of overlap between those areas that would meet the levels analyzed for the current 8-hr and alternative W126 forms than did the relatively low 2004 air quality year (See Appendix 7B). It is clear from this analysis that the degree to which the current 8-hr standard form and

level would overlap with areas of concern for vegetation expressed in terms of the 12-hr W126 standard is inconsistent and would depend greatly on the level of the 12-hr W126 and 8-hr forms selected and the distribution of hourly O_3 concentrations within the annual and/or 3 year average period. It is not clear how this relationship would change due to the change in O_3 patterns resulting from control strategies put in place to attain different levels of standards.

This view is consistent with those of CASAC who unanimously agree that it is not appropriate to try to protect vegetation from the substantial, known or anticipated, direct and/or indirect, adverse effects of ambient ozone by continuing to promulgate identical primary and secondary standards for ozone. Moreover, the members of CASAC and a substantial majority of the CASAC O_3 Panel agree with staff conclusions and encourage the Administrator to establish an alternative cumulative secondary standard for ozone and related photochemical oxidants that is distinctly different in averaging time, form, and level from the currently existing or potentially revised 8-hour primary standard. The suggested approach to the secondary standard is a cumulative seasonal growing standard such as the indices SUM06 or W126 aggregated over at least the three summer months exhibiting the highest cumulative O_3 levels and includes the O_3 exposures from at least 12 daylight hours (Henderson, 2006c).

Some other public commenters agreed that "directionally a cumulative form of the standard may better match the underlying data." However, they believe further work is needed to determine whether a cumulative exposure index for the form of the secondary standard is needed. Specifically, a few commenters were of the view that a W126 (or SUM06) was not sufficient in and of itself but should be combined with a measure of the number of peaks above 100 ppb (N100). Some of these same commenters also felt a 24-hr averaging time was supported by the data on nocturnal stomatal conductance.

Staff recognizes that the relationship between O_3 exposure and plant response is more complex than described by the single component indices and supports the need for further research into improvements on such indices to better capture factors that influence flux. Staff also recognizes that meeting the current 8-hr standard would result in air quality improvements that could potentially benefit vegetation in some areas. However, at this time, based on the weight of evidence in the scientific literature demonstrating the cumulative nature of O_3-induced plant effects and the need to give greater weight to higher concentrations, the advice of CASAC consistent with this view, and the results of the above analysis, staff again concludes that a secondary standard should, at a minimum, be defined in terms of a form that reflects the two components of exposure known to influence plant response, i.e. differentially weighted peak concentrations and cumulative seasonal exposures. Further, staff suggests caution should be used in evaluating the likely vegetation impacts associated with a given level of air quality expressed in terms of the 8-hr form in the absence of parallel SUM06 or W126 information.

Selecting a more biologically-relevant secondary standard form would also: 1) be directly responsive to the recommendation of the 2004 National Research Council's report titled *Air Quality Management in the United States* (NRC, 2004) which encourages the Agency to evaluate its historic practice of setting the secondary NAAQS equal to the primary; 2) provide support to important new Agency initiatives to enhance ecosystem-related program tracking and accountability; and 3) potentially spur more policy relevant vegetation effects research in the future.

8.3.4.2 Comparison of SUM06 and W126 Cumulative, Concentration-Weighted Forms

In addition to evaluating the 8-hr average form, we evaluated the appropriateness of the SUM06 alternative proposed in the last review by comparing it to another cumulative, concentration-weighted form discussed in the 1996 Staff Paper, the W126 index. In the 1996 Staff Paper, the preference for the SUM06 over other cumulative forms was based on the following science and policy considerations:

- All cumulative, peak-weighted exposure indices considered, including W126 and SUM06, were about equally good as exposure measures to predict exposure-response relationships reported in the NCLAN crop studies.

- The SUM06 form would not be influenced by PRB O_3 concentrations (defined at the time as 0.03 to 0.05 ppm) under many typical air quality distributions.

In the current review, we have reconsidered whether the SUM06 form should still be judged the most appropriate cumulative form for a secondary NAAQS protective of vegetation and ecosystems, based on the following:

- Model predictions of PRB in the range of 0.015 to 0.035 ppm for the current review are below the PRB range of 0.03 to 0.05 ppm described in the 1996 review, making PRB contributions much less of a factor influencing the choice of an appropriate cumulative index.

- There is no evidence in the extensive vegetation effects literature of a biological exposure threshold applicable across the broad array of O_3-sensitive species found growing in the U.S. The SUM06 index, with a threshold set at 0.06 ppm, artificially truncates exposures that have been shown to produce vegetation effects of concern given sufficient duration. The W126 index, on the other hand, cumulates all O_3 concentrations. However, because concentrations below 0.04 ppm, receive substantially smaller weights (3 percent or less), those concentrations within the range of PRB levels would not contribute significantly to the value of the index.

The CASAC Ozone Panel also views the 3-month growing season W126 index "…as a potentially more biologically-relevant index than the 3-month growing season SUM06 index. This is because the W126 index has no absolute minimum ozone concentration threshold and

only lightly weights the lower ozone concentrations. Therefore, a three-month seasonal W126 range that is the approximate equivalent of the SUM06 range of 10 to 20 ppm-hr is preferred" (Henderson, 2006c).

On the basis of the information highlighted above, staff concludes that the W126 form is a more appropriate biologically based and policy-relevant cumulative, concentration-weighted form and recommends the Administrator consider the W126 as a more appropriate form for the secondary standard. This recommendation is consistent with the views of CASAC. Given the legitimate policy interest in having a more stable standard form, the Administrator may want to give consideration to using a 3-year average of the 12-hr W126, in addition to consideration of an annual form.

8.3.5 Level of the Standard

The level at which a secondary standard should be set depends on a blending of science and policy judgments by the Administrator as to the level of air quality which is requisite to protect the public welfare from any known or anticipated adverse effects associated with the pollutant in the ambient air. The exposure and risk assessments conducted in Chapter 7 and summarized briefly above, provide information regarding the effects associated with a number of different welfare endpoints at different levels of air quality, often expressed in terms of both the current 8-hr average form and the W126 (or SUM06) seasonal form(s).

At the end of the last review, we identified a range for a 3-month, 12-hr SUM06 standard form of 25 to 38 ppm-hr, for the Administrator's consideration. These levels were estimated to allow 10% to 20% yield loss, respectively, to occur in no more than 50% of the studied NCLAN agricultural crop cases. These levels were also estimated to provide an increased level of protection for other categories of vegetation such as tree seedlings and mature trees in commercial, Class I, and other forested areas in urban, rural, and remote environments. It was recognized, however, that a standard set within this range would not protect the most sensitive species or individuals within a species from all potential effects related to O_3 exposures. The Administrator proposed the lower end of the range (e.g., 25 ppm-hr) as necessary to provide a requisite level of protection for vegetation against the adverse effects of O_3. Staff believes that this level is an appropriate upper bound for a range of levels recommended for consideration in this review, as it would continue to provide a level of crop and tree protection judged requisite by the Administrator in the last review. In addition, this level derives from the extensive and quantitative historic and recent crop effects database, as well as current staff exposure and risk analyses.

In identifying a lower bound for the range of alternative standard levels appropriate for consideration, staff concludes that several lines of evidence point to the need for greater

protection for tree seedlings, mature trees, and associated forested ecosystems. Tree growth is an important endpoint to consider because it can be related to other aspects of societal welfare such as sustainable production of timber and related goods, recreation, and carbon (CO_2) sequestration. Equally important, impacts on tree growth can also affect ecosystems through shifts in species composition and the loss of genetic diversity due to the loss of O_3 sensitive individuals or species. To help inform staff judgment about an appropriate level of protection to consider for trees, staff considered the results of a consensus-building workshop on the need for a long-term cumulative secondary O_3 standard. At this workshop, expert scientists expressed their judgments on what standard form(s) and level(s) would provide vegetation with adequate protection from O_3-related adverse effects. Consensus was reached with respect to selecting appropriate ranges of levels in terms of a 3-month, 12-hr SUM06 standard for a number of vegetation effects endpoints. These ranges are identified below, with the estimated approximate equivalent W126 shown in parentheses (See Appendix 7B for explanation of SUM06 to W126 equivalents). For yield reductions in agricultural crops – a range of 15 to 20 (13 to 17) ppm-hr; for growth effects to tree seedlings in natural forest stands – a range of 10 to 15 (7 to 13) ppm-hr; for growth effects to tree seedlings and saplings in plantations – a range of 12 to 16 (9 to 14) ppm-hr; and for visible foliar injury to natural ecosystems – a SUM06 range of 8 to 12 (5 to 9) ppm-hr (Heck and Cowling, 1997). In the 1997 final rule, the Administrator had pointed to the results of this workshop as providing important support to her view that the then current secondary standard was not adequately protective of vegetation, contributing to her rationale that revision of the secondary standard was needed (62 FR 38877)

In its October 24, 2006 letter to the Administrator, CASAC expressed its view regarding the appropriate form and range of levels for the Administrator to consider. The CASAC preferred a seasonal 3-month W126 standard in a range that is the approximate equivalent of the SUM06 at 10 to 20 ppm-hrs. Staff has determined that the approximate equivalent 3-month W126 range is 7 to 17 ppm-hrs. The lower end of this range (7 ppm-hr) is the same as the lower end of the range identified in the 1997 Consensus Workshop as protective of tree seedlings in natural forest stands from growth effects (Heck and Cowling, 1997).

Staff believes that O_3-related effects on forest tree species are an important public welfare effect of concern. Therefore staff concludes that it is appropriate to include as the lower bound of the recommended range, the lower end of the approximate range recommended by CASAC (Henderson, 2006c). Based on our analyses of risks of tree seedling biomass loss and mature tree growth reductions and on the basis of the scientific effects literature, we anticipate that the lower end of the range identified for the Administrator's consideration would substantially decrease the adverse effects of O_3 on forested ecosystems. Additionally, it is anticipated that the lower end of this range would provide increased protection from the more subtle impacts of O_3

acting in synergy with other natural and man-made stressors to adversely affect individual plants, populations and whole systems. By disrupting the photosynthetic process, decreasing carbon storage in the roots, increasing early senescence of leaves and affecting water use efficiency in trees, O_3 exposure could potentially disrupt or change the nutrient and water flow of an entire system. Weakened trees can become more susceptible to other environmental stresses such as pest and pathogen outbreaks or harsh weather conditions. Though it is not possible to quantify all the ecological and societal benefits associated with varying levels of alternative secondary standards, we conclude that this information should be weighed in considering the extent to which a secondary standard should be precautionary in nature in protecting against effects that have not yet been adequately studied and evaluated.

Based on all the above considerations, staff concludes that a 3-month, 12-hr W126 range of 21-7 ppm-hr is appropriate for consideration, with the upper bound equivalent to that proposed in the last review and with the lower bound being that recommended by CASAC.

In the absence of any information regarding a threshold of O_3 exposures for vegetation, staff recognizes that the level selected is largely a policy judgment as to the requisite level of protection needed. In determining the requisite level of protection for crops and trees, the Administrator will need to weigh the importance of the predicted risks of these effects in the overall context of public welfare protection, along with a determination as to the appropriate weight to place on the associated uncertainties and limitations of this information.

8.3.6 Summary of Staff Conclusions and Recommendations on the Secondary O_3 Standard

Staff conclusions and recommendations on the elements of the secondary O_3 standard for the Administrator's consideration in making decisions on the secondary O_3 standard are summarized below, together with supporting conclusions from sections 8.3.3 to 8.3.5 above. We recognize that selecting from among alternative policy options will necessarily reflect consideration of qualitative and quantitative uncertainties inherent in the relevant evidence and in the assumptions of the quantitative exposure and risk assessments. Any such standard should protect public welfare from any known or anticipated adverse effects associated with the presence of the pollutant in the ambient air. In recommending these options for consideration, we are mindful that the Act requires standards that, in the judgment of the Administrator, are requisite to protect public welfare. The standards are to be neither more nor less stringent than necessary.

In the last review, the Administrator took into account the following in reaching her final decision: 1) the varying degrees of protection afforded by the alternative primary standards recommended; 2) the incremental protection associated with alternative cumulative, seasonal

secondary standards under consideration; 3) the value of establishing a seasonal form for the secondary standard that is more representative of biologically relevant exposures; and 4) the extent to which a secondary standard should be precautionary in nature, given the possibility of O_3 impacts acting in synergy with other natural and manmade stressors to impact climate and other environmental endpoints, particularly given the potential significance at a regional scale and in Class I areas.

Staff notes that since the last review, several additional policy-relevant developments have occurred that may also warrant consideration by the Administrator when making decisions about what is requisite to protect public welfare. First, the Agency has undertaken a number of activities geared toward improving ecosystem-related program tracking and accountability and is currently engaged in efforts to identify relevant indicators for that purpose. Having a more biologically-relevant air quality index would allow the Agency to better track improvements in vegetation protection on the ground with specific program actions aimed at accomplishing that end. Second, the NRC report (described above) states: "Whatever the reason that led EPA to use identical primary and secondary NAAQS in the past, it is becoming increasingly evident that a new approach will be needed in the future. There is growing evidence that the current forms of the NAAQS are not providing adequate protection to sensitive ecosystems and crops" (NRC, 2004).

The following secondary standard recommendations encompass the breadth of policy-relevant considerations described above:

(1) It is appropriate to continue to use O_3 as the indicator for a standard that is intended to address effects associated with exposure to O_3, alone or in combination with related photochemical oxidants. Based on the available information, we conclude that there is no basis for considering any alternative indicator at this time.

(2) It is not appropriate to continue to use an 8-hr averaging time for the secondary O_3 standard. The 8-hr average form should be replaced with a cumulative, seasonal, concentration weighted form. Given the reasons stated in earlier discussions herein, staff concludes that the W126 form is more appropriate than the SUM06 form recommended in the last review.

(3) It is appropriate to consider the maximum consecutive 3 month period within the O_3 season as the seasonal averaging time over which to cumulate hourly O_3 exposures for the daily 12-hr daylight (8 am to 8 pm) window. Though the length of time in the growing season varies significantly between species, staff concludes that the 3-month

period of maximum O_3 exposure generally coincides with maximum biological activity for most plants. Staff notes that for certain welfare effects of concern (e.g., foliar injury, yield loss for annual crops, growth effects on other annual vegetation and potentially tree seedlings), an annual standard form may be more appropriate, while for other welfare effects (e.g., mature tree biomass loss), a 3-year average form may be more appropriate. Staff concludes it is appropriate to consider both the annual and 3-year average forms.

(4) It is appropriate to consider a range of levels when making a determination regarding what is requisite public welfare protection. Staff concludes that an appropriate upper bound of this range is 21 ppm-hrs, expressed in terms of the W126 index, which is roughly equivalent to that proposed by the Administrator in the last review as able to provide a requisite level of protection to vegetation. Our analyses indicate that this level will provide protection against O_3-related adverse impacts on vegetation such as tree growth and crop yield beyond that afforded by the current 8-hr standard. In large part, the basis for selecting the level in the last review was a judgment as to what was an appropriate level of protection against annual crop yield loss. Though crop data are still useful as a potential indicator of risk to other sensitive annual herbaceous plants, staff recognizes that agricultural systems are heavily managed. In addition, the annual productivity of agricultural systems is vulnerable to disruption from many other stressors (e.g., weather, insects, disease), whose impact in any given year can greatly outweigh the direct reduction in annual productivity resulting from elevated O_3 exposures. On the other hand, O_3 can also more subtly impact crop and forage nutritive quality and indirectly exacerbate the severity of the impact from other stressors. These latter effects cannot currently be quantified and deserve further study. Taking all of the above considerations into account, staff concludes that from a public welfare perspective, greater concern should be placed on the impacts of O_3 exposures on vegetation in less heavily managed and unmanaged ecosystems such as tree seedlings, mature trees, and forested ecosystems in general. Thus, staff concludes that the lower end of the range should incorporate the lower end of the range expressed by CASAC of a 3-month 12-hr W126 approximately equal to 7 ppm-hrs. This lower level will increase protection for the most sensitive tree species and the ecosystems where they are found.

Several additional factors should be considered when selecting an appropriate level for a secondary standard. These include 1) the fact that O_3 effects are cumulative and have been shown to have carry over effects from one year to the next; 2) some seedling tree species have sensitivities as great as annual crops and the importance of protecting

against small percentages of biomass loss on an annual basis has been expressed by some within the scientific community; 3) visible foliar injury impacts can occur within a growing season at very low levels of O_3 exposure; and 4) the extent to which a secondary standard should be precautionary in nature, given the possibility of O_3 impacts acting in synergy with other natural and manmade stressors. Should a 3-year average of a 12-hr W126 be selected, the level chosen should reflect the fact that annual impacts are still a concern for visible foliar injury, tree seedling biomass loss, and crop yield loss, so that a potentially lower level might be considered to reduce the potential of adverse impact from the single high O_3 year that could still occur while attaining a 3-year average.

8.4 SUMMARY OF KEY UNCERTAINTIES AND RESEARCH RECOMMENDATIONS RELATED TO SETTING A SECONDARY O_3 STANDARD

Staff has identified the following key uncertainties and research questions that have been highlighted in Chapter 9 of the CD and Chapter 7 herein, associated with this review of the welfare-based secondary standards. The first set of key uncertainties and research recommendations discussed below is that associated with the extrapolation to plant species and environments outside of specific experimental or field study conditions. The second set of key uncertainties and research recommendations pertain to our ability to assess the impact of O_3 on other welfare effects categories such as climate, ecosystem components such as wildlife, and whole ecosystem structure and function. Third, we identify research recommendations related to the development of approaches, tools, or methodologies useful in characterizing the relationship between O_3 and public welfare in a policy context. These three areas are described below.

(1) Plant Species-Level Research Needs:

- To reduce uncertainties associated with extrapolating plant response for a given level of O_3 using composite response functions across differing regions and climates, studies using large numbers of plant species across regions where those species are indigenous are recommended. In addition, to better understand the full range of response of plant species to O_3, research on more species is recommended.

- To reduce uncertainty associated with estimating the risk to vegetation of differing amounts of O_3-induced visible foliar injury over the plant's leaf area, research to explore the relationship between visible foliar injury and other O_3-related effects is recommended.

- To reduce uncertainty associated with the impact of differing levels of O_3 on the nutritive quality of forage and other crops, additional research is needed.

- To reduce uncertainty associated with estimated or modeled flux into plants, research is recommended to evaluate the factors that affect O_3 flux into plants, including the genetic determinants of O_3 sensitivity and the range of variability among species with respect to detoxification/compensation and nocturnal uptake and response. Research that explores the relative importance of flux rate versus total cumulative flux or dose, and that leads to a database of O_3 flux-response relationships for vegetation; similar to the extensive concentration-response database that currently exists is recommended to further reduce existing uncertainties.

- To reduce uncertainties in extrapolating from O_3 effects on juvenile to mature trees and from trees grown in the open versus those in a closed forest canopy in a competitive environment, additional research is recommended.

- To reduce uncertainties in extrapolating individual plant response spatially or to higher levels of biological organization, including ecosystems, research that explores and better quantifies the nature of the relationship between O_3, plant response and multiple biotic and abiotic stressors, including those associated with climate change, is recommended.

(2) Ecosystem Level Impacts:

- To reduce uncertainties associated with projections of the effects of O_3 on the ecosystem processes of water, carbon, and nutrient cycling, particularly at the stand and community levels, research is needed on the effects on below ground ecosystem processes in response to O_3 exposure alone and in combination with other stressors. These below ground processes include interactions of roots with the soil or microorganisms, effects of O_3 on structural or functional components of soil food webs and potential impacts on plant species diversity, changes in the water use of sensitive trees, and if the sensitive tree species is dominant, potential changes to the hydrologic cycle at the watershed and landscape level.

- To conclusively show whether O_3 affects biodiversity or genetic diversity, research on competitive interactions under elevated O_3 levels are recommended. This research could be strengthened by modern molecular methods to quantify impacts on diversity.

- To fill the data gaps regarding interactions and potential feedback mechanisms between O_3 and O_3 precursor (e.g., volatile organic carbons) production, atmospheric processes, and climate change variables, research is recommended to evaluate whether O_3 will negate the positive effects of an elevated CO_2 environment on plant carbon and water balance, whether the likelihood of various biotic stressors such as pest epidemics and insect outbreaks would be expected to increase in the future

- To reduce uncertainties associated with scaling O_3 effects up from the responses of single or a few plants to effects on communities and ecosystems, additional research is recommended. Because these uncertainties are multiple and significant due to the complex interactions involved, new research will likely require a combination of manipulative experiments with model ecosystems, community and ecosystem studies

along natural O_3 gradients, and extensive modeling efforts to project landscape-level, regional, national and international impacts of O_3.

- To fill the data gaps regarding O_3 impacts to other non-plant welfare effects categories such as climate, as well as potential direct impacts of O_3 on some sensitive species of animals and wildlife, more research is needed.

(3) Approaches, Tools, Methodologies:

- To reduce uncertainties associated with valuing improved vegetation and ecosystem function from improved O_3 air quality, research is needed on methodologies to determine the values associated with important services and benefits derived from natural ecosystems such that these could be used in comprehensive risk and benefits assessments for O_3 effects on natural ecosystems.

- To reduce uncertainties associated with evaluating the performance of different exposure indices given different patterns of O_3 exposures, experiments would need to be designed to specifically test the performance of different indices in predicting plant response under different exposure regimes.

- To reduce uncertainties associated with the generation of rural O_3 exposures, improved model capabilities are needed, including a more refined spatial grid for the western U.S., better handling of O_3 movement in complex terrain and predicting nocturnal concentrations. Further, research is needed regarding whether strategic placement of passive or mobile monitors might benefit the estimation of impact to particular resources of concern.

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ATTACHMENT A

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON D.C. 20460

OFFICE OF THE ADMINISTRATOR
SCIENCE ADVISORY BOARD

October 24, 2006

EPA-CASAC-07-001

Honorable Stephen L. Johnson
Administrator
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue, NW
Washington, DC 20460

Subject: Clean Air Scientific Advisory Committee's (CASAC) Peer Review of the Agency's 2nd Draft Ozone Staff Paper

Dear Administrator Johnson:

EPA is in the process of reviewing the national ambient air quality standards (NAAQS) for ozone (O_3) and related photochemical oxidants, which the Agency most recently revised in July 1997. As part of its ongoing review of the ozone NAAQS, EPA's Office of Air Quality Planning and Standards (OAQPS) developed a 2nd Draft Ozone Staff Paper, entitled, *Review of the National Ambient Air Quality Standards for Ozone: Policy Assessment of Scientific and Technical Information* (July 2006). At the request of the Agency, EPA's Clean Air Scientific Advisory Committee (CASAC or Committee), supplemented by subject-matter-expert panelists — collectively referred to as the CASAC Ozone Review Panel (Ozone Panel) — met in a public meeting in Durham, NC, on August 24-25, 2006, to conduct a peer review of this draft Ozone Staff Paper and three related draft technical support documents.

In its summary of EPA staff conclusions on the primary (health-related) ozone NAAQS found in Chapter 6 of the 2nd Draft Ozone Staff Paper, OAQPS set-forth two options with regard to revising the level and the form of the standard: (1) retain the current primary eight-hour (8-hr) NAAQS of 0.08 parts per million (ppm); or (2) consider a reduction in the level of the primary O_3 NAAQS within the range of alternative 8-hr standards included in Staff's exposure and risk assessments (which included a range from 0.064 to 0.084 ppm) with primary focus on an O_3 level of 0.07 ppm with a range of forms from third- through fifth-highest daily maximum. The Ozone Panel unanimously concludes that:

1. There is no scientific justification for retaining the current primary 8-hr NAAQS of 0.08 parts per million (ppm), and

2. The primary 8-hr NAAQS needs to be substantially reduced to protect human health, particularly in sensitive subpopulations.

Therefore, *the CASAC unanimously recommends a range of 0.060 to 0.070 ppm for the primary ozone NAAQS*. With regard to the secondary (welfare-related) ozone NAAQS, t*he Ozone Panel is in strong agreement* with the scientific and technical evidence presented in the summary of EPA staff conclusions on the secondary ozone NAAQS found in Chapter 8 of the draft Staff Paper *in support of the alternative secondary standard of cumulative form that extends over an entire growing season.*

The Ozone Panel members agree that this letter adequately represents their views. The chartered Clean Air Scientific Advisory Committee fully endorses the Panel's letter and hereby forwards it to you as the Committee's consensus report on this subject. A discussion of each chapter in the 2nd Draft Ozone Staff Paper follows this letter, and the comments of individual Panel members on the 2nd Draft Ozone Staff Paper and three related draft technical support documents are attached as Appendix D.

1. Background

Section 109(d)(1) of the CAA requires that the Agency periodically review and revise, as appropriate, the air quality criteria and the NAAQS for the "criteria" air pollutants, including ambient ozone. Pursuant to sections 108 and 109 of the Act, EPA is in the process of reviewing the ozone NAAQS. OAQPS, within the Office of Air and Radiation (OAR), developed the 2nd Draft Ozone Staff Paper as part of this activity. In February 2006, the Agency's National Center for Environmental Assessment, Research Triangle Park, NC (NCEA-RTP), within the Agency's Office of Research and Development (ORD), released its final *Air Quality Criteria for Ozone and Related Photochemical Oxidants*, *Volumes I, II, and III*, (EPA/600/R-05/004aF-cF, Final Ozone Air Quality Criteria Document) for this current review cycle for the ozone NAAQS. The 2nd Draft Ozone Staff Paper evaluates the policy implications of the key scientific and technical information contained in the Final Ozone AQCD and identifies critical elements that the Agency believes should be considered in its review of the ozone NAAQS. The Ozone Staff Paper is intended to "bridge the gap" between the scientific review contained in the Ozone AQCD and the public health and welfare policy judgments required of the EPA Administrator in reviewing the ozone NAAQS.

The Ozone Panel met in a public meeting on December 8, 2005 to conduct a consultation on EPA's 1st Draft Ozone Staff Paper and two related technical support documents. However, given that the OAQPS' first draft Staff Paper did not contain Agency staff conclusions about whether to retain or revise the existing primary and secondary Ozone standards, the CASAC's activity only amounted to a technical assessment of that document. The Committee's letter to you from that meeting (EPA-CASAC-CON-06-003), dated February 16, 2006, is posted at URL: http://www.epa.gov/sab/pdf/casac_con_06_003.pdf.

2. CASAC Ozone Review Panel's Peer Review of the 2nd Draft Ozone Staff Paper and Related Technical Support Documents

The Ozone Panel reviewed the 2nd Draft Ozone Staff Paper and found it improved over the earlier version that had been reviewed as part of a consultation process. *However, the Panel did not agree with the EPA staff conclusions that it was appropriate to consider retaining the current NAAQS as an option that would be protective of public health and welfare.* The Ozone Panel's recommendations for reducing the level of the primary ozone standard, and its rationale for these recommendations, are provided immediately below. Following a detailed discussion on the primary and secondary NAAQS are the Panel's major, chapter-specific comments. Finally, the individual written comments of Ozone Panel members on the 2nd Draft Ozone Staff Paper and the three related draft technical support documents are attached in Appendix D. Panelists' responses to the Agency's charge questions are included in these individual review comments.

Primary Ozone NAAQS

New evidence supports and build-upon key, health-related conclusions drawn in the 1997 Ozone NAAQS review. Indeed, in the 2nd Draft Ozone Staff Paper, EPA staff themselves arrived at this same conclusion:

> "Based on the above considerations and findings from the [Final Ozone AQCD], while being mindful of important remaining uncertainties, staff concludes that the newly available information generally reinforces our judgments about causal relationships between O_3 exposure and respiratory effects observed in the last review and broadens the evidence of O_3 -related associations to include additional respiratory-related endpoints, newly identified cardiovascular-related health endpoints, and mortality. Newly available evidence also has identified increased susceptibility in people with asthma. While recognizing that important uncertainties and research questions remain, we also conclude that progress has been made since the last review in advancing our understanding of potential mechanisms by which ambient O_3, alone and in combination with other pollutants, is causally linked to a range of respiratory- and cardiovascular-related health endpoints." (Pages 6-6 and 6-7)

Several new single-city studies and large multi-city studies designed specifically to examine the effects of ozone and other pollutants on both morbidity and mortality have provided more evidence for adverse health effects at concentrations lower than the current standard. (See the numerous ozone epidemiological single-city studies shown in Figure 3-4 on page 3-53 of the 2nd Draft Staff Paper and, in addition, Appendix 3B of the staff paper, which contains the summary of effect estimates and air quality data for these studies and multi-city epidemiological studies.) These studies are backed-up by evidence from controlled human exposure studies that also suggest that the current primary ozone NAAQS is not adequate to protect human health (Adams, 2002; McDonnell, 1996).

Furthermore, we have evidence from recently reported controlled clinical studies of healthy adult human volunteers exposed for 6.6 hours to 0.08, 0.06, or 0.04 ppm ozone, or to filtered air alone during moderate exercise (Adams, 2006). Statistically-significant decrements in lung function were observed at the 0.08 ppm exposure level. Importantly, adverse lung function effects were also observed in some individuals at 0.06 ppm (Adams, 2006). *These*

results indicate that the current ozone standard of 0.08 ppm is not sufficiently health-protective with an adequate margin of safety. It should be noted these findings were observed in healthy volunteers; similar studies in sensitive groups such as asthmatics have yet to be conducted. However, people with asthma, and particularly children, have been found to be more sensitive and to experience larger decrements in lung function in response to ozone exposures than would healthy volunteers (Mortimer *et al.*, 2002).

Going beyond spirometric decrements, adverse health effects due to low-concentration exposure to ambient ozone (that is, below the current primary 8-hour NAAQS) found in the broad range of epidemiologic and controlled exposure studies cited above include: an increase in school absenteeism; increases in respiratory hospital emergency department visits among asthmatics and patients with other respiratory diseases; an increase in hospitalizations for respiratory illnesses; an increase in symptoms associated with adverse health effects, including chest tightness and medication usage; and an increase in mortality (non-accidental, cardiorespiratory deaths) reported at exposure levels well below the current standard. *The CASAC considers each of these findings to be an important indicator of adverse health effects.* As demonstrated in Chapter 5 of the 2nd Draft Ozone Staff Paper (specifically, Figures 5.5, 5.7, 5.8, and 5.9), a significant decrease in adverse effects due to ozone exposures can be achieved by lowering the exposure concentrations below the current standard, which is effectively 0.084 ppm. Beneficial effects in terms of reduction of adverse health effects were calculated to occur at the lowest concentration considered (*i.e.*, 0.064 ppm). (See also Figure 3-4, "Effect estimates (with 95% confidence intervals) for associations between short-term ozone exposure and respiratory health outcomes," on page 3-53.)

The justification provided in the 2nd Draft Ozone Staff Paper for retaining the current level of the primary ozone standard as an option for the Administrator was based on results of controlled human exposure studies measuring modest declines in FEV_1 after exposures to 0.08 ppm ozone. However, as stated in the Staff Paper (page 3-6), while average decrements in the FEV_1 were relatively small, 26% of the subjects had greater than 10% decrements, which can be clinically significant. Also, while measures of FEV_1 are quantitative and readily obtainable in humans, they are not the only measures — and perhaps not the most sensitive measures — of the adverse health effects induced by ozone exposure. As stated on page 6-32 of the Final Ozone AQCD, "Spirometric responses to ozone are independent from inflammatory responses and markers of epithelial injury (Balmes *et al.*, 1996; Bloomberg *et al.*, 1999; Hazucha *et al.*, 1996; Torres *et al.*, 1997). Significant inflammatory responses to ozone exposures that did not elicit significant spirometric responses have been reported (Holz *et al.*, 2005; McBride *et al.*, 1994)." Agency staff's analyses placed most emphasis on spirometric evidence and not enough emphasis on serious morbidity (*e.g.*, hospital admissions) and mortality observed in epidemiology studies (see page 6-44).

Therefore, on the basis of the large amount of recent data evaluating adverse health effects at levels at and below the current NAAQS for ozone, it is the unanimous opinion of the CASAC that the current primary ozone NAAQS is not adequate to protect human health. Furthermore, the Ozone Panel is in complete agreement both that: the EPA staff conclusion in Section 6.3.6 arguing that "consideration could be given to retaining the current 8-hr ozone standard" is not supported by the relevant scientific data; and that the current primary 8-hr

standard of 0.08 ppm needs to be substantially reduced to be protective of human health, particularly in sensitive subpopulations.

Additionally, we note that the understanding of the associated science has progressed to the point that *there is no longer significant scientific uncertainty regarding the CASAC's conclusion that the current 8-hr primary NAAQS must be lowered.* A large body of data clearly demonstrates adverse human health effects at the current level of the 8-hr primary ozone standard. Retaining this standard would continue to put large numbers of individuals at risk for respiratory effects and/or significant impact on quality of life including asthma exacerbations, emergency room visits, hospital admissions and mortality. (Scientific uncertainty does exist with regard to the lower level of ozone exposure that would be fully-protective of human health. The Ozone Panel concludes that it is possible that there is no threshold for an ozone-induced impact on human health and that some adverse events may occur at policy-relevant background.)

Moreover, EPA staff concluded that changes in the concentration-based form of the standard (*i.e.*, whether to use the third-, fourth-, or fifth-highest daily maximum 8-hr average concentration) should also be considered. The analysis found in the 2nd Draft Ozone Staff Paper indicates that modest changes in the form of the standard can have substantial impacts on the frequency of adverse health effects. Therefore, the CASAC recommends that the Agency conduct a broader evaluation of alternative concentration-based forms of the primary 8-hr ozone standard and the implications of those alternative forms on public-health protection and stability (*i.e.*, with respect to yearly variability to ensure a stable target for control programs).

The CASAC further recommends that the ozone NAAQS should reflect the capability of current monitoring technology, which allows accurate measurement of ozone concentrations with a precision of parts per *billion*, or equivalently to the third decimal place on the parts-per-million scale. In addition, given that setting a level of the ozone standard to only two decimal places inherently reflects upward or downward "rounding," *e.g.*, 0.07 ppm includes actual measurements from 0.0651 ppm to 0.0749 ppm, the CASAC chooses to express its recommended level, immediately below, to the third decimal place.

Accordingly, the CASAC unanimously recommends that the current primary ozone NAAQS be revised and that the level that should be considered for the revised standard be from 0.060 to 0.070 ppm, with a range of concentration-based forms from the third- to the fifth-highest daily maximum 8-hr average concentration. While data exist that adverse health effects may occur at levels lower than 0.060 ppm, these data are less certain and achievable gains in protecting human health can be accomplished through lowering the ozone NAAQS to a level between 0.060 and 0.070 ppm.

Secondary Ozone NAAQS

An important difference between the effects of acute exposures to ozone on human health and the effects of ozone exposures on welfare is that vegetation effects are more dependent on the *cumulative* exposure to, and uptake of, ozone over the course of the entire growing season (defined to be a minimum of at least three months). *Therefore, there is a clear need for a*

secondary standard which is distinctly different from the primary standard in averaging time, level and form. Developing a biologically-relevant ozone air quality index would be directly responsive to the 2004 National Research Council (NRC) recommendations on Air Quality Management in the United States (NAS, 1994) and will help support important new Agency initiatives to enhance ecosystem-related program tracking and accountability.

In its 1996 review of the ozone NAAQS, EPA staff proposed several cumulative seasonal ozone exposure indices, including SUM06, the concentration-weighted metric (*i.e.*, the seasonal sum of all hourly average concentrations > 0.06 ppm), and W126, the integrated exposure index with a sigmoidal weighting function, as candidates for a secondary standard. The Administrator considered a three-month, 12-hr SUM06 secondary standard at a level of 25 ppm-hr as an appropriate, biologically-relevant secondary standard, but ultimately rejected this option in favor of simply setting the secondary standard equal to the primary. It was rationalized that efforts to attain the new 8-hr primary standard would also eliminate most adverse effects on vegetation, and at that time there were uncertainties in how cumulative seasonal exposures would change with efforts to reduce peak 8-hour concentrations. Additionally, it was assumed that future ozone/vegetation effects research over the coming years would clarify the very uncertain quantitative relationships between ozone exposures and vegetation/ecological responses under ambient field conditions.

Unfortunately, however, the Agency has supported very little new vegetation/ecological ozone effects research over the past decade. The net result is that the quantitative evidence linking specific ozone concentrations to specific vegetation/ecological effects must continue to be characterized as having high uncertainties due to the lack of data for verification of those relationships. It is not surprising that substantial research needs remain, as indicated both in Chapter 8 and in individual reviewer comments. The quantitative evidence linking specific ozone concentrations to specific vegetation effects — especially at the complex ecosystem level — must continue to be characterized as having high uncertainties due to the lack of data for verification of those relationships. To a large extent, this is an unavoidable consequence of the inherent complexities of ecosystem structure and function, interactions among biotic and abiotic stressors and stimuli, variability among species and genotype, detoxification and compensatory mechanisms, *etc.* Nevertheless, the compelling weight of evidence provided in Chapter 7 of the 2nd Draft Ozone Staff Paper results from the convergence of results from many various and disparate assessment methods including chamber and free air exposure, crop yield and tree seedling biomass experimental studies, foliar injury data from biomonitoring plots, and modeled mature tree growth.

Despite limited recent research, it has become clear since the last review that adverse effects on a wide range of vegetation including visible foliar injury are to be expected and have been observed in areas that are below the level of the current 8-hour primary and secondary ozone standards. Such effects are observed in areas with seasonal 12-hr SUM06 levels below 25 ppm-hr (the lower end of the range of a SUM06 secondary standard suggested in the 1996 review and the upper end of the range suggested in Chapter 8 of the 2nd Draft Ozone Staff Paper). Seasonal SUM06 (or equivalent W126) ranges well below 25 ppm-hr were recommended for protecting various managed and unmanaged crops and tree seedlings in the 1997 workshop on secondary ozone standards (Heck and Cowling, 1997). The absence of clear-

cut lower effects thresholds for sensitive vegetation combined with the lower recent estimates of policy-relevant background (typical range of 0.015 to 0.035 ppm) emphasizes the importance of efforts to reduce low- to mid-range environmental exposures below 0.060 ppm.

Based on the Ozone Panel's review of Chapters 7 and 8, *the CASAC unanimously agrees that it is* not *appropriate to try to protect vegetation from the substantial, known or anticipated, direct and/or indirect, adverse effects of ambient ozone by continuing to promulgate identical primary and secondary standards for ozone. Moreover, the members of the Committee and a substantial majority of the Ozone Panel agrees with EPA staff conclusions and encourages the Administrator to establish an alternative cumulative secondary standard for ozone and related photochemical oxidants that is distinctly different in averaging time, form and level from the currently existing or potentially revised 8-hour primary standard.* The suggested approach to the secondary standard is a cumulative seasonal growing standard such as the indices SUM06 or W126 aggregated over at least the three summer months exhibiting the highest cumulative ozone levels and includes the ozone exposures from at least 12 daylight hours. The CASAC suggests a range of 10 to 20 ppm-hours for the three-month growing season SUM06 index for agricultural crops rather than the 15-25 ppm-hours proposed in Chapter 8.

However, the Ozone Panel views the three-month growing season W126 index as a potentially more biologically-relevant index than the 3-month growing season SUM06 index. This is because the W126 index has no absolute minimum ozone concentration threshold and only lightly weights the lower ozone concentrations. Therefore, a three-month seasonal W126 that is the approximate equivalent of the SUM06 at 10 to 20 ppm-hr is preferred. As shown by the references cited at the end of Chapter 8, the consensus view among expert persons in the ecological communities of both this country and elsewhere around the world is that *a secondary standard of cumulative form and extending over an entire growing season will be far more effective than a secondary standard that is not cumulative in form and does not include the whole growing season.*

In conclusion, the Clean Air Scientific Advisory Committee is pleased to provide its scientific advice and recommendations to the Agency on the primary and secondary ozone NAAQS. We recognize that our recommendation of lowering of the current primary ozone standard would likely result in a large portion of the U.S. being in non-attainment. *Nevertheless, we take very seriously the statutory mandate in the Clean Air Act not only for the Administrator to establish, but also for the CASAC to recommend to the Administrator, a primary standard that provides for an "adequate margin of safety … requisite to protect the public health."*

Finally, as announced during the Ozone Panel's August meeting, once the Agency releases the Final Ozone Staff Paper in early January 2007, the CASAC intends to hold a public teleconference in late January or early February 2007 for the members of the Ozone Panel to review — and, prospectively, to offer additional, unsolicited advice to the Agency concerning — Chapter 6 (Staff Conclusions on Primary O_3 NAAQS) and Chapter 8 (Staff Conclusions on Secondary O_3 NAAQS) in that final Agency document. The purpose of such advice would be to

inform EPA's efforts as it develops the forthcoming, proposed rule for ozone and related photochemical oxidants. As always, the CASAC wishes EPA well in this important endeavor.

Sincerely,

/Signed/

Dr. Rogene Henderson, Chair
Clean Air Scientific Advisory Committee

Appendix A – Clean Air Scientific Advisory Committee Roster (FY 2006)

Appendix B – CASAC Ozone Review Panel Roster

Appendix C – Charge to the CASAC Ozone Review Panel

Appendix D – Review Comments from Individual CASAC Ozone Review Panel Members

CASAC Chapter-Specific Discussion Comments on EPA's 2nd Draft Ozone Staff Paper

Sub-groups of the CASAC Ozone Review Panel who led the discussion on individual chapters of the Staff Paper summarized their comments in the following paragraphs:

Chapter 2 (Air Quality Characterization): A better introduction to the central role of photochemical oxidation reactions as the key reactions governing the behavior of air pollutants in the atmosphere would improve this chapter. Ozone is the key indicator of the extent of oxidative chemistry and serves to integrate multiple pollutants. Oxidation in the atmosphere leads to the formation of particulate matter from SO_2, NO_X, and volatile organic compounds (VOCs) as well as gas phase irritants (formaldehyde, acrolein, etc). Thus, although ozone itself has direct effects on human health and ecosystems, it can also be considered as indicator of the mixture of photochemical oxidants and of the oxidizing potency of the atmosphere. Section 2.2.6 only briefly covers the relationship of ozone to other photochemical oxidants. It would be beneficial to add a short paragraph outlining the role of ozone and other photochemical oxidants in the atmospheric transformation processes that may results in the formation of more toxic products (both in an outdoor and indoor environment), as provided in the individual comments appended to this letter.

The section on policy-relevant background (2.7) continues to have problems. Although the section briefly cites the results of comparison of different models and measurements, it does not adequately address the uncertainties of the global GEOS-CHEM model, and how these uncertainties are reflected in the health risk analysis. Since ozone health effects are observed down to concentrations of the order of 0.04–0.05 ppm, it is important to know how the PRB is related to the considered primary ozone standard and what uncertainties there are in the risk attributed to controllable sources.

Chapter 3 (Policy-Relevant Assessment of Health Effects Evidence): The latest draft of Chapter 3 is much improved over the previous draft. Efforts to respond to some of the earlier concerns expressed by the CASAC are appreciated. While in general this chapter is well written, and is a credible basis for the risk analyses that follow, there are inconsistencies and inaccuracies that still need to be addressed. Typically, there is appropriate use of cautionary phrases when the data are not as strong as they might be, but this use is inconsistent across the chapter, and there are instances where EPA staff appear to be stretching to infer that data support their statement. While the individual comments of Ozone Panel members attached to this letter provide specifics on these points, some of the Panel's more significant concerns are discussed briefly below.

Discussion of measurement error is convoluted, confusing, and contains some mistakes. The primary issue in the use of central ambient monitors for ozone in time-series epidemiological studies is whether they provide any information at all that reflects daily personal ozone exposure in susceptible populations. The discussion on p. 3-37 of the impact of various types of exposure measurement error is incorrect; the difference between true and measured ambient concentrations is an example of classical measurement error that results in bias of effect

estimates to the null, not just an increase in standard error. Claiming that the difference between average personal exposure and ambient concentrations results in "attenuation of risk" is not appropriate.

The Ozone Panel does not completely agree with staff's conclusion that "the use of routinely monitored ambient ozone concentrations as a surrogate for personal exposures is not generally expected to change the principal conclusions from ozone epidemiological studies." Indeed, Panel members have little insight as to what we would find if we had actual exposure measurements. Personal exposures most likely correlate better with central site values for those subpopulations that spend a good deal of time outdoors, which coincides, for example, with children actively engaged in outdoor activities, and which happens to be a group that the ozone risk assessment focuses upon.

Some statements about which individuals are at greatest risk of ozone-induced effects are not adequately supported by the information discussed in the chapter. Individuals with chronic obstructive pulmonary disease (COPD) and cardiovascular disease (CVD) are likely to be at increased risk, but the hypothesis that such "hyper-responsiveness" can be used to identify individuals with COPD or CVD who are at greatest risk of O_3-induced health effects has not been confirmed. A more appropriate conclusion would be that individuals with COPD and CVD are at increased risk of O_3-induced health effects.

The discussion of the ranges for changes in FEV_1 that are considered to be small, moderate, or large for persons with impaired respiratory systems is not consistent. While EPA staff state that the table values for the ranges do not need to be changed, staff indirectly acknowledge that a 10% reduction in this variable in asthmatics could have serious consequences, an interpretation that is used in Chapters 4-6.

The 30 subjects studied by Adams had a great influence on the analyses presented in Chapters 5 and 6. While the discussion of the low-level exposures used in the controlled human studies by Adams and colleagues is technically correct that no statistically significant changes were found in FEV_1 for ozone at 40 to 60 ppb compared to filtered air, there were clearly a few individuals who experienced declines in lung function at these lower concentrations. These were healthy subjects, so the percentage of asthmatic subjects, if they had been studied, would most likely be considerably greater.

The lack of statistical power is consistently offered in Chapter 3 for why there appears to be an inconsistent effect seen for COPD mortality. Coherence of respiratory effects for ozone suffers from neither no more nor no less power considerations that do those for particulate matter (PM). Yet the Agency did not argue a lack of power when assessing PM risks, so consistency is needed here relative to ozone effect estimates for COPD mortality.

The relatively strong and relatively consistent effect of ozone on emergency department visits for respiratory disease, especially asthma, as evidenced in Figure 3-4 is misrepresented in several places in the Chapter (and in Chapters 5 and 6) as "inconclusive" or "inconsistent." This should be corrected.

Chapter 4 (Characterization of Human Exposure to Ozone): The second draft of Chapter 4 has responded to many of the comments made on the first draft, and is thus clearer than before. The panel was pleased to see the reanalysis for 2002 in addition to 2004.

It would be helpful to have the estimated exposures for current (2002 and 2004) levels displayed in Tables 4-8 & 4-9 (p. 4-32) and Figures 4-4 to 4-21 (pp. 4-33 to 4-41), in addition to only those for just meeting the current standard and alternative more stringent standards. This would be analogous to the way estimated effects are displayed in Chapter 5 (Figures 5-5 to 5-9 [pp.5-58 to 5-65]).

On the whole, Chapter 4 provides a clear "road map" for what was done to characterize available knowledge about human exposure to ozone in the framework of generally accepted modeling approaches of appropriately selected populations in 12 urban areas of the U.S. Much of the text reads like a basic textbook on human exposure assessment using state-of-the-art modeling approaches, such as the Air Pollutants Exposure Model (APEX), including adjustments for lung ventilation of delivered ozone dose. This extension, beyond exposure characterization, is particularly important for ozone where the extent of measurable human responses is very sensitive to the amount of ozone inhaled and to where it deposits along the respiratory tract. Further extension of the methodology to estimate dose would have important implications and should be discussed.

There is an explicit discussion of the limitations of the APEX model in terms of variability and the quality of the input data, which is appropriate and fine as far as it goes. There are good reasons presented for selection of urban areas and the time periods to be modeled. However, there was inadequate consideration of the populations selected for modeling. Those selected were appropriate, but the omission of the elderly, the population most at risk for ozone-associated premature daily mortality, was notable and not even mentioned in terms of why it was not considered.

The chapter was very good at exposition and clear presentation of modeling results, but was deficient it its discussion of seemingly counterintuitive results, and of a potentially large influence of measurement biases. As an example of the first of these issues, the children in LA & Houston are estimated to have far fewer exposures above 0.07 ppm (8-hr) than in most other cities with lower ozone concentrations and fewer children. This was likely due to the greater within-day and sampler-to-sampler variations in concentration within these two cities than in the others, the fact that the entire year was modeled while for other sites the winter was not included and/or the greater extent of air conditioning, especially in Houston. Whatever the reasons, there should have been some discussion of the causes. The quadratic rollback methodology should have been better described since this strategy has important consequences for the modeled results.

The second issue that was presented, but left hanging without an adequate discussion is at the bottom of page 4-47, where it was simply stated that "in general, APEX systematically under-predicts the measured values by 0.001 to 0.02 ppm (zero to 50 percent)." If this is so, is it due to a really serious failure of the APEX model, or to unreliable measurements? The measurements at issue were six-day average concentrations based on the use of passive

(diffusion) samplers, which are known to be subject to significant errors when the air velocity across the inlet is variable. The comparison of measured and modeled concentrations depicted in Figure 4-22 is certainly worthy of further analysis and discussion.

Chapter 5 (Characterization of Health Risks): Generally the panel found Chapter 5 and its accompanying risk assessment to be well done, balanced and reasonably communicated. Additional text is needed at the beginning and end of the chapter to put the limited risk assessment into the context of the much larger body of evidence of ozone health effects. The discussion of uncertainty in these risk estimates is expanded in section 5.3.2.5. Although a number of issues are raised, their impacts on the estimates have not been thoroughly explored. Additional sensitivity analyses seem warranted. In particular, it is essential that the sensitivity of the risk assessment to the shape of the dose-response curve for FEV_1 be evaluated. Although the 3 parameter logistic (3PL) model emulates the pattern seen in the five "data points," these points are aggregates of the original data, and may give a misleadingly optimistic picture of the quality of the fit. More importantly, although the problem of model uncertainty is noted it has not been addressed even though methods exist for doing so. Even if only the linear and logistic models were included in the analysis, the error bands around the estimated response probabilities would likely increase to better reflect that uncertainty. In addition, a suggestion to deal with the uncertainties surrounding estimation of PRB, particularly as related to Table 5.5 (for lung function) and Table 5.11 (mortality), would be to change the form of the analyses to assess the impact of the concentration change in the expected number of health effects relative to the current standard. The key advantage of estimating the effect of concentration change is that it does not depend on the choice of the PRB.

With regard to the controlled human exposure studies, Ozone Panel members believe that the selection of changes in pulmonary function expressed as percent change in FEV_1 in children is a fair indicator of an adverse effect at 15% change in all active children; and, in asthmatic children, a 10% change is indicative of adverse effects. However, the presentation of the figures showing these effects needs to be revised to indicate the uncertainties in the results used, particularly at the lower levels of exposure. The potential mechanisms whereby these changes are a reflection of both pain on breathing, partial inflammation of smaller airways, other effects on airways, and potentially triggers for more significant respiratory morbidity, particularly in asthmatic children, are not adequately discussed. In addition, some added discussion is necessary to indicate that these measures are generally taken in areas with relatively high background levels of ozone exposure, and that the role that tolerance may play in minimizing the degree of adverse effect observed needs to be considered.

From the perspective of the epidemiological data, the Ozone Panel judged the selection of: respiratory symptoms in moderate/severe asthmatic children (ages zero [birth] to 12); hospital admissions for respiratory illness among asthmatic children; and premature total non-accidental and cardiorespiratory mortality for inclusion in the quantitative risk assessment to be appropriate. However, the CASAC believes that several other endpoints should be discussed qualitatively to support the findings that these endpoints indicate that significant adverse effects are occurring at exposure concentrations well below the current standard. Other endpoints deemed worthy of additional discussion included respiratory emergency department visits among asthmatics and patients with other respiratory diseases, increased medication usage, and increased

symptomatology reported at exposure levels well below the current standard. Taken together, members of the Ozone Panel felt strongly that these findings preclude including the current standard as a scientifically defensible option for the Administrator (see discussion about Chapter 6 found in the main portion of the letter above).

Another problem in the health effects calculations (see Table 5-5 and 5-11) is that they are based on computations of the form $R_x - R_{PRB}$, where R_x is the risk at a given concentration x of O_3 and R_{PRB} is the corresponding risk at policy-relevant background (PRB) for O_3. As discussed at the Ozone Panel's August meeting, the PRB is highly-problematic to calculate and is, in some sense, "unknowable." One can avoid this problem by calculating the $\Delta = R_{0.8} - R_x$ for various concentrations x. This form would allow focus on the question, "What is the difference in the expected number of health effects that will occur at various concentrations of O_3, relative to the current standard of 0.08?" A key advantage of Δ is that it does not depend on the choice of PRB, and thus is free of the uncertainties surrounding estimation of PRB.

Chapter 6 (Staff Conclusions on Primary O_3 NAAQS): See the discussion on Chapter 6 found in the main portion of the letter above. It would also be helpful to have the estimated exposures for current (2002 and 2004) levels displayed in figures 6-1 to 6-6 (pp. 6-34 to 6-39), in addition to only those for just meeting the current standard and alternative more stringent standards. This would be analogous to the way estimated effects are displayed in Chapter 5 (Figures 5-5 to 5-9 [pp.5-58 to 5-65]).

Chapters 7 (Policy-Relevant Assessment of Welfare Effects Evidence) and 8 (Staff Conclusions on Secondary O_3 NAAQS): Chapter 7 is a well-developed and persuasively presented assessment of the welfare effects of ozone on vegetation, which forms the technical basis for the range of secondary standards recommended in Chapter 8. That having been said, the potential for significant propagation of error/uncertainty in the underlying technical documentation draws into question the conclusions drawn by EPA Staff. As observed in the Agency's 1989 and 1996 Ozone Staff Papers, ozone remains the most prevalent phytotoxic compound in the ambient air "impairing crop production and injuring native vegetation and ecosystems more than any other air pollutant" (USEPA 1989, 1996). Furthermore, as has been noted in the current assessment of human health effects, there also appears to be no safe threshold concentration below which ozone effects on sensitive vegetation are eliminated. See the additional discussion on Chapter 8 found in the main portion of the letter above.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON D.C. 20460

OFFICE OF THE ADMINISTRATOR
SCIENCE ADVISORY BOARD

March 26, 2007

EPA-CASAC-07-002

Honorable Stephen L. Johnson
Administrator
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue, NW
Washington, DC 20460

Subject: Clean Air Scientific Advisory Committee's (CASAC) Review of the Agency's Final Ozone Staff Paper

Dear Administrator Johnson:

The Clean Air Scientific Advisory Committee (CASAC or Committee), augmented by subject-matter-expert Panelists — collectively referred to as the CASAC Ozone Review Panel (Ozone Panel) — completed its review of the Agency's 2nd Draft Ozone Staff Paper in October 2006 (EPA-CASAC-07-001). In that letter, dated October 24, 2006, the CASAC indicated it would review the Agency's Final Ozone Staff Paper and offer additional, unsolicited advice to the Agency on the chapters concerned with setting the primary and secondary National Ambient Air Quality Standards (NAAQS) for ozone.

On March 5, 2007, the Ozone Panel met via a public teleconference to review EPA's *Final Review of the National Ambient Air Quality Standards for Ozone: Policy Assessment of Scientific and Technical Information* (Final Ozone Staff Paper, January 2007). The Panel focused on Chapter 6 (The Primary O_3 NAAQS) and Chapter 8 (The Secondary O_3 NAAQS). The CASAC roster is attached as found in Appendix A, the Ozone Panel roster is provided as Appendix B, and Ozone Panel members' individual review comments are found in Appendix C.

Members of the CASAC Ozone Review Panel were pleased to review EPA's Final Ozone Staff Paper. The members of CASAC and the Ozone Panel were unanimous in their praise of both the responsiveness of the Agency to our previous recommendations and of the clarity of this document. While the CASAC recognizes that the Ozone Staff Paper is a final document, the Committee offers the following advice to aid the Administrator and Agency staff in developing EPA's proposed rule for ozone and related photochemical oxidants, to be published in June 2007.

Primary Standard

- The CASAC Ozone Review Panel agreed with the choice of indicator, statistical form and averaging time for the primary Ozone NAAQS suggested by Agency staff.

- The Final Ozone Staff Paper recommended that "consideration be given to a standard level within the range of somewhat below 0.080 ppm to 0.060 ppm," adding that "[s]tandard levels within this range that were considered in staff analyses of air quality, exposure, and risk include 0.074, 0.070, and 0.064 ppm, representative of levels within the upper, middle, and lower parts of this range, respectively." Reiterating what was stated in the CASAC's previous letter to you on this review (EPA-CASAC-07-001), *Ozone Panel members were unanimous in recommending that the level of the current primary ozone standard should be lowered from 0.08 ppm to no greater than 0.070 ppm.* The above-referenced CASAC letter (from October 24, 2006), in addition to EPA's own findings in the Final Ozone Air Quality Criteria Document (AQCD) and the Final Ozone Staff Paper, provide overwhelming scientific evidence for this recommendation. *Furthermore, the Ozone Panel recommends that the NAAQS should be specified to the third decimal place of the ppm scale to avoid any rounding issues* — as indicated by the standard levels that the Agency itself considered in the Final Ozone Staff Paper.

- Pursuant to the Clean Air Act, the primary NAAQS for criteria air pollutants must be set to protect the public health with an adequate margin of safety. *Significantly, the Final Ozone Staff Paper does not address the issue of a margin of safety.* (On page 6-86, the authors conclude that the proposed standard would "…provide an appropriate degree of public health protection…;" however, there is no explicit mention of a margin of safety, *per se*.) Such a discussion should be added to the document and taken into consideration in setting the primary ozone standard.

- There is an underestimation of the affected population when one considers only twelve urban "Metropolitan Statistical Areas" (MSAs). The CASAC acknowledges that EPA may have intended to illustrate a range of impacts rather than be comprehensive in their analyses. However, it must be recognized that ozone is a regional pollutant that will affect people living outside these 12 MSAs, as well as inside and outside other urban areas.

- *There is an urgent need to fund more research on the effects on sensitive subpopulations of low levels of the photochemical oxidant mixture for which ozone is used as a surrogate.* In addition to the three field studies pointing to higher responses to the oxidant mixtures than to pure ozone that the Agency has already referenced in the Final Ozone AQCD (1–3), three other such studies are referenced below (4–6). More information on the effects of low levels of oxidant mixtures on public health is essential to inform the future decision-making process.

- Finally, with respect to policy-relevant background (PRB), the Ozone Panel wishes to point out that the Final Ozone Staff Paper does not provide a sufficient base of evidence from the peer-reviewed literature to suggest that the current approach to determining a PRB is the best method to make this estimation. One reason is that part of the PRB is not

controllable by EPA. It would require international cooperation beyond the bounds of North America. A better scientific understanding of the PRB and its relationship to intercontinental transport of air pollutants could serve as the basis for a more concerted effort to control its growth and preserve the gains in air quality achieved by control efforts within the U.S. In any case, there is no apparent need to define PRP in the context of establishing a health-based (primary) ozone NAAQS. The effects of inhaled ozone on decreases in respiratory function have been seen in healthy children exposed to ozone within ambient air mixtures in summer camps (1–6). Furthermore, the concentration-response functions above 40 ppb are either linear, or indistinguishable from linear. Thus, PRB is irrelevant to the discussion of where along the concentration-response function a NAAQS with an 8-hour averaging time that provides enhanced public health protection should be.

Secondary Standard

- *The CASAC Ozone Review Panel members were unanimous in supporting the recommendation in the Final Ozone Staff Paper that protection of managed agricultural crops and natural terrestrial ecosystems requires a secondary Ozone NAAQS that is substantially different from the primary ozone standard in averaging time, level and form.*

- The recommended metric for the secondary ozone standard is the (sigmoidally-weighted) W126 index, accumulated over at least the 12 "daylight" hours and over at least the three maximum ozone months of the summer "growing season."

- The Ozone Panel agrees with EPA Staff recommendations that the lowest bound of the range within which a seasonal W126 welfare-based (secondary) ozone standard should be considered is 7.5 ppm-hrs; however, it *does not* agree with Staff's recommendations that the upper bound of the range should be as high as 21 ppm-hours. Rather, the Panel recommends that the upper bound of the range considered should be no higher than 15 ppm-hour, which the Panel estimates is approximately equivalent to a seasonal 12-hour SUM06 level of 20 ppm-hours.

- Multi-year averaging to promote a "stable" secondary Ozone NAAQS is less appropriate for a cumulative, seasonal secondary standard than for a primary standard based on maximum eight-hour concentrations. If multi-year averaging is employed to increase the stability of the secondary standard, the level of the standard should be revised downward to assure that the desired threshold is not exceeded in individual years.

- There was an effective, Federally-funded program of ozone environmental effects research during the 1970s and 1980s, but such research support has been neglected in recent years. It is reasonable to conclude that changes in the distribution and genetic makeup of crop cultivars and naturally occurring plant species has and will take place over time along with modification of levels and distribution of ambient ozone exposures. Therefore, future refinements of the secondary Ozone NAAQS will require both: (1) a significant future investment in effects research to ensure that data for plant response to ozone are representative of the species and genetic composition of current crop and forest

species utilized by society; and (2) a clear understanding of the sources and propagation of uncertainty in the results of that research.

Additional details on the general recommendations listed above are provided in the comments of the individual members of the Ozone Panel that are included in Appendix C.

The CASAC appreciate this opportunity to work with the Agency is using science to help inform the setting of primary and secondary NAAQS to protect public health. While this is the last of a long series of Agency NAAQS-related staff papers, the Committee will continue to provide you with scientific advice related to setting criteria air pollutant standards protective of the public health and public welfare under EPA's revised NAAQS review process. As always, the CASAC wishes the Agency well in this important endeavor.

Sincerely,

/Signed/

Dr. Rogene Henderson, Chair
Clean Air Scientific Advisory Committee

Appendix A – Roster of the Clean Air Scientific Advisory Committee

Appendix B – Roster of the CASAC Ozone Review Panel

Appendix C – Review Comments from Individual CASAC Ozone Review Panel Members

References

1. Spector, D.M., Lippmann, M., Lioy, P.J., Thurston, G.D., Citak, K., James, D.J., Bock, N., Speizer, F.E., and Hayes, C. Effects of ambient ozone on respiratory function in active normal children. *Am. Rev. Respir. Dis.* 137:313-320 (1988).
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Appendix A – Clean Air Scientific Advisory Committee Roster (FY 2006)

U.S. Environmental Protection Agency
Science Advisory Board (SAB) Staff Office
Clean Air Scientific Advisory Committee (CASAC)

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Appendix B – CASAC Ozone Review Panel Roster

U.S. Environmental Protection Agency
Science Advisory Board (SAB) Staff Office
Clean Air Scientific Advisory Committee (CASAC)
CASAC Ozone Review Panel

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