Review of the Primary National Ambient Air Quality Standards for Nitrogen Dioxide:

Risk and Exposure Assessment Planning Document

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Risk and Exposure Assessment Planning Document

U. S. Environmental Protection Agency

Office of Air Quality Planning and Standards

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LIST OF ACRONYMS/ABBREVIATIONS

AADT   Annual average daily traffic

AERMOD   EPA's Air Dispersion Model

AHR   Airway hyperresponsiveness

ANPR   Advanced notice of proposed rulemaking

APEX   EPA's Air Pollutants Exposure model

AQI   Air Quality Index

AQS   EPA's Air Quality System

CAA   Clean Air Act

CAPS   Cavity attenuated phase shift

CASAC   Clean Air Scientific Advisory Committee

CBSA   Core-based statistical area

CFR   Code of Federal Regulations

CO   Carbon monoxide

C-R   Concentration-response

EPA   Environmental Protection Agency

E-R   Exposure-response

FEM   Federal Equivalent Method

FEV_1 Forced expiratory volume in one second, volume of air exhaled in first second of exhalation

FR   Federal Register

FRM   Federal Reference Method

HA   Hospital admission

HERO   Health and Environmental Research Online

HONO   Nitrous acid

HNO_3 Nitric acid

IRP   Integrated Review Plan

ISA   Integrated Science Assessment

$\mu g/m^3$ micrograms per cubic meter

m   Meters

ME   Microenvironmental

MSA   Metropolitan statistical area

NAAQS   National ambient air quality standards

NCEA   National Center for Environmental Assessment

NCore   National Core Monitoring Network

NO   Nitric oxide

NO_2	Nitrogen dioxide
NO_3^-	Nitrate
NO_X	$NO+NO_2$
NO_Y	Total oxides of nitrogen ($NO_X + NO_Z$)
NO_Z	Reactive oxides of nitrogen (e.g., HNO_3, HONO, PAN, particulate nitrates)
O_3	Ozone
OAQPS	Office of Air Quality Planning and Standards
OAR	Office of Air and Radiation
OMB	Office of Management and Budget
OR	Odds ratio
ORD	Office of Research and Development
PA	Policy Assessment
PAN	Peroxyacetyl nitrate
PCA	Principal Components Analysis
PM	Particulate matter
$PM_{2.5}$	In general terms, particulate matter with an aerodynamic diameter less than or equal to a nominal 2.5 microns (μm); a measurement of fine particles
ppb	Parts per billion
ppm	Parts per million
QA	Quality assurance
QMP	Quality Management Plan
REA	Risk and Exposure Assessment
RIA	Regulatory Impact Analysis
RR	Relative risk
RTP	Research Triangle Park
SES	Socioeconomic status
SLAMS	State and local monitoring stations
SO_2	Sulfur dioxide
TBD	To be determined

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1 INTRODUCTION

     The U.S. Environmental Protection Agency (EPA) is conducting a review of the air quality criteria and the primary (health-based) national ambient air quality standards (NAAQS) for nitrogen dioxide (NO_2).[1] The establishment and periodic review of NAAQS are governed primarily by sections 108 and 109 of the Clean Air Act (Act). The NAAQS are established for pollutants that 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. The Act requires that NAAQS are to be based on air quality criteria, which are to accurately reflect the latest scientific knowledge useful in indicating the kind and extent of identifiable effects on public health or welfare that may be expected from the presence of the pollutant in ambient air. Based on periodic reviews of the air quality criteria and standards, the Administrator is to make revisions in the criteria and standards, and promulgate any new standards, as may be appropriate. The Act also requires that an independent scientific review committee advise the Administrator as part of this NAAQS review process, a function now performed by the Clean Air Scientific Advisory Committee (CASAC).

     The overall plan for this review is presented in the *Integrated Review Plan for the Primary National Ambient Air Quality Standards for Nitrogen Dioxide* (IRP) (U.S. EPA, 2014a). The IRP summarizes the Clean Air Act (CAA) requirements related to the establishment and review of the NAAQS; the history of the primary NO_2 NAAQS, including the key science and policy issues considered in the last review; the anticipated process and schedule for the current review of the primary NO_2 NAAQS; and the anticipated scope and organization of key assessment documents in the current review, including the Integrated Science Assessment (ISA), the Risk and Exposure Assessment (REA), if warranted, and the Policy Assessment (PA). The IRP also lays out the key policy-relevant issues to be addressed in this review as a series of questions that will frame our[2] approach to reaching conclusions on the degree to which the available evidence and information could support retaining or revising the current primary NO_2 NAAQS.

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[1] The EPA is separately reviewing the welfare effects associated with oxides of nitrogen and the protection provided by the secondary NO_2 standard, in conjunction with a review of the secondary standard for sulfur dioxide (SO_2) (U.S. EPA, 2014a, section 1.4).

[2] In this document, the terms "we" and "our" refer to staff in the EPA's Office of Air Quality Planning and Standards (OAQPS).

1   As a further step in planning for the current review, this document is intended to facilitate
2   CASAC advice and public input to the EPA on potential support for updated quantitative
3   analyses of NO_2 exposures and/or health risks. To facilitate such advice and input, the EPA staff
4   has considered the degree to which important uncertainties identified in quantitative analyses
5   from previous reviews have been addressed by newly available scientific evidence, tools, or
6   information. Based on these considerations, this document presents our preliminary conclusions
7   on the extent to which updated quantitative analyses of exposures and/or health risks are
8   warranted in the current review. For updated analyses that are supported, this planning document
9   also presents our anticipated approaches to conducting such analyses and, where appropriate,
10   preliminary results based on illustrative examples.
11   Staff's considerations and preliminary conclusions in this planning document draw from
12   the scientific evidence assessed in the second draft of the *Integrated Science Assessment for*
13   *Oxides of Nitrogen – Health Criteria* (ISA) (U.S. EPA, 2015), the discussions of key issues in
14   the IRP (U.S. EPA, 2014a), the *NO₂ Risk and Exposure Assessment Report* from the last review
15   of the primary NO_2 NAAQS (U.S. EPA, 2008a), advances in modeling tools and techniques, and
16   new air quality data that have become available since the last review. This document is being
17   submitted for review by the Clean Air Scientific Advisory Committee (CASAC) and made
18   available for public comment. The EPA staff will consider advice from CASAC and input from
19   the public in reaching conclusions regarding updated quantitative analyses in the current review.
20   These staff conclusions will be reflected in future documents[3] generated as part of this review of
21   the primary NO_2 NAAQS, as described in Chapter 5 below.
22   The remainder of this chapter provides overviews of the history of the primary NO_2
23   NAAQS (section 1.1); potential approaches to characterizing risks with quantitative analyses
24   (section 1.2); staff's key considerations in evaluating the degree to which updated quantitative
25   analyses are supported in the current review (section 1.3); and the organization of the remainder
26   of this planning document (section 1.4).

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[3] Future documents (i.e. Risk and Exposure Assessment, Policy Assessment) will also be reviewed by CASAC and made available for public comment.

1.1 HISTORY OF THE PRIMARY NO₂ NAAQS

On April 30, 1971, EPA promulgated NAAQS for NO_2 under section 109 of the CAA. The primary standard was set at 0.053 parts per million (ppm) (53 ppb), annual average (36 FR 8186).[4] The EPA completed reviews of the air quality criteria and NO_2 NAAQS in 1985 and 1996, with decisions to retain the annual standard without revision (50 FR 25532, June 19, 1985; 61 FR 52852, October 8, 1996).

In the last review of the primary NO_2 NAAQS, completed in 2010 (75 FR 6474, February 9, 2010), the EPA determined that the annual standard alone was not requisite[5] to protect the public from respiratory effects that could result from short-term exposures to ambient NO_2. To provide increased public health protection, including for at-risk populations such as people with asthma, the EPA added a new short-term NO_2 standard with a level of 100 ppb, based on the 3-year average of the 98th percentile of the annual distribution of daily maximum 1-hour NO_2 concentrations. The EPA also retained the existing annual NO_2 standard, with a level of 53 ppb, to continue to provide protection for effects potentially associated with long-term exposures.[6]

The Administrator's final decisions on the standard placed primary emphasis on the scientific evidence for respiratory effects attributable to short-term NO_2 exposures. She viewed the results of quantitative exposure and risk analyses as providing information in support of her decision (75 FR 6498, February 9, 2010).[7] The approaches employed in the last review to

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[4] The secondary standard for NO_2 was set identical to the primary standard.

[5] In setting primary standards that are requisite to protect public health, as provided in section 109(b) of the Clean Air Act, the EPA's task is to establish standards that are neither more nor less stringent than necessary for these purposes.

[6] The existing primary NO_2 NAAQS are specified at 40 CFR 50.11.

[7] The decisions made in the last review of the primary NO_2 standard were informed by the extensive body of scientific evidence published through early 2008 and assessed in the *Integrated Science Assessment for Oxides of Nitrogen – Health Criteria* (2008 ISA, U.S. EPA, 2008b); the quantitative exposure and risk analyses in the *Risk and Exposure Assessment to Support the Review of the NO₂ Primary National Ambient Air Quality Standard* (2008 REA, U.S. EPA, 2008a); the policy-relevant evidence- and exposure-/risk-based considerations related to the primary NO_2 NAAQS; the advice and recommendations of EPA's Clean Air Scientific Advisory Committee (CASAC, Henderson, 2008; Samet, 2008a,b, 2009); and public comments (75 FR 6474).

1 estimate NO$_2$ exposures and health risks are summarized briefly in section 1.2 below, and are

2 discussed in more detail in subsequent chapters of this planning document.

3   In conjunction with the revised primary NO$_2$ NAAQS, the EPA established a two-tiered

4 monitoring network comprised of: (1) near-road monitors to be placed in locations of expected

5 maximum 1-hour NO$_2$ concentrations near heavily trafficked roads in urban areas and (2)

6 monitors located to characterize areas with the highest expected NO$_2$ concentrations at the

7 neighborhood and larger spatial scales (referred to as *area-wide* monitors) (75 FR 6505 to 6506,

8 February 9, 2010). Some near-road NO$_2$ monitors are currently in operation, with the remainder

9 of the anticipated monitors scheduled to become operational by January 1, 2017.[8]

10 ## 1.2 APPROACHES TO CHARACTERIZING RISKS

11   In each NAAQS review, selection of the appropriate model for the characterization of

12 risks is influenced by the nature and strength of the evidence for the subject pollutant. Depending

13 on the type of evidence available, analyses may include quantitative risk assessments based on

14 dose-response, exposure-response, or ambient concentration-response relationships. Analyses

15 may also be based on comparisons of health effect benchmark concentrations, drawn from

16 controlled human exposure studies, with modeled exposure estimates or ambient air quality

17 concentrations (i.e., as surrogates for potential ambient exposures). The variety of approaches

18 that have been employed in NAAQS reviews is summarized in Figure 1-1.

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[8] Subsequent to the 2010 rulemaking, the EPA revised the deadlines by which the near-road monitors are to be operational in order to implement a phased deployment approach (78 FR 16184, March 14, 2013).
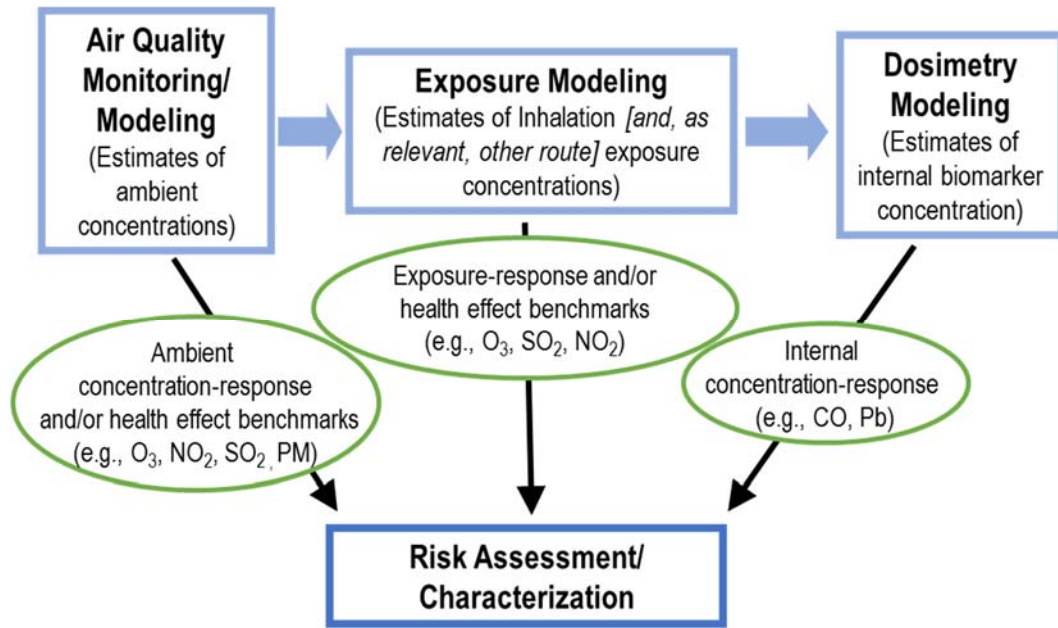
1

2 **Figure 1-1. Risk characterization models employed in NAAQS Reviews.**

3   The conceptual model for the NO_2 health risk characterization conducted in the last

4 review is summarized below in Figure 1-2. This model was based on the available scientific

5 evidence assessed in the 2008 ISA, recognizing that the strongest evidence was for respiratory

6 effects attributable to short-term NO_2 exposures (U.S. EPA, 2008b, section 5.3).[9]

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[9] As indicated in Figure 1-2, the 2008 REA focused on exposures to ambient NO_2, though indoor sources of NO_2 and indoor exposures were also evaluated (U.S. EPA, 2008a, Chapter 8).

Key Sources of NO$_2$

- Mobile source emissions to ambient air
- Stationary source emissions to ambient air

Exposure Pathways

- Air - Outdoors
- Air – Indoors

Routes of Exposure

- Inhalation

Key At-Risk Populations Identified

- People with respiratory disease, focus on asthma (children, adults)

Endpoints

- Airway responsiveness, based on information from controlled human exposure studies
- Emergency department visits for respiratory causes, based on information from epidemiologic study

Risk Metrics

- Number of days per year with ambient NO$_2$ concentrations at or above health effect benchmarks
- Number/percent of people with asthma estimated to experience NO$_2$ exposure concentrations at or above health effect benchmarks
- Estimated NO$_2$-associated emergency department visits for respiratory causes

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2 **Figure 1-2. Conceptual model for risk characterization in the last review of the primary**
3 **NO$_2$ NAAQS**

4   Based on the conceptual model summarized in Figure 1-2, the risk characterization in the

5 last review employed three approaches to quantify NO$_2$ exposures and health risks (U.S. EPA,

6 2008a):

7   1) Health effect benchmarks were identified based on information from controlled human
8   exposure studies of NO$_2$-induced increases in airway responsiveness. Ambient NO$_2$
9   concentrations were compared to these health effect benchmarks. In urban areas across
10   the U.S., such comparisons were made for ambient NO$_2$ concentrations at locations of
11   NO$_2$ monitoring sites and simulated concentrations on/near roadways (U.S. EPA, 2008a,
12   Chapter 7).

13   2) Modeled estimates of personal NO$_2$ exposures were compared to health effect
14   benchmarks in a single urban area (Atlanta, GA). Exposures were characterized for
15   children with asthma and for people of all ages with asthma (U.S. EPA, 2008a, Chapter
16   8).

1   3) Concentration-response relationships from an epidemiologic study were used to estimate
2   NO_2-associated emergency department visits for respiratory causes in Atlanta, GA (U.S.
3   EPA, 2008a, Chapter 9).

4   Exposures and risks were estimated for multiple NO_2 air quality scenarios, including for
5   ambient concentrations adjusted to just meet the existing annual NO_2 standard (i.e., the existing
6   NO_2 standard at the time of the last review) and for concentrations adjusted to just meet potential
7   alternative 1-hour standards with levels from 50 to 200 ppb. The quantitative analyses conducted
8   in the last review are discussed in more detail in the subsequent chapters of this planning
9   document.

10   **1.3   KEY CONSIDERATIONS IN THE CURRENT REVIEW**
11   In the current review, preliminary conclusions regarding the extent to which the newly
12   available evidence and information address important uncertainties and support updated
13   quantitative analyses are based on our consideration of a variety of factors. These include the
14   available health evidence; the available technical information, tools, and methods; and judgments
15   as to the likelihood that particular quantitative analyses will add substantially to our
16   understanding of NO_2 exposures or health risks beyond the insights gained from the analyses
17   conducted in the last review. These key considerations are summarized in Figure 1-3, below.
18
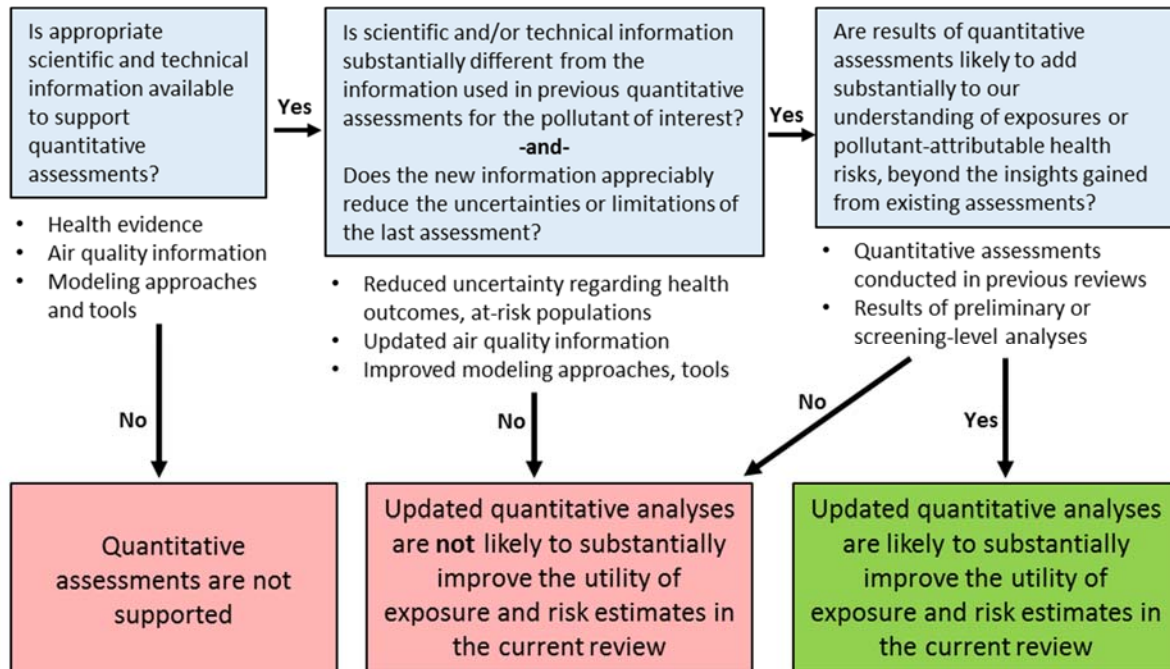
20   **Figure 1-3. Key considerations for updated quantitative analyses.**
21   An initial consideration is the available health effects evidence, and the foundation it may
22   provide for updated quantitative analyses. Our evaluation of the scientific evidence in this

1   planning document is based on the assessment of that evidence in the 2nd draft ISA (U.S. EPA,

2   2015).[10] In particular, we focus on information newly available in this review that addresses

3   uncertainties identified in the last review and/or that may change major conclusions of the last

4   review, such as causality determinations for NO_2-associated health effects and conclusions

5   regarding at-risk populations and lifestages (U.S. EPA, 2015).[11]

6   Consistent with prior reviews, in considering the evidence with regard to support for

7   quantitative analyses, we give primary consideration to health endpoints for which the ISA

8   concludes the evidence supports a "causal" relationship or indicates that there is "likely to be a

9   causal" relationship. In the current review, the 2nd draft ISA (U.S. EPA, 2015) reaches the

10   following conclusions in this regard:

11   - The evidence supports "a causal relationship between short-term NO_2 exposure and
12   respiratory effects" and the "strongest evidence is for effects on asthma exacerbation"
13   (U.S. EPA, 2015, Table 1-1, pp. 1-19).[12] Key supporting evidence for these
14   conclusions comes from controlled human exposure studies of airway responsiveness
15   and from epidemiologic studies of asthma-related hospital admissions, emergency
16   department visits, and respiratory symptoms (U.S. EPA, 2015, section 1.5.1).

17   - The evidence "indicates there is likely to be a causal relationship between long-term
18   NO_2 exposure and respiratory effects" (U.S. EPA, 2015, section 1.5.1, pp. 1-21 and 1-
19   21) and the "strongest evidence is for effects on asthma development" (U.S. EPA,
20   2015, Table 1-1).[13] Key supporting evidence comes from epidemiologic cohort studies

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[10] Staff will further consider the preliminary conclusions presented in this planning document in light of the assessment of the evidence in the in the final ISA.

[11] Conclusions in the 2nd draft ISA are based on a thorough evaluation of the available scientific evidence, taking into account factors such as the consistency and coherence of the evidence within and across disciplines (e.g., epidemiology, controlled human exposure, and toxicology), biological plausibility, and strength and specificity of effects (U.S. EPA, 2015, Preamble, section 5). With regard to health effects, the 2nd draft ISA uses a five-level hierarchy to classify the overall weight-of-evidence into one of the following categories: causal relationship, likely to be a causal relationship, suggestive but not sufficient to infer a causal relationship, inadequate to infer a causal relationship, and not likely to be a causal relationship (U.S. EPA, 2015, section 1.2). With regard to potential at-risk populations, the 2nd draft ISA classifies the evidence into one of the following categories: "adequate evidence," "suggestive evidence," "inadequate evidence," and "evidence of no effect" (U.S. EPA, 2015, section 7.2).

[12] The ISA in the last review of the Primary NO_2 NAAQS concluded the available evidence indicated that there was "likely to be a causal relationship" between short-term NO_2 exposure and respiratory effects.

[13] The ISA in the last review of the Primary NO_2 NAAQS concluded the available evidence was "suggestive, but not sufficient" to infer a causal relationship between long-term NO_2 exposure and respiratory effects.

reporting associations between long-term ambient NO_2 concentrations (i.e., averaged over 1−10 years) and asthma incidence in children. Support for the biological plausibility of effects attributable to long-term exposures is provided by "a small body of experimental studies" (U.S. EPA, 2015, Table 1-1).

- For all other health endpoints evaluated, the evidence is either "suggestive, but not sufficient, to infer a causal relationship" or "inadequate to infer a causal relationship" (U.S. EPA, 2015, section 1.5.2).[14]

Evaluation of the support for quantitative analyses additionally gives primary consideration to populations and lifestages for which the ISA judges there is "adequate" evidence for increased risk.[15] In the current review, the 2nd draft ISA concludes that "there is adequate evidence that people with asthma, children, and older adults are at increased risk for NO_2-related health effects" (U.S. EPA, 2015, Table 7-26). The second draft ISA concludes that there is greater uncertainty for other at-risk populations because the evidence is inconsistent and/or because the evidence is for effects that "are not clearly related to NO_2 exposure" (U.S. EPA, 2015, section 1.6.5, pp. 1-45 to 1-46).

Given these conclusions with regard to health endpoints and at-risk populations, our consideration of potential updated quantitative analyses in this document is focused on health outcomes related to asthma exacerbation in children and adults (short-term NO_2 exposures) and the development of asthma in children (long-term NO_2 exposures). We focus particularly on the key health studies that informed the 2nd draft ISA's causality determinations.

1.4 ORGANIZATION OF THIS DOCUMENT

The remainder of this planning document presents our evaluations and preliminary conclusions regarding the degree to which the newly available evidence and information addresses important uncertainties and the support for updated quantitative analyses in the current review. Chapters 2 through 4 present our considerations and preliminary conclusions regarding the following:

- Analyses comparing ambient NO_2 concentrations with health effect benchmarks derived from controlled human exposure studies (Chapter 2).

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[14] Health outcomes for which the evidence is judged "suggestive, but not sufficient, to infer a causal relationship" can be considered as part of the overall consideration of the health evidence in the Policy Assessment.

[15] This is consistent with the approach adopted in the ongoing review of the ozone (O_3) NAAQS (U.S. EPA, 2014d). The ISA framework for drawing conclusions about the role of various factors in modifying risks of air pollution exposures has been developed since the last review of the primary NO_2 NAAQS (U.S. EPA, 2015, section 1.6.5).

1   • Assessment of human exposures based on modeled estimates in people with asthma, and
2   comparing modeled 1-hour average exposures to1-hour health effect benchmarks
3   (Chapter 3).

4   • Health risk assessment (Chapter 4).

5   Chapter 5 summarizes the conclusions from chapters 2 through 4, and discusses the next steps in
6   the current review of the primary NO_2 NAAQS.

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2 AIR QUALITY AND HEALTH BENCHMARK COMPARISONS

A key part of the body of scientific evidence identified in the 2nd draft ISA as supporting "a causal relationship between short-term NO_2 exposure and respiratory effects" (U.S. EPA, 2015, Table 1-1) comes from controlled human exposure studies of airway responsiveness in people with asthma (U.S. EPA, 2015, section 1.5.1). In the last review, the REA used information from such studies to identify NO_2 health effect benchmarks. The REA compared these benchmarks with the ambient NO_2 concentrations estimated to occur under various air quality scenarios of interest (just meeting the existing and potential alternative standards). In these analyses, ambient NO_2 concentrations served as surrogates for potential exposure concentrations.[16]

This chapter presents the considerations leading to staff's preliminary conclusion that new information available in the current review is expected to add substantially to our understanding of the potential for population exposures to ambient NO_2 concentrations at or above health effect benchmarks. Updated analyses that incorporate this new information would be expected to provide additional perspective, beyond the analyses from the last review, on the extent to which NO_2 exposures allowed by the current standard (and potential alternative standards, as appropriate) could have important implications for public health.

Section 2.1 below provides an overview of the NO_2 air quality benchmark comparison from the last review. Section 2.2 provides an overview of the information that is available in the current review to inform updated analyses and presents staff's preliminary conclusion that updated analyses should be considered. Section 2.3 provides an overview of staff's proposed

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[16] If ambient concentrations are properly characterized (i.e., they appropriately capture temporal and spatial variability in concentrations across the selected study area), they would serve as a conservative estimate of ambient-related exposures. This is because ambient NO_2 concentrations are attenuated within indoor microenvironments where people commonly spend substantial time throughout their day.

1   analytical approach for updated analyses and presents preliminary results for a single illustrative

2   urban study area.

3   ## 2.1 OVERVIEW OF THE ASSESSMENT FROM THE LAST REVIEW

4   In the last review, the 2008 REA included analyses comparing ambient NO_2

5   concentrations at monitoring sites and on/near roadways to health effect benchmarks ranging

6   from 100 to 300 ppb (U.S. EPA, 2008a, Chapter 7). Health effect benchmarks reflected the range

7   of NO_2 concentrations that had been reported to increase airway responsiveness in the majority

8   of people with asthma, based on a meta-analysis of individual study data presented in the 2008

9   ISA (U.S. EPA, 2008b, Table 3.1-3).[17] These comparisons of ambient NO_2 concentrations to

10   health effect benchmarks provided perspective on the extent to which, under various air quality

11   scenarios, populations could potentially experience 1-hour exposures to NO_2 concentrations that

12   could be of concern, particularly for people with asthma (U.S. EPA, 2008a, Chapter 7).

13   The 2008 REA's air quality assessment was based on NO_2 concentrations measured at

14   available U.S. monitoring sites,[18] with a particular focus on 18 Core Based Statistical Areas

15   (CBSAs).[19] The 2008 REA examined the potential for ambient NO_2 concentrations to be greater

16   than or equal to health effect benchmarks when air quality in the CBSAs was adjusted to just

17   meet the then-existing standard (i.e., annual standard with a level of 53 ppb) or potential

18   alternative 1-hour standards with levels ranging from 50 to 200 ppb (and 98th or 99th percentile

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[17] Health effect benchmarks are discussed in more detail in section 2.2.2 below.

[18] Air quality data was separated into two six-year time periods, 1995 to 2000 (representing historical air quality) and 2001 to 2006 (representing recent air quality) (U.S. EPA, 2008a, section 7.2.2). After applying a 75% data completeness criterion the final analytical data base included 627 monitors collecting ambient concentrations for 4,177 site-years of data (a valid monitoring day had ≥18 hourly measurements; monitors included in the analysis had ≥75% valid monitoring days in a year). Note, current validity criteria use calendar quarters (75% valid monitoring days in a quarter, having all four quarters complete) to ascertain a complete year.

[19] At the time the assessment was conducted, we used the terms CMSA/MSA to describe the monitors associated with metropolitan statistical areas. We replaced that terminology here with CBSA to reflect current terminology. First, using the complete set of ambient monitor data, we identified whether or not monitors belonged to a CBSA.. Then, CBSA-named study areas were identified as those having annual mean NO_2 concentrations occurring at a minimum of one monitor in the CBSA at or above 25.7 ppb (i.e., the 90th percentile concentrations across all study areas and site-years) and/or had at least one reported 1-hour NO_2 concentration greater than or equal to 200 ppb. All remaining sites not included in this collection of CBSA-named study areas were aggregated into either one of two groups: all other CBSA or all other non-CBSA.

1   forms).[20] For the air quality scenarios evaluated, the 2008 REA highlighted the number of days in

2   each CBSA and each year with ambient NO_2 concentrations at or above one or more of the

3   health effect benchmarks.

4   At the time of the last review, we also focused portions of the air quality analyses on

5   characterizing NO_2 concentrations occurring on roads and in near-road environments. Mobile

6   sources are the largest contributors to total annual NO_X emissions in the U.S. (U.S. EPA, 2008b,

7   section 2.2.1) and monitor-based research studies had demonstrated large gradients in ambient

8   NO_2 concentrations around major roadways, with higher concentrations occurring closer to roads

9   and lower concentrations occurring farther away from those roads (U.S. EPA, 2008b, sections

10   2.5.3.2, 2.5.4). Because the ambient monitoring network present at the time of the last review

11   was not designed to systematically measure NO_2 concentrations near the most heavily trafficked

12   roadways, the 2008 REA simulated ambient NO_2 concentrations on-/near-roads using

13   information from monitoring studies published in the scientific literature.[21] Specifically, to

14   estimate on-/near-road concentrations the 2008 REA categorized ambient NO_2 monitors based

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[20] Because annual average ambient NO_2 concentrations were below the level of the annual standard (i.e., 53 ppb) and most of the potential alternative 1-hour standards evaluated, ambient concentrations were primarily adjusted upwards to simulate just meeting the then-existing and potential alternative standards (U.S. EPA, 2008a, section 6.3.1). For the adjusted air quality standard scenarios, a proportional adjustment approach was used. This approach was supported by within-monitor comparisons of low and high NO_2 concentration years that largely demonstrated characteristics of a proportional relationship. Specifically, linear regressions were performed using the distributions of daily maximum 1-hour concentrations for a low-concentration year and a high-concentration year, measured at the same ambient monitor. Statistically significant linear regression slopes and model R^2 values strongly supported features of linearity. However, in a few instances this analysis identified the presence of statistically significant regression intercepts and deviations from linearity at upper percentile concentrations, introducing uncertainty into the conclusion that a proportional relationship existed at all monitors (Rizzo, 2008; U.S. EPA, 2008a, section 7.4.5).

[21] At the time of the last review, based on the available evidence, there was uncertainty regarding the locations of maximum NO_2 concentrations with respect to roadway emissions and transformation of NO to NO_2. Therefore, we characterized these simulated concentrations as on-/near-road.

1 on their distance from a road[22] and applied literature-derived factors to concentrations at
2 monitoring sites ≥ 100 meters (m) from a road (U.S. EPA, 2008a, section 7.2).[23]
3   For each CBSA and monitor year, the air quality at monitor locations was first adjusted to
4 just meet the existing annual standard or potential alternative 1-hour standards. In cases where
5 monitors were sited ≥ 100 meters (m) from a road, simulated on-road concentrations were
6 obtained by applying on-road simulation factors after the air quality adjustment. The 2008 REA
7 presented the number of days per year with simulated 1-hour NO_2 concentrations on-, near-, and
8 away-from-roads at or above the health effect benchmarks (U.S. EPA, 2008a, section 7.2.4).

9 ## 2.1.1 Summary of Results

10   The 2008 REA presented a number of results from the analyses of NO_2 air quality and
11 health effect benchmarks (U.S. EPA, 2008a, sections 7.3.3 and 7.3.4; Appendix A), including the
12 following:

13   1. On average, simulated NO_2 concentrations on/near roads were about 80% higher than
14   measured ambient concentrations in the same CBSA at monitoring sites ≥ 100 m from a
15   road (U.S. EPA, 2008a, section 7.3.4).

16   2. When air quality was adjusted to just meet the existing annual NO_2 standard, most
17   CBSAs were estimated to have between 100 and 300 days per year with simulated on-
18   /near-road 1-hour NO_2 concentrations ≥ 100 ppb; between 25 and 100 days per year with
19   simulated NO_2 concentrations ≥ 200 ppb (U.S. EPA, 2008a, Figure 7-6); and between 1
20   and 20 days per year with simulated NO_2 concentrations ≥ 300 ppb (U.S. EPA, 2008a,
21   Appendix A, Table A-122). There were fewer days per year with such NO_2
22   concentrations at the locations of the ambient monitors (U.S. EPA, 2008a, Figure 7-3).

23   3. Compared to just meeting the existing annual standard:

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[22] In this assessment, road distances to each monitor were generally determined using a Tele-Atlas roads database in a GIS application. The road types used to identify near-road monitors were those defined as: 1=primary limited access or interstate, 2=primary US and State highways, 3=Secondary State and County, 4=freeway ramp, 5=other ramps. Note only the monitors falling within the 18 identified study areas had estimated distances to these identified roads types, all other monitors (either characterized as 'other CMSA/MSA' or 'all other non-CMSA/MSA') were not used to simulate on-road concentrations.

[23] The 2008 REA derived a distribution of factors using data from eleven published studies that reported NO_2 concentrations on-roads (5 studies) and/or near-roads (6 studies) and NO_2 concentrations within and/or beyond 100 meters (m) from a road and assuming an exponential model for fitting the data. The 2008 REA then probabilistically applied these factors to ambient NO_2 concentrations reported at ambient monitor sites ≥ 100 m from a road (assumed in the 2008 REA to represent background NO_2 concentrations, not influenced by roads). Major road types were defined in the 2008 REA as primary limited access or interstate, primary US and State highways, Secondary State and County, freeway ramp, and other ramps (2008 REA Appendix A, Table A-7). See Table 7-10 of the 2008 REA for the specific values of distributions that were used and Appendix A, section 8 for the studies used and the derivation methodology (U.S. EPA, 2008a).

a. When air quality was adjusted to just meet alternative 1-hour standards with levels of either 50 or 100 ppb, fewer days per year had simulated 1-hour NO_2 concentrations at or above health effect benchmarks (U.S. EPA, 2008a, Table 7-29).

b. When air quality was adjusted to just meet an alternative 1-hour standard with a level of 150 ppb, similar numbers of days per year had simulated 1-hour NO_2 concentrations at or above health effect benchmarks (U.S. EPA, 2008a, compare estimates in Figure 7-6 to those in Figures 7-7 and 7-8).

c. When air quality was adjusted to just meet an alternative 1-hour standard with a level of 200 ppb, generally larger numbers of days per year had simulated 1-hour NO_2 concentrations at or above health effect benchmarks (U.S. EPA, 2008a, compare estimates in Figure 7-6 to those in Figures 7-7 and 7-8).

2.1.2 Uncertainties and Limitations

The 2008 REA identified several sources of uncertainty associated with these analyses of ambient air quality (U.S. EPA, 2008a, section 7.4, Table 7-31). Key sources of uncertainty are summarized below.

1. Spatial representativeness of assessment: The 2008 REA noted that, relative to the area encompassed by the CBSAs that comprised the urban study locations, there are a relatively small number of ambient monitors in each location. To the extent there are locations where ambient NO_2 concentrations exceed those measured by ambient monitors, the occurrence of NO_2 concentrations at or above health effect benchmarks could be underestimated. To address this uncertainty in part, the 2008 REA developed the approach to estimate on-road NO_2 concentrations, though it is possible that other local sources exist, perhaps differing in emissions from mobile sources, and are not accounted for by the existing monitoring network (U.S. EPA, 2008a, section 7.4.4).

2. Simulated on-/near-road concentrations: The statistical model developed in the 2008 REA to simulate on-/near-road NO_2 concentrations was based on measurement data reported in a limited number of peer-reviewed studies. Most of these studies used averaging times much longer than the 1-hour concentrations relevant for the health benchmarks (i.e., 7-14 days or longer). The relationships between the study-related longer-term averaging times and our use of short-term averaging times (1-hour) was not known at that time. In addition, the derived factors were applied to concentrations at the away-from-road sites ($\geq$100 m from roads) without considering the potential relationship between the derived factors and ambient concentrations. The 2008 REA noted that if there is a concentration dependence in the relationship between NO_2 on/near roads and NO_2 away from roads, the approach used would bias the simulated concentrations, though the direction of such potential bias was not known. Other uncertainties related to the appropriateness of applying the literature-derived factors to specific U.S. urban study areas include; not accounting for in-vehicle penetration and decay of NO_2 that would likely be associated with actual on-road exposures; the potential for emissions from non-road sources to influence the monitors $\geq$ 100 m from the road affecting their representativeness of

1   background concentrations; and the selection of an exponential decay model (U.S. EPA,
2   2008a, section 7.4.6) to define the concentration decline with distance from the roadway.

3   3. Adjusting ambient concentrations to just meet air quality standards: The 2008 REA noted
4   that there is uncertainty in the approach used to adjust air quality to just meet the existing
5   annual standard and potential alternative 1-hour NO_2 standards. This reflects the
6   uncertainty in the true relationship between the adjusted concentrations meant to simulate
7   a hypothetical future scenario and the historical unadjusted air quality. The adjustment
8   factors used to simulate just meet the existing annual and alternative 1-hour standards
9   assumed that all hourly concentrations would change proportionately at each ambient
10   monitoring site. The 2008 REA's discussion of uncertainty with the air quality
11   adjustment focused on two areas: (1) uncertainty in the appropriateness of using a
12   proportional adjustment approach and (2) uncertainty in applying the same approach to
13   all ambient monitors within each urban study location (U.S. EPA, 2008a, section 7.4.5).

14   4. Health effect benchmarks: The health effect benchmarks used were based on a meta-
15   analysis of individual data from controlled human exposure studies presented in the 2008
16   ISA (U.S. EPA, 2008b). The 2008 ISA meta-analysis evaluated the direction of the
17   change in airway responsiveness, though it did not evaluate the magnitude of this change.
18   Therefore, there was uncertainty in the magnitude and severity of effects that occur
19   following exposures to NO_2 concentrations at or above health effect benchmarks (U.S.
20   EPA, 2008a, section 4.2.5). In addition, the 2008 REA highlighted uncertainties related to
21   the use of benchmarks based on studies using a variety of exposure periods (generally 30
22   minutes to 2 hours) and subjects with asthma whose disease status was characterized as
23   mild, as opposed to those more severely affected (U.S. EPA, 2008a, section 7.4.7).

2.2   OVERVIEW OF INFORMATION AVAILABLE IN THIS REVIEW

25   The following sections provide an overview of the information available in the current
26   review that would be expected to reduce uncertainties from the last review and to inform the
27   design and interpretation of updated analyses. Section 2.2.1 discusses the data available in the
28   current review to inform the characterization of ambient NO_2 concentrations, including
29   concentrations on and near roadways. Section 2.2.2 provides an overview of the health
30   information assessed in the 2nd draft ISA (U.S. EPA, 2015) that could inform the identification of
31   NO_2 health effect benchmarks in the current review. Section 2.2.3 presents staff's preliminary
32   conclusion that an updated analysis comparing ambient NO_2 concentrations to health effect
33   benchmarks is supported in the current review.

2.2.1   Characterizing Ambient NO_2 Concentrations

35   Given the importance of roadway-associated NO_2 concentrations in the last review, a
36   critical consideration in the current review is the extent to which new information could better
37   inform our understanding of ambient NO_2 concentrations on and near major roadways. When
38   evaluating the information available in this review to inform the characterization of ambient NO_2
39   concentrations, we consider the available ambient NO_2 measurement data (section 2.2.1.1),

1 information on important sources of NO$_X$ emissions (section 2.2.1.2), and information from

2 modeling analyses of ambient NO$_2$ concentrations (section 2.2.1.3).

3 **2.2.1.1 Ambient measurement data**

4   This section discusses the ambient measurement data available in the current review that

5 could provide the air quality basis for updated analyses. This includes data available from the

6 existing NO$_2$ ambient monitoring network (section 2.2.1.1.1), including the recently deployed

7 near-road monitors and data available from research studies that have characterized ambient NO$_2$

8 concentrations (section 2.2.1.1.2).

9 ***2.2.1.1.1   NO$_2$ ambient monitoring network***

10   The existing NO$_2$ ambient monitoring network in the U.S. includes over 400 monitors.

11 Ambient NO$_2$ monitors are sited to represent various spatial scales, including microscale (in

12 close proximity, up to 100 m from a source), middle scale (several city blocks, 100 to 500 m),

13 neighborhood scale (0.5 to 4 km), and urban scale (4 to 50 km) (40 CFR Part 58, Appendix D).[24]

14 In the last review of the primary NO$_2$ NAAQS, EPA promulgated new monitoring requirements

15 mandating that state and local air monitoring agencies install near-road NO$_2$ monitoring stations

16 in large urban areas. Under these new requirements, state and local air agencies will operate one

17 near-road NO$_2$ monitor in any CBSA with a population of 500,000 or more and two near-road

18 NO$_2$ monitors in CBSAs with populations of 2,500,000 or more or in any CBSA with a

19 population of at least 500,000 and with roadway segments carrying traffic volumes of at least

20 250,000 vehicles per day. These monitors are intended to measure ambient NO$_2$ concentrations

21 in the near-road environment where evidence indicates that peak ambient NO$_2$ concentrations

22 due to on-road mobile source activity can occur. The network is developing over time; the first

23 of three phases became operational in January of 2014 and the second phase in January of 2015.

24 In the current review, these near-road monitors will provide a key source of new information on

25 NO$_2$ concentrations around major roadways.

26   Table 2-1, below, lists the CBSAs with near-road monitors currently in operation. All

27 near-road monitors are required to be within 50 m of the target roadway, though the majority are

28 within 30 m. Any updated air quality analyses conducted in the current review will consider

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[24] Criteria for siting ambient NO$_2$ monitors are given in the State and Local Air Monitoring Stations/National Air Monitoring Stations/Photochemical Monitoring Stations (SLAMS/NAMS/PAMS) Network Review Guidance (U.S. EPA, 1998).

1   information from these monitors, as well as updated information on ambient NO_2 concentrations
2   from the entire monitoring network, as it becomes available.
3   In addition to the newly available hourly NO_2 concentrations from the near-road
4   monitors, updated air quality information is also available in the current review from the broader
5   NO_2 ambient monitoring network. Based on these monitors, Figures 2-11 and 2-12 of the 2nd
6   draft ISA (U.S. EPA, 2015) summarize the 98th percentiles of daily maximum 1-hour NO_2
7   concentrations and annual average NO_2 concentrations, respectively. From 2011 to 2013, all
8   areas of the U.S. met the existing primary NO_2 NAAQS (U.S. EPA, 2015, Figures 2-11 and 2-
9   12, Tables 2-3 and 2-4). For the NO_2 air quality assessment in this review, we will consider
10   further updated information from these monitors as it becomes available.

11

Table 2-1. Near-road NO₂ monitoring sites – active as of September 2014.

CBSA Name	Target Road	Annual Average Daily Traffic	Distance to Road (m)	Probe Height (m)	Start Date
Atlanta-Sandy Springs-Roswell, GA	I-85	284,920	2	4.5	6/15/14
Austin-Round Rock, TX	I-35	188,150	27	4	4/16/14
Baltimore-Columbia-Towson, MD	I-95	186,750	16.15	4	4/1/14
Birmingham-Hoover, AL	I-20	141,190	23.2	5.5	1/1/14
Boise, ID	I-84	103,000	32	4.6	4/1/12
Boston-Cambridge-Newton, MA-NH	I-93	193,000	10	4	6/1/13
Buffalo-Cheektowaga-Niagara Falls, NY	I-90	131,019	20	4	3/24/14
Charlotte-Concord-Gastonia, NC-SC	I-77	153,000	30	4.5	6/22/14
Cincinnati, OH-KY-IN	I-75	163,000	8	4.7	1/1/14
Cleveland-Elyria, OH	I-271	153,660	-	-	9/1/2014
Columbus, OH	I-270	142,361	32	5.3	1/1/14
Dallas-Fort Worth-Arlington, TX	I-635	235,790	24	4	4/2/14
Denver-Aurora-Lakewood, CO	I-25	249,000	8.7	5	6/1/13
Des Moines-West Des Moines, IA	I-235	110,000	13	3	1/1/13
Detroit-Warren-Dearborn, MI	I-96	140,500	8.5	5.2	7/27/11
Hartford-West Hartford-East Hartford, CT	I-84	159,900	17.7	3.6	4/1/13
Houston-The Woodlands-Sugar Land, TX	I-69/US 59	324,119	24	4	1/22/14
Indianapolis-Carmel-Anderson, IN	I-70	189,760	24.5	4	2/7/14
Jacksonville, FL	I-95	139,000	20	4.6	1/1/14
Kansas City, MO-KS	I-70	114,495	20	3	1/1/14
Los Angeles-Long Beach-Anaheim, CA	I-5	272,000	9	4.5	1/1/14
Louisville/Jefferson County, KY-IN	I-264	163,000	32	4.7	1/1/14
Memphis, TN-MS-AR	I-40	140,850	23.75	4.3	7/1/14
Milwaukee-Waukesha-West Allis, WI	I-94	133,000	14	3.5	1/1/14
Minneapolis-St. Paul-Bloomington, MN-WI	I-94/I-35W	277,000	32.5	4.9	4/1/13
Nashville-Davidson–Murfreesboro–Franklin, TN	I-40/I-24	144,204	30	4.5	1/1/14
New Orleans-Metairie, LA	I-610	68,015	28.5	4.2	3/18/14
New York-Newark-Jersey City, NY-NJ-PA	I-95/US 1	311,234	20	4.6	6/26/14
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	I-95	124,610	12	5	1/1/14
Phoenix-Mesa-Scottsdale, AZ	I-10	320,138	12	5.1	2/13/14
Pittsburgh, PA	I-376	87,534	18	3	7/29/14
Portland-Vancouver-Hillsboro, OR-WA	I-5	156,000	25	3	4/21/14
Providence-Warwick, RI-MA	I-95	186,300	5	3.9	4/1/14
Raleigh, NC	I-40	141,000	20	4.3	1/1/14
Richmond, VA	I-95	151,000	20	3.3	10/17/13
Riverside-San Bernardino-Ontario, CA	I-10	245,300	50	4.5	8/1/14
San Antonio-New Braunfels, TX	I-35	201,840	20	4	1/8/14
San Francisco-Oakland-Hayward, CA	I-880	216,000	20	6.4	2/1/14
San Jose-Sunnyvale-Santa Clara, CA	US 101	191,000	32	6.4	8/21/14
Seattle-Tacoma-Bellevue, WA	I-5	237,000	4.5	3	3/24/14
St. Louis, MO-IL	I-64	159,326	25	3	1/1/13
Tampa-St. Petersburg-Clearwater, FL	I-275	190,500	20	5	3/1/14

2.2.1.1.2 *NO$_2$ measurement research studies*

As noted above (section 2.1), the 2008 REA simulated on-road NO$_2$ concentrations by applying a distribution of factors to NO$_2$ concentrations at monitor locations $\geq$ 100 m from the road. In the current review, in addition to the data from recently sited near-road NO$_2$ monitors, our characterization of roadway-associated NO$_2$ concentrations will be informed by available research studies that have evaluated ambient NO$_2$ concentrations on-road (either in-traffic or curbside) and near roadways.

The 2nd draft ISA identifies a number of studies that have characterized ambient NO$_2$ concentrations around roadways (U.S. EPA, 2015, section 2.5.3). Most studies that were available in the last review used passive samplers requiring sampling periods on the order of a week or longer (U.S. EPA, 2008a, section 7.2).[25] This was identified as one of the important uncertainties in estimating on-road concentrations (i.e., simulating 1-hour on-road NO$_2$ concentrations based on studies that used longer averaging times). Thus, at the time of the last review we had a limited understanding of short-term (1-hour) NO$_2$ concentrations on/near-roads relative to concentrations measured away from roads. Several recent studies have used sampling methods that allow for improved temporal resolution (U.S. EPA, 2015, section 2.5.3.1, Table 2-6, Figures 2-17 and 2-18) and, as described below (section 2.3), information from such studies could inform an updated characterization of on- and near-road NO$_2$ concentrations in the current review.

There are two recent near-road transect measurement studies conducted in the U.S. where temporally and spatially refined NO$_2$ concentration data will be available for this review. The first study was conducted in Las Vegas, Nevada from December 2008 to 2009 (Kimbrough et al., 2013). This study used continuous gas analyzers to collect 5-minute measurements of NO$_2$ and NO$_X$ concentrations at both upwind (100 m) and downwind (20, 100, and 300 m) sites from a major roadway (Interstate 15). The four monitoring locations used in the study were selected along an east-west transect, approximately perpendicular to the roadway and occurring along a railroad spur right-of-way. The second study was conducted in Detroit, Michigan from September 2010 to June 2011 (Batterman et al., 2014). This study also collected 5-minute NO$_2$

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[25] The exception was the study by Rodes and Holland (1981), which evaluated 1-hour NO$_2$ concentrations in Los Angeles, CA. Authors reported that hourly NO$_2$ concentrations were about 80 to 200% higher near a major highway (8 m from road) than concentrations away from the highway (400 to 500 m from the road). Because this is an older study, the 2nd draft ISA notes that "the vehicle fleet was not strictly regulated for NO$_X$ emissions" and "[a]s a result, the concentrations observed may not be relevant to current conditions" (U.S. EPA, 2015, p. 2-55).

1 concentrations. While the Detroit site did not favor a perpendicular transect (to the freeway)
2 similar to the one used in Las Vegas, four stations were deployed for this study—three
3 downwind and one upwind. Near-road measurement data from these two near-road transect
4 studies are expected to provide information regarding the overall relationship between NO_2
5 concentrations and distance from the roadway. Data from these studies could inform or be used
6 to evaluate an updated mathematical/statistical approach to use in simulating on-road
7 concentrations using concentrations at a distance from the road and/or could be used to evaluate
8 similar on- and near-road concentrations predicted using air quality.
9   Further, there are recent air pollution roadway studies that, in addition to having near-road
10 measurements, also collected on-road NO_2 concentrations. As part of EPA's Geospatial
11 Measurement of Air Pollution (GMAP) program, mobile and stationary measurements of NO_2
12 concentrations were collected in five study areas, two of which may have temporally and
13 spatially informative NO_2 data available for this review to support the approach proposed to
14 simulate on-road concentrations. The first study was conducted in Research Triangle Park, North
15 Carolina, during morning-hour commutes (7:00 AM-11:30 AM) from August to October 2012
16 (Mukerjee et al., 2015). This study used an electric vehicle instrumented with a cavity attenuated
17 phase shift spectrometry-based monitor to measure in-traffic NO_2 concentrations at 1-second
18 intervals on an interstate, major arterials, and collector roadways. Fixed site measurements
19 included meteorological parameters only (wind speed and direction) and not NO_2 concentrations,
20 an important study limitation to directly informing on-road to near-road concentration
21 relationships in this assessment. Study results however, could provide insight into the spatial
22 distribution of on-road NO_2 concentrations in an urban study area and the effect of important
23 influential factors.
24   A second GMAP study that could provide more complete data for informing the on-road
25 to away-from-road concentration relationship was conducted in Phoenix, Arizona from October
26 to November 2013 (Baldauf et al., 2015). This study also used a mobile and fixed-site
27 measurement approach along two segments of Interstate 17. In-traffic NO_2 measurements were
28 made at 1-second intervals during morning (9:00 AM-12:00 PM) and afternoon hours (2:00-5:00
29 PM) on interstate, arterial, collector, and residential roadways giving on-major-road (i.e., the
30 interstate) and away-from-major-road (i.e., arterial and residential) NO_2 measurements. We

1   expect that the measurement data from both of these studies could provide further support to

2   developing a factor(s) to simulate on-road concentrations from existing near-road monitor data.[26]

3   **2.2.1.2 Emissions information**

4   If updated air quality analyses are conducted, information on NO_X emissions[27] will inform

5   our characterization of the important sources contributing to monitored NO_2 concentrations. This

6   section provides an overview of the information available on NO_X emissions at a national level

7   based on recent updates to EPA's National Emissions Inventory (NEI) and incorporated into the

8   2011 NEI.[28]

9   At a national level, anthropogenic sources account for more than 90% of NO_X emissions

10   in the 2011 NEI. Vehicles are the largest source, with highway and off-highway vehicles

11   contributing almost 60% of the total NO_X emissions nationally. Other important sources include

12   fuel combustion-utilities (14% of total), fuel combustion-other (11% of total), and biogenics and

13   wildfires (8% of total) (U.S. EPA, 2015, section 2.3.1, Figure 2-3). Compared to the national

14   averages, urban areas have greater contributions to total NO_X emissions from both highway

15   vehicles and off-highway vehicles and smaller contributions from other sources (U.S. EPA,

16   2015, Figure 2-4, Table 2-1). For example, in the 21 largest CBSAs in the U.S., more than half

17   of the urban NO_X emissions are from highway vehicles. Together, highway vehicles and off-

18   highway vehicles and engines account for more than three quarters of total emissions in these

19   large CBSAs (U.S. EPA, 2015, section 2.3.2).

20   While an emissions summary at a national level is useful, important emissions sources can

21   vary across locations. As discussed below (section 2.3.2), NO_X emissions sources, including

22   mobile sources and important stationary sources, will be characterized in more detail in specific

23   urban study areas selected for any updated air quality analyses.

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[26] The three additional study areas identified in the GMAP program include Detroit, Michigan; San Francisco, California; and Charleston, South Carolina.

[27] Oxidized nitrogen compounds are emitted to the atmosphere primarily as NO, with NO converting to NO_2 following its reaction with O_3. Collectively, NO and NO_2 are referred to as NO_X (U.S. EPA, 2008b, section 2.2).

[28] The NEI is a national compilation of emissions estimates from all source sectors, collected from state, local, and tribal air agencies as well as those developed by EPA. The NEI is developed on a tri-annual basis, with 2011 being the most recent base year currently available and referred to as 2011 NEI. The next NEI base year will be 2014 and will be available in 2016. For information on the NEI, see http://www.epa.gov/ttn/chief/eiinformation.html.

2.2.1.3 NO$_2$ modeling research studies

Two new modeling analyses could also inform our characterization of ambient NO$_2$ concentrations around roadways in the current review. Model estimated NO$_2$ concentrations at road-side receptors and at varying distances from major roadways could become available in this review to further inform an updated approach to simulate on-road NO$_2$ concentrations. The intended purpose of these modeled concentrations is to provide support for estimating on-road concentrations based on either using the new near-road monitor concentrations or using other away-from-road concentrations. Having modeled concentrations at varying distances from a road affords great flexibility in developing potential on-road simulation factors to be used, particularly in knowing the influential factors that could affect the relationship between road-side and away-from-road concentrations (e.g. wind speed/direction, mixing heights, presence of nearby stationary sources)

The first modeling analysis used hourly link-based emissions varied by day type (weekday vs. weekend) and hour of day developed for 17 road segments of interstate 95 in Broward County, Florida (including Pompano, Ft. Lauderdale, and Dania beaches) using EPA's AERMOD dispersion model (Thurman et al., 2013). The model was used to predict hourly NO$_2$ concentrations at road-side receptors generally spaced 5, 10, 20, 25, 50, 75, and 100 m from the road, as well as to predict roadway median concentrations over a five-year period (2006-10). While there are six ambient monitors available that measured NO$_2$ concentrations to evaluate model predictions, their time-averaging (weekly-average) and siting far from the roads evaluated are an important limitation to this analysis.

The second modeling analysis is planned to technically correspond (e.g., same site characteristics, meteorology, years) to the 2008-09 Las Vegas, NV measurement study (Kimbrough et al., 2013) described above in section 2.2.1.1.2. The planned approach is to be similar to that used for the Broward County modeling analysis, though differing by using the latest version of AERMOD, incorporating the most recent emissions data, and in having a robust measurement data set available (i.e., to better inform and evaluate existing model parameter settings and assumptions). Once complete, it is expected that these modeled data could provide further support to developing a simulation factor(s) or other approach to simulate on-road concentrations from existing near-road monitoring data or from existing area-wide monitoring data.

2.2.2 Evidence Informing Health Effect Benchmarks

The primary goal of any updated NO$_2$ air quality analyses will be to inform conclusions regarding the likelihood that the existing or potential alternative standards would allow for exposures to ambient NO$_2$ concentrations that could be of concern for public health. One way to

1   accomplish this, as was done in the 2008 REA , is to compare ambient NO_2 concentrations

2   adjusted to just meet the existing (and alternative, if appropriate) NO_2 standards with health

3   effect benchmarks.

4   In the last review, 1-hour NO_2 health effect benchmarks from 100 to 300 ppb were

5   evaluated. These benchmarks were based on the results of an ISA meta-analysis of individual-

6   level data from controlled human exposure studies of non-specific airway responsiveness in

7   people with asthma.[29] The results of this meta-analysis indicated that the majority of study

8   volunteers experienced increased airway responsiveness following exposures to NO_2

9   concentrations of 100 to 300 ppb (or higher) for 30 minutes to 2 hours (U.S. EPA, 2008b, Tables

10   3.1-2 and 3.1-3). At the time of the last review, airway responsiveness was the only health effect

11   endpoint for which controlled human exposure studies had reported effects following exposures

12   to NO_2 at or near ambient concentrations (U.S. EPA, 2008b, section 3.4).[30]

13   In the current review, the evidence to inform the identification of NO_2 health effect

14   benchmarks has not changed substantially from that used in the 2008 REA, though the 2nd draft

15   ISA includes expanded analyses of that evidence (U.S. EPA, 2015, section 5.2.2.1). Specifically,

16   the 2nd draft ISA includes an expanded evaluation of the magnitude and potential clinical

17   relevance of reported changes in airway responsiveness[31] and a discussion of the limitations

18   impacting characterization of the exposure-response relationship between NO_2 and airway

19   responsiveness (U.S. EPA, 2015, section 5.2.2.1). The updated meta-analysis in the 2nd draft ISA

20   also presents results based on a broader range of studies, including studies of both non-specific

21   airway responsiveness and of specific airway responsiveness following allergen challenge (U.S.

22   EPA, 2015, Tables 5-5 and 5-6). While the available evidence has not changed substantially, as

23   summarized below these expanded analyses and discussions provide additional information to

24   better inform our understanding of the potential public health implications of exposures at or

25   above NO_2 health effect benchmarks of 100 ppb and higher.

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[29] Increased airway responsiveness indicates the potential for worsened control of asthma symptoms. Non-specific airway responsiveness in these studies was assessed using stimuli such as carbachol, methacholine, histamine, cold air, or SO_2 (U.S. EPA, 2008b, Table 3.1-2).

[30] The lowest NO_2 exposure concentration for which airway responsiveness has been evaluated is 100 ppb.

[31] Analyses of the individual data from a subset of studies indicates that a statistically significant fraction of study participants exposed to NO_2 concentrations from 100 to 530 ppb experienced clinically-relevant increases in airway responsiveness, indicated by a halving of the provocative dose (U.S. EPA, 2015, Figure 5-1). Evidence for clinically relevant increases was stronger for NO_2 exposure concentrations ≤ 250 ppb (80%, p=0.035) than for concentrations >250 ppb (73%, p=0.052).

1   As was the case in the last review, 100 ppb NO_2 is the lowest exposure concentration that
2   has been evaluated for its impact on airway responsiveness. Of the five studies assessed in the
3   ISA that evaluated exposures to 100 ppb NO_2 (Orehek et al., 1976; Ahmed et al., 1983a; Ahmed
4   et al. 1983b; Hazucha et al., 1983; Tunnicliffe et al., 1994), one (Orehek et al., 1976) was later
5   reported (Dawson and Schenker, 1979) to show statistically significant increases in airway
6   responsiveness. When individual data from across studies was analyzed together in a meta-
7   analysis, 66% of study participants (a statistically significant percentage; $p < 0.05$) were
8   estimated to experience a NO_2-induced (i.e., relative to filter air) increase in non-specific airway
9   responsiveness following 1-hour exposures to 100 ppb NO_2 (U.S. EPA, 2015, Table 5-4). When
10   this meta-analysis was updated to also include information from studies evaluating specific
11   airway responsiveness following allergen challenge, 61% of study participants experienced
12   increased airway responsiveness following exposures to 100 ppb NO_2 (a marginally statistically
13   significant percentage; $p = 0.08$) (U.S. EPA, 2015, Table 5-6).
14   Controlled human exposure studies have also evaluated airway responsiveness following
15   exposures to NO_2 concentrations greater than 100 ppb (U.S. EPA, 2015, Table 5-2).[32] These
16   include one study that did not report a statistically significant increase in airway responsiveness
17   following resting exposures to 140 ppb NO_2 (Bylin et al., 1988); five studies that evaluated
18   resting exposures to NO_2 concentrations from 200 to 270 ppb (Orehek et al., 1976; Jorres et al.,
19   1990; Bylin et al., 1988; Strand et al., 1997; 1998; Barck et al., 2002), three of which reported
20   statistically significant NO_2-induced increases in airway responsiveness (i.e., at 250 to 270 ppb);
21   and four studies that evaluated resting exposures to NO_2 concentrations from 400 to 530 ppb
22   (Bylin et al., 1985; Mohsenin et al., 1987; Bylin et al., 1988; Tunnicliffe et al., 1994), three of
23   which reported statistically significant NO_2-induced increases in airway responsiveness (i.e., 400
24   to 500 ppb).
25   When individual-level data were combined across subsets of these studies, the meta-
26   analyses in the 2nd draft ISA indicate that statistically significant majorities of study participants
27   experienced increased airway responsiveness following exposures to NO_2 concentrations from
28   100 to 200 ppb (labeled as "$100 \leq [NO_2] < 200$" in U.S. EPA, 2015, Tables 5-4 and 5-6); 200 to

[32] These include studies where participants were at rest during exposure periods and studies where participants engaged in exercise during exposures. As noted in the 2nd draft ISA, "the literature on airway responsiveness supports the development of a refractory period following bouts of exercise" (U.S. EPA, 2015, p. 5-35). Consistent with the possibility that exercise may lead to a period of reduced airway responsiveness, the 2nd draft ISA notes larger increases in airway responsiveness in studies with participants at rest than in studies with participants engaged in exercise. In identifying NO_2 health effect benchmarks, we focus on studies that evaluated participants while at rest.

1   270 ppb (labeled as "200 ≤ [NO$_2$] ≤ 300" in U.S. EPA, 2015, Tables 5-4 and 5-6), and 400 to

2   530 ppb (labeled as "[NO$_2$] > 300" in U.S. EPA, 2015, Tables 5-4 and 5-6). These percentages

3   were statistically significant in analyses that included data from studies of non-specific airway

4   responsiveness and in analyses that combined data from studies of non-specific and specific

5   airway responsiveness.

6   With regard to the health benchmarks appropriate for evaluation in this review, 100 ppb is

7   the lowest NO$_2$ exposure concentration for which the evidence indicates the potential for NO$_2$-

8   induced increases in airway responsiveness. Given this, we reach the preliminary conclusion that

9   100 ppb is an appropriate health effect benchmark to evaluate. However, we also recognize the

10   important uncertainties associated with the evidence for increased airway responsiveness

11   following exposures to 100 ppb NO$_2$. These include the general lack of statistically significant

12   results in individual studies at 100 ppb and the lack of an exposure-response relationship based

13   on available studies. Such uncertainties will be taken into consideration when interpreting the

14   potential public health implications of NO$_2$ air quality concentrations that equal or exceed the

15   100 ppb health effect benchmark.

16   With respect to exposures to higher NO$_2$ concentrations, meta-analyses of pooled data

17   consistently indicate that statistically significant majorities of study participants experienced

18   increased airway responsiveness. In addition, individual studies have reported NO$_2$-induced

19   increases in airway responsiveness with greater consistency and statistical precision. Thus, as

20   NO$_2$ exposure concentrations increase, we have increasing confidence in estimates of the

21   percentage of individuals with asthma who could experience increased airway responsiveness.

22   Given the evidence and the results of meta-analyses presented in the 2nd draft ISA, we reach the

23   preliminary conclusion that it is appropriate in the current review to evaluate NO$_2$ health effect

24   benchmarks as high as 400 ppb.

25 ### 2.2.3 Preliminary Conclusions

26   As indicated in section 2.1.1 above, an important uncertainty identified in the 2008 REA

27   was the characterization of 1-hour NO$_2$ concentrations on and around roadways, given the

28   limited information available in the last review. Based on the information discussed above, we

29   have a substantially improved body of information available in the current review to inform an

30   updated characterization of 1-hour NO$_2$ concentrations around roadways (section 2.2.1). In

31   particular, data from recently deployed NO$_2$ monitors near major roads, combined with new

32   information from monitoring and modeling studies of NO$_2$ concentration gradients around roads,

33   will substantially improve our understanding of ambient NO$_2$ concentrations in the on-road and

34   near-road environments. This new information is expected to provide important perspective,

35   beyond what is available from the last review, on the extent to which NO$_2$ exposures on and near

1 roads could have potentially important implications for public health. Therefore, we reach the

2 preliminary conclusion that an updated analysis comparing ambient NO_2 concentrations to health

3 effect benchmarks is supported in the current review, with a particular focus on updating

4 analyses of concentrations on and near major roadways.

2.3 PROPOSED ANALYTICAL APPROACH AND ILLUSTRATIVE EXAMPLE

7 Given the preliminary conclusion that updated analyses comparing ambient NO_2

8 concentrations to health effect benchmarks are supported in the current review, this section

9 describes our proposed technical approach to conducting such analyses (section 2.3.1) and

10 preliminary results for an example urban study area (section 2.3.2).

2.3.1 Overview of Proposed Approach

12 Conducting an air quality assessment requires health effect benchmark concentrations of

13 concern for the general population or sensitive study group of interest, identification of a study

14 area(s) of interest, and characterization of respective air quality (including measured, adjusted,

15 and simulated, depending on the air quality scenario and concentration type). Each of these

16 components and an overview of the output data metrics are described in the following sections.

2.3.1.1 Identification of health effect benchmark levels

18 An evaluation of the controlled human exposure-based literature in the 2nd draft ISA (U.S.

19 EPA, 2015) and summarized in section 2.2.2 above has identified a range of 1-hour

20 concentrations to consider in this air quality assessment. Because there is no apparent dose-

21 response relationship (see section 4.1 below), a range of concentrations of concern (100-400

22 ppb) will be evaluated in 100 ppb increments, yielding benchmark levels of 100, 200, 300, and

23 400 ppb. Instances when ambient concentrations in selected study areas are at or above these

24 levels will be counted and summarized using the approach described in the following sections

25 (2.3.1.2 and 2.3.1.3).

2.3.1.2 Initial selection of study areas

27 While all of the existing ambient monitoring data are considered in this assessment, a few

28 of the air quality scenarios (e.g., air quality adjusted to just meet the existing standards) and

29 microenvironmental evaluations (e.g., on-road concentrations) warrant the defining of a specific

30 geographic domain. The following are the proposed criteria for identifying study areas to

31 evaluate in the air quality assessment, followed with a list of candidate study areas when

32 applying the criteria to recent (2010-2013) ambient concentrations.

33

1. One of the most important attributes of a study area is the ability to use the monitoring data available to characterize the NO_2 area-wide and microenvironmental concentrations (both the highest annual and daily maximum 1-hour, DM1H) within the CBSA (e.g., near roadway, area influenced by significant stationary source(s), and to a lesser extent, background concentrations). Monitors sited within each CBSA will be identified using delineation files available through the U.S. Census Bureau.[33] Then, the distribution of hourly concentrations for each monitor year is screened to assess whether or not standard completeness criteria are met. Ascertaining a valid year of monitoring data is a multi-step process. First, valid days are defined as those having at least 18 hours of measurements. Next, a valid quarter is identified as having at least 75% of valid days within a three-month calendar period (~68-70 days). Finally, where all four quarters in a calendar year are valid, the year of monitoring data is considered complete. Ambient monitor data will be grouped into the following four categories:

   a. *Area-wide concentrations*. CBSAs having the maximum number of monitors and monitoring years of data will be given selection preference, considering the availability of both recent and historical air quality data.

   b. *Near-road concentrations*. CBSAs having near-road monitoring data will be given preference for selection, particularly if they meet completeness criteria. However, given that most of these monitors just began collecting concentrations in 2014, near-road data will not be excluded for having an incomplete year. Planned analyses would consider the relationship of near-road to area-wide concentrations, particularly when simultaneous measurements are collected. These analyses can remain informative if using

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[33] The counties comprising each CBSA are listed at http://www.census.gov/population/metro/data/def.html, as originally defined by the Office of Management and Budget (OMB) February 2013 Bulletin 13-01 (available at http://www.whitehouse.gov/sites/default/files/omb/bulletins/2013/b-13-01.pdf). Rather than using partial counties (if any were identified as such for a given CBSA), a monitor was included as part of the CBSA if it were situated anywhere within the listed county.

less than a complete year of data, particularly when understanding seasonal variation. CBSAs having the maximum number of hours in a year simultaneously monitored at near-roads and area wide monitors would be given selection preference. CBSAs not having a new near-road monitor would still be considered as a potential study area,[34] though still would require a minimum number of area-wide monitors along with having other potential near-source monitoring data (subsection c. and d. immediately below).

  c. *Background concentrations*. Of lesser importance than criteria a. and b., the selected study area should have valid hourly concentrations measured at either a background or low-concentration monitor to some provide context for better understanding spatial variability in concentrations across the study area (i.e., concentrations relative to those in likely high-concentration environments) and/or possibly for use in estimating on-road NO_2 concentrations in a generally similar manner as was done in the 2008 REA.

  d. *Other high NO_2 concentration environments*. Of lesser importance than criteria a., b., and c., CBSAs having valid monitoring data that can characterize potential highly influential emission sources other than on-road (e.g., stationary sources, airports) will be given preference for selection.

2. CBSAs having the highest annual and/or DM1H concentrations in the U.S. will be given preference for selection as a study area. Justification for this criterion would include, 1) monitors having the highest concentrations would require the smallest adjustment upwards to just meet the existing standard, possibly limiting uncertainty in generated results and 2) the risk associated with highest concentrations (even considering unadjusted concentrations) is by definition of greatest importance when performing an assessment that uses health effect benchmark concentrations. Existing design values would be used to inform this selection criterion.[35]

3. The list of selected CBSAs should capture areas where large portions of the U.S. population reside, as this is a better representation of potential risks to populations at

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[34] There are a few CBSAs that have an existing monitor sited in close proximity to a roadway not necessarily meeting the current near-road monitor requirements. For example, Chicago monitor ID 170313103 and El Paso monitor ID 481410044 were estimated to be about 20 m and 38 m, respectively from a major road in the 2008 REA (Appendix A, Table A-7).

[35] Monitor design values or the annual and 1-hour primary standards (annual average and 98th percentile DM1H averaged across 3-years, respectively) for the period extending from 2002-2013 are available at http://www.epa.gov/airtrends/values.html.

1	an local, urban, and national scales as well as increasing the likelihood for
2	appropriately representing important study groups (e.g., children with asthma). In
3	addition, study area selection will be further guided by overall geographical location
4	(e.g., climatic regions of the U.S.) to adequately represent areas across the U.S.
5	having seasonal, atmospheric, or other influential factors that contribute to variability
6	in concentrations.
7	4. Following the initial screening of potential study areas described above, additional
8	information and data analyses are required to inform the decision to select study
9	areas, but more so to retain an initially selected study area for analysis in the air
10	quality assessment. The purpose is to provide additional support to the approach used
11	to adjust NO_2 concentrations to just meet the existing standards, not just at the highest
12	design value monitors alone, but also considering all monitors in a CBSA (area-wide,
13	near-road, other source-oriented, where available). Further, the availability of this
14	information and data analyses could also inform decisions made in simulating other
15	high NO_2 concentrations not captured by the existing ambient measurements. This
16	criterion is mostly applied during the early stages of the air quality assessment, with
17	some preliminary concentration analyses provided for study area selection here.[36] The
18	available information and data analyses needed would include:
19	a. *Ambient monitor meta-data.* Preference for retaining an initially identified
20	study area would be given to CBSAs having information readily available that
21	characterizes important monitor site attributes (e.g., geographic coordinates,
22	local land use, monitor type, etc.) to indicate potentially important emission
23	sources, such as roads and stationary emission sources.
24	b. *Local NO_X source emissions data*. Preference for retaining an initially
25	identified study area would be given to CBSAs having proximally located and
26	detailed source emissions information available to characterize potential
27	individual monitor near-source influences.

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[36] In general, most CBSAs selected using the first three criteria would generally meet this fourth selection criterion when considering the extent of available metadata used to describe the attributes of each monitor. However, some elements of the proposed evaluation, such as the historical concentrations, number of active monitors, or the degree of specificity regarding proximal emission source types will not necessarily inform the selection of a study area but rather inform the characterization of uncertainties associated with concentrations adjusted to just meet air quality standards, the simulated high-concentration environments (if any), and the estimated number of benchmark exceedances. We provide a preliminary application of the fourth criterion below, largely as an assessment of inter-monitor concentration ranges and correlations using the most recent data (2011-2014, See Table 2-3). A detailed application of the fourth criterion is given in the illustrative example that follows in section 2.3.

1   c. *Historical ambient monitoring concentrations.* Preference for retaining an
2   initially identified study area would be given to CBSAs where there are
3   substantial data or information available to characterize trends in NO_2
4   concentrations that reflect changes in emissions over a period of time. Needed
5   are a collection of monitors having both current (2010-2014) and historical
6   (1980's – 2000's) concentrations. Of interest are intra- and inter-monitor
7   concentration ranges and correlations, particularly at the upper percentiles of
8   DM1H concentration distribution.

9   We first evaluated the recent ambient monitoring data using the first three selection
10   criteria above, with results summarized in four tables that follow here.

11   Table 2-2 indicates the CBSAs having a newly sited near-road monitor (and expected to
12   have near-road data available for this review). In using the most recent design values available
13   for the two 3-year averaging periods from 2010-2013, the CBSAs are listed in order first by the
14   maximum number of monitors available in a given year or over a 3-year averaging period and
15   then by 2010-2012 design values. Three categories ("strong", "moderate", or "limited") are used
16   to characterize the strength of information to support the selection of a CBSA as a study area for
17   the air quality analysis. Regarding the number of monitors available, having a minimum of three
18   area-wide monitors indicated moderate support given that this number can form the simplest 2-
19   dimensional geometric shape (and to a limited extent, approximate an air quality surface), with
20   CBSAs having greater than 3 monitors indicating strong support. The thresholds used for
21   categorizing the design values were based on the overall distribution of concentrations, selecting
22   for where concentrations were greatest (and thus requiring the least adjustment for just meeting
23   the existing standard). For annual average concentrations, the general range of design values
24   extended from a few to 30 ppb, thus the upper portions of this range (i.e. ≥15 ppb) indicated
25   areas having moderate support, with those ≥20 ppb indicated areas having strong support. For the
26   DM1H design value, having a value of at least 50 ppb (or a factor of 2 less than the standard
27   level of 100 ppb) indicated areas having moderate support, with those ≥60 ppb indicated areas
28   having strong support. And finally, CBSAs identified as a strong candidates for the air quality
29   analyses had at a minimum, a strong rating for the number of monitors available and at least a
30   moderate rating when considering design values, while CBSAs identified as a possible
31   candidates had moderate ratings for the number of monitors available. Eleven CBSAs are
32   identified as strong candidates using these criteria, with an additional six CBSAs indicated as
33   possible candidates to include in our air quality assessment.

34   Table 2-3 summarizes preliminary analyses of the ambient concentration data for CBSAs
35   having newly sited near-road monitors, considering all hours where the newly sited near-road
36   monitors have measured NO_2 concentrations. Similar to the application of the selection criteria

1 described above though considered here are availability of concurrent measurements at the area-
2 wide monitors and near-road monitors. "Moderate" or "strong" support was indicated by the
3 number of available area-wide monitors (at least 3 or >3, respectively), the range in 98th
4 percentile DM1H concentrations (at least 10-20 ppb or ≥20 ppb, respectively), the number of
5 years of on-road monitoring (>1 year, indicated strong support only) and number of hours per
6 year (≥4,000 and ≥6000 hours, or about 50% or 75% or the year, respectively).[37] In this
7 evaluation, CBSAs identified as strong candidates was largely based on the having a strong
8 characterization assigned for the number of available monitors as well having a strong
9 characterization regarding the range in concentrations. CBSAs identified as possible candidates
10 had at least a moderate rating assigned for the number of available monitors. In reviewing this
11 table, one CBSA (i.e., St. Louis) is now elevated from being a possible to strong candidate,
12 bringing the total number of CBSAs strongly considered to twelve. In addition, two new CBSAs
13 are now considered as possible candidates to include (i.e., Denver, Providence), when
14 considering the number of near-road 1-hour measurements, NO_2 concentration ranges and
15 correlations of upper percentile concentrations with area-wide monitors.
16   Finally, Table 2-4 provides a summary of 2010-2013 design value information similar as
17 that in Table 2-2, though for CBSAs for which we do not anticipate having data available from
18 newly sited near-road monitors. We categorized the supporting information regarding the
19 number of area-wide monitors available and respective design values identical to that used for
20 Table 2-2. Based on the selection criteria, four additional CBSAs are indicated as strong
21 candidates, along with two additional CBSAs indicated as possible candidates to evaluate. Thus
22 to summarize the application of the first two study area selection criteria, 16 CBSAs are
23 identified a strong candidates, while 10 CBSAs are identified as possible candidates.
24   Regarding the third study area selection criterion, Table 2-5 lists CBSAs having at least
25 800,000 residents estimated for 2013, ordered by descending population. When considering the
26 16 CBSAs indicated above as strong candidates, these CBSAs collectively would include just
27 over 97 million people (or approximately 31% of total U.S. population). When considering the
28 10 CBSAs identified as possible candidates, an additional 27 million people could be included to
29 the group of study areas, thus comprising approximately 39% of the total U.S. population. The

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[37] Not entirely used for selecting study areas though informative to understanding relationship of near-road monitor concentrations with area-wide monitors were correlations of the upper percentile concentrations and the presence or not of a low-concentration monitor. A preliminary assessment of these particular attributes is provided in Table 2-2, however these criteria would be evaluated in greater detail when the air quality assessment is performed for a selected study area.

1   locations of these 26 study area candidates are illustrated in Figure 2-1. Included in the figure

2   also are the 10 most populated CBSAs identified as a limited study area candidate using the

3   selection criteria, most of which do or may have near-road monitoring data available for analysis

4   in this current review.

5

Table 2-2. Number of NO₂ monitors and 2010-2013 design values for CBSAs that have or are expected to have ambient concentration data available from newly sited near-road monitors.

CBSA Name/Selection Indicator		Number of monitors per year						Annual Average NO₂ (ppb)				DM1H 98th pct 3-year avg NO₂ (ppb)	
(Strong candidate)	(possible candidate)	2010	2011	2012	2013	2010-2012	2011-2013	2010 max	2011 max	2012 max	2013 max	2010-12 max	2011-13 max
Los Angeles-Long Beach-Anaheim, CA		16	14	9	13	4	5	26	25	21	23	67	64
Houston-The Woodlands-Sugar Land, TX		16	16	16	16	9	9	15	14	15	13	60	59
Riverside-San Bernardino-Ontario, CA		12	9	11	11	6	3	23	21	22	21	72	62
Dallas-Fort Worth-Arlington, TX		11	14	11	11	9	9	13	13	12	12	56	53
San Francisco-Oakland-Hayward, CA		8	10	11	10	8	8	16	16	15	17	74	68
San Diego-Carlsbad, CA		8	8	8	8	6	5	21	20	20	19	73	73
New York-Newark-Jersey City, NY-NJ-PA		7	7	8	8	5	6	22	25	22	22	70	67
Phoenix-Mesa-Scottsdale, AZ		6	5	5	5	4	4	25	25	26	25	66	64
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD		5	4	5	6	3	3	23	20	18	17	65	61
Pittsburgh, PA		5	5	5	6	4	4	15	16	14	11	53	49
Boston-Cambridge-Newton, MA-NH		5	5	6	5	5	4	19	20	19	18	51	50
Miami-Fort Lauderdale-West Palm Beach, FL		5	5	3	4	2	2	10	8	8	8	47	46
Atlanta-Sandy Springs-Roswell, GA		3	3	3	3	3	3	14	13	12	9	56	51
Kansas City, MO-KS		3	3	3	3	3	1	15	15	14	13	53	52
Richmond, VA		3	3	3	2	3	2	12	10	10	8	52	47
Minneapolis-St. Paul-Bloomington, MN-WI		3	3	3	3	2	3	10	9	11	9	46	44
St. Louis, MO-IL		3	3	2	3	0	2	13	13	14	11		53
Baltimore-Columbia-Towson, MD		2	2	2	2	2	2	18	18	16	15	57	52
Oklahoma City, OK		2	2	2	2	2	2	9	10	9	9	54	54
Tampa-St. Petersburg-Clearwater, FL		2	2	1	2	1	1	6	5	5	5	35	34
Cincinnati, OH-KY-IN		2	2	1	2	1	1	15	13	4	12	32	30
Buffalo-Cheektowaga-Niagara Falls, NY		2	1	2	0	0	0	13	8	10			
Denver-Aurora-Lakewood, CO		2	2	2	2	0	1	28	24	25	24		62
San Antonio-New Braunfels, TX		2	1	1	2	0	0	4	4	4	5		
Cleveland-Elyria, OH		1	1	1	1	1	1	16	15	14	13	52	50
San Jose-Sunnyvale-Santa Clara, CA		1	2	2	2	1	2	14	15	13	15	50	51
Milwaukee-Waukesha-West Allis, WI		1	1	1	1	1	1	13	11	12	10	49	49
Detroit-Warren-Dearborn, MI		1	1	2	2	1	1	12	12	13	12	48	44
New Orleans-Metairie, LA		1	1	1	1	1	1	8	8	8	6	48	48
Hartford-West Hartford-East Hartford, CT		1	1	1	1	1	0	10	11	9	8	46	
Nashville-Davidson-Murfreesboro-Franklin, TN		1	1	1	1	1	1	13	12	12	10	43	42
Des Moines-West Des Moines, IA		1	1	1	1	1	1	9	8	8	7	42	39
Providence-Warwick, RI-MA		1	1	2	2	1	1	10	11	10	10	42	43
Jacksonville, FL		1	1	1	1	1	1	9	8	8	8	40	38
Orlando-Kissimmee-Sanford, FL		1	1	1	1	1	1	6	5	5	5	36	34
Portland-Vancouver-Hillsboro, OR-WA		1	1	1	1	1	1	9	9	9	10	34	34
Austin-Round Rock, TX		1	1	0	1	0	0	3	3		5		
Boise City, ID		1	0	0	0	0	0	10					
Charlotte-Concord-Gastonia, NC-SC		1	1	1	1	0	0	12	10	9	8		
Indianapolis-Carmel-Anderson, IN		1	1	1	2	0	0	13	11	10	12		
Louisville/Jefferson County, KY-IN		1	1	1	1	0	1	12	11	11	11		44
Memphis, TN-MS-AR		1	1	1	1	0	0	10	10	9	8		
Birmingham-Hoover, AL		0	0	0	0	0	0						
Columbus, OH		0	0	0	0	0	0						
Raleigh, NC		0	0	0	0	0	0						
Seattle-Tacoma-Bellevue, WA		0	0	0	0	0	0						
Support Color Scheme:	strong	> 3 monitors						Annual avg NO₂ ≥ 20 ppb				DM1H NO₂ ≥ 60 ppb	
	moderate	at least 3 monitors						Annual avg NO₂ ≥ 15 ppb				DM1H NO₂ ≥ 50 ppb	

1 **Table 2-3. Preliminary near-road and area-wide CBSA data analysis using all available**
2 **and concurrent NO₂ ambient monitor measurement data (2011-2014).**

CBSA Name/Selection Indicator (Strong candidate) (possible candidate)	Area wide monitors with concurrent measurements (n)	Near-road to area-wide 98th pct concentration range	Monitor years (n)	Simultaneous Hours/year (n)	Correlation of near-road and area-wide >90th pct concentrations	Low-concentration monitor?
Houston-The Woodlands-Sugar Land, TX	16	0 to 33 ppb	1	5500	small – moderate	likely
Los Angeles-Long Beach-Anaheim, CA	16	-17 to 28 ppb	1	4000	small – moderate	uncertain
Dallas-Fort Worth-Arlington, TX	11	-9 to 21 ppb	1	4000	moderate	likely
San Francisco-Oakland-Hayward, CA	11	1 to 38 ppb	1	5500-7500	small – moderate	likely
New York-Newark-Jersey City, NY-NJ-PA	9		1	2000		uncertain
Boston-Cambridge-Newton, MA-NH	7	-6 to 27 ppb	2	4000-6500	small – strong	yes
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	7	-4 to 22 ppb	1	4500-6000	small – moderate	likely
Phoenix-Mesa-Scottsdale, AZ	5	-4 to 22 ppb	1	5000-7500	small – moderate	likely
Pittsburgh, PA	4	1 to 26 ppb	1	1000-2500	small – moderate	likely
St. Louis, MO-IL	4	2 to 25 ppb	2	6000-8500	moderate – strong	yes
Atlanta-Sandy Springs-Roswell, GA	3	7 to 39 ppb	1	3000	small	likely
Denver-Aurora-Lakewood, CO	3	-12 to 6 ppb	2	3000-7000	small – moderate	no
Kansas City, MO-KS	3	-7 to 33 ppb	2	3000-6000	small – moderate	no
Minneapolis-St. Paul-Bloomington, MN-WI	3	-6 to 21 ppb	2	6000	small – moderate	no
Providence-Warwick, RI-MA	3	14 to 43 ppb	1	5000	small – moderate	likely
Baltimore-Columbia-Towson, MD	2	2 to 12 ppb	1	4000	small	no
Cincinnati, OH-KY-IN	2	14 to 29 ppb	1	6500-8000	small	likely
Detroit-Warren-Dearborn, MI	2	0 to 16 ppb	4	2000-8500	small – strong	likely
Indianapolis-Carmel-Anderson, IN	2	11 to 13 ppb	1	4000-6000	moderate	no
Richmond, VA	2	4 to 13 ppb	2	2000-5000	small – moderate	no
San Antonio-New Braunfels, TX	2	10 to 19 ppb	1	6500	small – moderate	uncertain
Tampa-St. Petersburg-Clearwater, FL	2	17 to 21 ppb	1	4000	moderate	likely
Austin-Round Rock, TX	1	24 ppb	1	4000	moderate	uncertain
Birmingham-Hoover, AL	1	12 ppb	1	7000	small	no
Buffalo-Cheektowaga-Niagara Falls, NY	1	0 ppb	1	5500	moderate	no
Charlotte-Concord-Gastonia, NC-SC	1	-2 ppb	1	4000	moderate	no
Cleveland-Elyria, OH	1	2 ppb	1	2000	moderate	no
Columbus, OH	1	-4 ppb	1	7000	moderate	no
Des Moines-West Des Moines, IA	1	-3 to 0 ppb	2	8000	moderate	no
Hartford-West Hartford-East Hartford, CT	1	6 to 15 ppb	2	5000-6000	small – moderate	no
Jacksonville, FL	1	8 ppb	1	4000	small	no
Louisville/Jefferson County, KY-IN	1	19 ppb	1	500	small	uncertain
Memphis, TN-MS-AR	1	14 ppb	1	1000	small	uncertain
Milwaukee-Waukesha-West Allis, WI	1	14 ppb	1	5000	moderate	uncertain
New Orleans-Metairie, LA	1	6 ppb	1	6500	moderate	no
Portland-Vancouver-Hillsboro, OR-WA	1	5 ppb	1	3000	moderate	no
Raleigh, NC	1	-1 ppb	1	3500	moderate	no
San Jose-Sunnyvale-Santa Clara, CA	1	1 ppb	1	4000	moderate	no
Seattle-Tacoma-Bellevue, WA	1	21 ppb	1	4000	moderate	uncertain
Boise City, ID	No Area-wide Monitor Data Available At Time of Analysis					
Miami-Fort Lauderdale-West Palm Beach, FL	No Near-Road Monitor Data Available At Time of Analysis					
Nashville-Davidson–Murfreesboro–Franklin, TN						
Oklahoma City, OK						
Orlando-Kissimmee-Sanford, FL						
Riverside-San Bernardino-Ontario, CA						
San Diego-Carlsbad, CA						

Support Color Scheme:	strong	> 3 monitors	Max range ≥ +20 ppb	>1 year	≥6000 hrs	Strong: r > 0.66	yes
	moderate	3 monitors	Max range +10-20 ppb	na	≥4000 hrs	Moderate: r =0.33 – 0.66 Small: r <0.33	likely

Table 2-4. Number of NO₂ monitors and 2010-2013 design values for CBSAs that are not expected to have ambient concentration data available from newly sited near-road monitors.

| CBSA Name/Selection Indicator		Number of monitors per year						Annual Average NO$_2$ (ppb)				DM1H 98th pct 3-year avg NO$_2$ (ppb)	
(Strong candidate)	(possible candidate)	2010	2011	2012	2013	2010-2012	2011-2013	2010 max	2011 max	2012 max	2013 max	2010-12 max	2011-13 max
Santa Maria-Santa Barbara, CA		11	11	11	11	11	10	9	9	10	10	43	36
Baton Rouge, LA		8	8	8	8	7	7	13	12	11	10	54	52
Sacramento–Roseville–Arden-Arcade, CA		8	8	8	6	7	4	12	13	12	10	51	50
Chicago-Naperville-Elgin, IL-IN-WI		6	5	6	5	2	0	25	23	22	21	62	
Washington-Arlington-Alexandria, DC-VA-MD-WV		6	6	6	7	5	4	18	16	17	13	55	51
Farmington, NM		4	3	3	3	2	3	12	13	13	12	38	41
Beaumont-Port Arthur, TX		4	4	4	4	4	4	8	7	6	6	37	35
El Paso, TX		3	2	2	3	1	1	17	17	16	14	61	59
Bakersfield, CA		3	3	3	4	3	2	14	15	15	14	58	46
Springfield, MA		3	3	3	3	3	3	15	16	14	14	47	46
San Luis Obispo-Paso Robles-Arroyo Grande, CA		3	3	3	3	3	3	6	6	7	7	38	38
Gillette, WY		3	3	4	4	1	1	7	6	8	9	32	32
Durango, CO		3	3	3	2	0	0	6	7	6	6		
Wichita, KS		2	2	2	2	1	1	8	12	10	9	64	65
El Centro, CA		2	2	2	2	1	1	14	14	14	13	62	64
Las Vegas-Henderson-Paradise, NV		2	2	2	3	2	0	13	13	14	14	54	
Stockton-Lodi, CA		2	2	2	2	1	1	14	15	14	16	51	53
Ogden-Clearfield, UT		2	2	2	2	1	2	17	16	16	17	49	55
Tucson, AZ		2	2	2	2	1	1	12	12	13	11	47	46
Oxnard-Thousand Oaks-Ventura, CA		2	2	2	2	2	2	10	9	10	9	38	37
Bismarck, ND		2	2	2	2	2	2	6	5	5	5	36	35
Sioux City, IA-NE-SD		2	2	2	2	1	1	3	3	6	4	30	37
Vernal, UT		2	3	4	2	2	1	5	5	7	11	30	34
Harrisburg-Carlisle, PA		2	2	1	1	1	1	11	12	3	3	22	21
Urban Honolulu, HI		2	1	1	1	1	1	3	3	3	3	22	21
Fresno, CA		2	3	4	4	0	0	13	13	13	14		
Las Cruces, NM		2	2	2	2	0	1	9	8	7	7		41
Riverton, WY		2	3	3	2	0	1	1	1	2	1		5
Provo-Orem, UT		1	1	1	1	1	1	15	17	17	19	58	66
New Haven-Milford, CT		1	1	1	1	1	1	14	15	13	14	57	55
Reno, NV		1	1	1	1	1	1	16	17	14	16	55	56
Visalia-Porterville, CA		1	1	1	1	1	1	13	12	12	13	53	52
Worcester, MA-CT		1	1	1	1	1	1	14	17	13	12	52	52
Albuquerque, NM		1	1	1	1	1	1	12	13	14	12	51	48
Little Rock-North Little Rock-Conway, AR		1	1	1	1	1	1	10	10	11	10	51	50
Lexington-Fayette, KY		1	1	1	1	1	1	9	8	8	7	49	45
York-Hanover, PA		1	1	1	1	1	1	13	13	12	10	49	44
Kingsport-Bristol-Bristol, TN-VA		1	1	1	1	1	1	10	10	11	11	46	52
Yuba City, CA		1	1	1	1	1	1	8	8	10	10	45	47
Portland-South Portland, ME		1	1	1	1	1	1	10	9	10	8	43	44
Support Color Scheme:	strong	> 3 monitors						Annual avg NO$_2$ ≥ 20 ppb				DM1H NO$_2$ ≥ 60 ppb	
	moderate	At least 3 monitors						Annual avg NO$_2$ ≥ 15 ppb				DM1H NO$_2$ ≥ 50 ppb	

1 **Table 2-5. Population estimates for CBSA having at least 800,000 residents in 2013,**
2 **ordered by descending population.**

CBSA Name (strong candidate) or (possible candidate) based on Tables 2-2 through 2-4.	2013 Population	Near Road Monitor?
New York-Newark-Jersey City, NY-NJ-PA	19,949,502	Yes
Los Angeles-Long Beach-Anaheim, CA	13,131,431	Yes
Chicago-Naperville-Elgin, IL-IN-WI	9,537,289	No
Dallas-Fort Worth-Arlington, TX	6,810,913	Yes
Houston-The Woodlands-Sugar Land, TX	6,313,158	Yes
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	6,034,678	Yes
Washington-Arlington-Alexandria, DC-VA-MD-WV	5,949,859	No
Miami-Fort Lauderdale-West Palm Beach, FL	5,828,191	Yes
Atlanta-Sandy Springs-Roswell, GA	5,522,942	Yes
Boston-Cambridge-Newton, MA-NH	4,684,299	Yes
San Francisco-Oakland-Hayward, CA	4,516,276	Yes
Phoenix-Mesa-Scottsdale, AZ	4,398,762	Yes
Riverside-San Bernardino-Ontario, CA	4,380,878	Yes
Detroit-Warren-Dearborn, MI	4,294,983	Yes
Seattle-Tacoma-Bellevue, WA	3,610,105	Yes
Minneapolis-St. Paul-Bloomington, MN-WI	3,459,146	Yes
San Diego-Carlsbad, CA	3,211,252	Yes
Tampa-St. Petersburg-Clearwater, FL	2,870,569	Yes
St. Louis, MO-IL	2,801,056	Yes
Baltimore-Columbia-Towson, MD	2,770,738	Yes
Denver-Aurora-Lakewood, CO	2,697,476	Yes
Pittsburgh, PA	2,360,867	Yes
Charlotte-Concord-Gastonia, NC-SC	2,335,358	Yes
Portland-Vancouver-Hillsboro, OR-WA	2,314,554	Yes
San Antonio-New Braunfels, TX	2,277,550	Yes
Orlando-Kissimmee-Sanford, FL	2,267,846	Yes
Sacramento–Roseville–Arden-Arcade, CA	2,215,770	No
Cincinnati, OH-KY-IN	2,137,406	Yes
Cleveland-Elyria, OH	2,064,725	Yes
Kansas City, MO-KS	2,054,473	Yes
Las Vegas-Henderson-Paradise, NV	2,027,868	No
Columbus, OH	1,967,066	Yes
Indianapolis-Carmel-Anderson, IN	1,953,961	Yes
San Jose-Sunnyvale-Santa Clara, CA	1,919,641	Yes
Austin-Round Rock, TX	1,883,051	Yes
Nashville-Davidson–Murfreesboro–Franklin, TN	1,757,912	Yes
Virginia Beach-Norfolk-Newport News, VA-NC	1,707,369	No
Providence-Warwick, RI-MA	1,604,291	Yes

CBSA Name (strong candidate) or (possible candidate) based on Tables 2-2 through 2-4.	2013 Population	Near Road Monitor?
Milwaukee-Waukesha-West Allis, WI	1,569,659	Yes
Jacksonville, FL	1,394,624	Yes
Memphis, TN-MS-AR	1,341,746	Yes
Oklahoma City, OK	1,319,677	Yes
Louisville/Jefferson County, KY-IN	1,262,261	Yes
Richmond, VA	1,245,764	Yes
New Orleans-Metairie, LA	1,240,977	Yes
Hartford-West Hartford-East Hartford, CT	1,215,211	Yes
Raleigh, NC	1,214,516	Yes
Salt Lake City, UT	1,140,483	No
Birmingham-Hoover, AL	1,140,300	Yes
Buffalo-Cheektowaga-Niagara Falls, NY	1,134,115	Yes
Rochester, NY	1,083,278	No
Grand Rapids-Wyoming, MI	1,016,603	No
Tucson, AZ	996,554	No
Urban Honolulu, HI	983,429	No
Tulsa, OK	961,561	No
Fresno, CA	955,272	No
Bridgeport-Stamford-Norwalk, CT	939,904	No
Worcester, MA-CT	926,710	No
Albuquerque, NM	902,797	No
Omaha-Council Bluffs, NE-IA	895,151	No
Albany-Schenectady-Troy, NY	877,905	No
Bakersfield, CA	864,124	No
New Haven-Milford, CT	862,287	No
Knoxville, TN	852,715	No
Greenville-Anderson-Mauldin, SC	850,965	No
Oxnard-Thousand Oaks-Ventura, CA	839,620	No
El Paso, TX	831,036	No
Allentown-Bethlehem-Easton, PA-NJ	827,048	No
Baton Rouge, LA	820,159	No
McAllen-Edinburg-Mission, TX	815,996	No
Dayton, OH	802,489	No

1
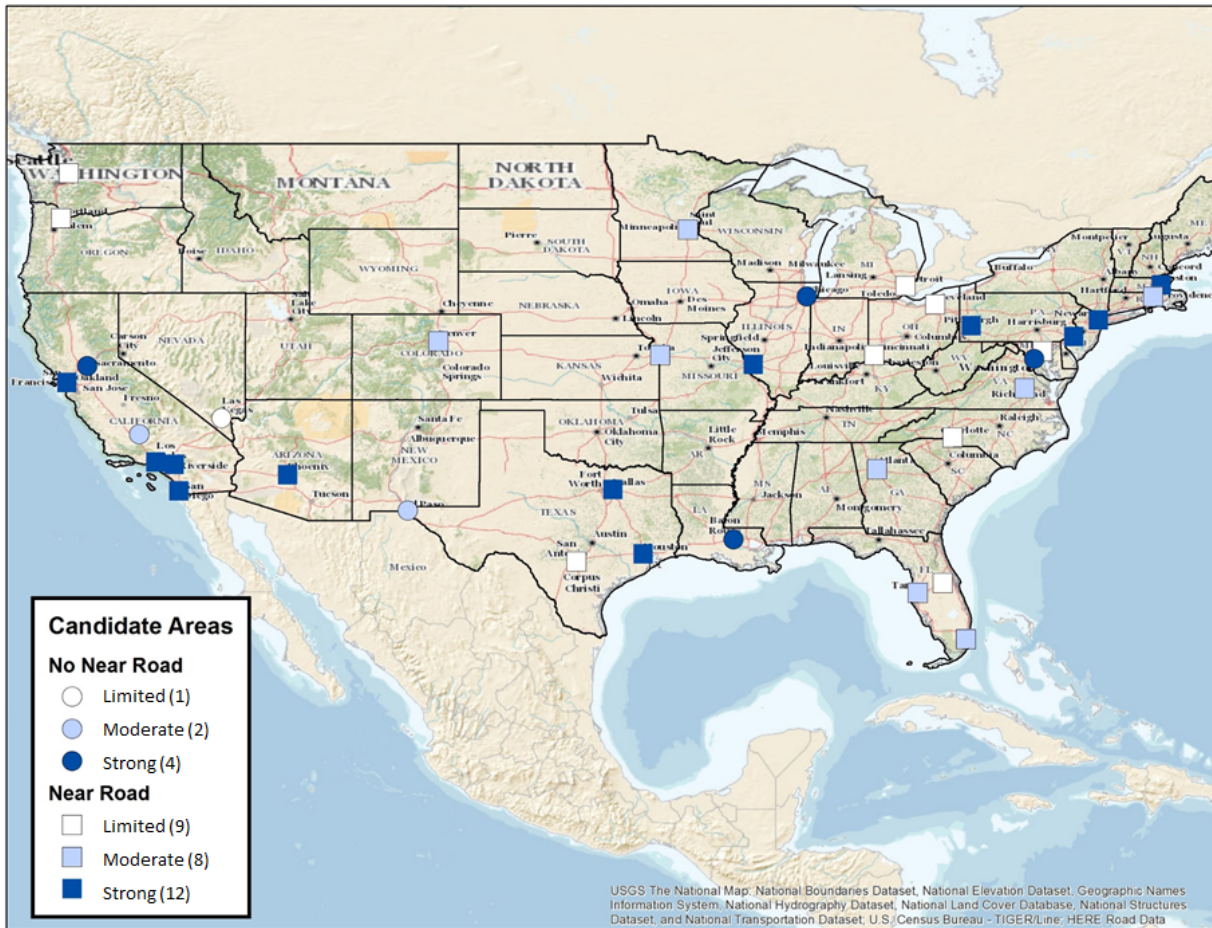
2

Figure 2-1. Locations of potential study areas to analyze in the air quality assessment categorized by availability of near-road monitor data and selection criteria ranking scheme.

2.3.1.3 Estimated ambient NO$_2$ concentrations

This section describes the approaches used to extend the information provided by the ambient monitor data alone, by developing additional air quality standard scenarios and addressing selected high-concentration environments that may not necessarily be captured by the existing monitoring network. The approach describing how air quality would be adjusted to just meet the existing and potential alternative standards (if any) is provided in subsection 2.3.1.3.1. This is followed by a discussion of how we plan on using the ambient monitors combined with factors to characterize NO$_2$ concentrations across urban study areas, including those occurring near-roads and on-roads (subsection 2.3.1.3.2). The final subsection (2.3.1.3.3) briefly discusses the approach to be used in characterizing uncertainties and limitations associated with these estimated concentrations.

1   *2.3.1.3.1 Adjusting air quality to just meet the existing standards*

2   Unadjusted air quality, termed "as is" in the prior review, represent ambient conditions as

3   they are at the time of measurement. While unadjusted air quality presents perspective regarding

4   existing conditions, it does not provide the specific effect that just meeting a particular standard

5   has on ambient concentrations, exposures, and health risk. To evaluate the ability of a specific air

6   quality standard to protect public health, ambient NO_2 concentrations need to be adjusted such

7   that they simulate levels of NO_2 that would just meet the existing standards (i.e., 100 ppb, 98th

8   percentile DM1H averaged across 3-years; 53 ppb, annual average) or potential alternative

9   standards. Such adjustments allow comparisons of the level of public health protection that could

10   be associated with just meeting the existing and potential alternative standards.

11   All areas of the United States currently have ambient NO_2 levels below the existing

12   standards, albeit to varying degrees.[38] Therefore, to simulate just meeting the existing standards,

13   NO_2 air quality levels in all study areas must be adjusted upward. There are a few ways this can

14   be performed, although any method must consider a few factors including representativeness,

15   applicability, and degree of complexity. The method used to adjust ambient concentrations to

16   just meet air quality standards in the 2008 REA was based on analysis of historical ambient air

17   quality occurring in six study areas (Rizzo, 2008), using concentrations that reflected actual

18   high-concentration conditions (i.e., generally supporting a representativeness criterion). Based on

19   the largely consistent patterns observed across study areas and relative simplicity (requiring only

20   the calculation of design values), a proportional approach was selected to adjust ambient

21   concentrations upwards in all study areas (i.e., supporting applicability), assuming that the

22   overall concentration distribution that exists with recent ambient conditions is identical to that

23   which would exist with higher concentrations that just meet the existing standard(s).

24   As mentioned in section 2.1, there were instances in the 2008 REA NO_2 air quality

25   characterization where the recent distribution of DM1H concentrations was not entirely

26   proportional to the historical air quality distribution. For example, Figure 2-2 shows the 0 to

27   100th percentile DM1H ambient concentrations measured at a single monitor (ID: 340273001) in

28   the New York CBSA, for years 1984 (high-concentration year) and 2007 (low-concentration

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[38] In general, only one of the two existing standards (1-hour or annual) would be the controlling standard in a particular area, and is identified by the monitor design value that is closest to the particular standard. Given the form and level of the existing 1-hour standard and overall degree of variability in hourly ambient concentrations, it is likely that the 1-hour standard is the controlling standard in most areas, where such a 3-year average value can be calculated. Preliminary analyses shown below indicate this is always the case for the study areas considered in the air quality assessment.

1   year) (Figure 2-2, left panel). Across this 28 year time frame, DM1H concentrations across the

2   entire distribution have decreased proportionally by about the same amount (about 40%) except

3   for the single DM1H concentration which has decreased by a greater amount (about 50%) over

4   that same time period. This observed non-linearity at upper percentile concentrations (i.e., the

5   rate of decrease of upper percentile concentrations over time was greater than that of other

6   percentiles of the distribution) was common for most of the monitors evaluated by Rizzo (2008)

7   from the six selected study areas. More specifically, for the years where the higher ambient

8   concentrations were measured, the upper percentile concentrations tended to be greater than what

9   would be expected when assuming an entirely proportional relationship with more recent low

10 ambient concentrations. There can be some variability in this relationship of course, as illustrated

11 by the most recent low-concentration year data available for the same NY monitor (Figure 2-2,

12 right panel), whereas a few of the upper percentile concentrations appear to deviate from

13 linearity in the opposite direction, though the maximum still expresses the greatest deviation

14 from linearity. Thus, when using a proportional approach to adjust recent ambient conditions

15 upwards to reflect a higher concentration air quality scenario, the estimated upper-most

16 percentile concentrations would tend to be less than those measured in an actual, and similarly

17 high, concentration year (i.e., the historical ambient concentrations).

18   Variable changes in selected percentiles of the concentration distribution occurring over

19 time could be driven by a number of influential factors. For example, particular emissions

20 reductions and control strategies that target sporadic NO_X emission release events could lead to

21 reduction in sporadic high-concentration events observed at ambient monitors. It is also possible

22 that NO_2-favorable atmospheric conditions and other important precursor emissions (e.g., NO_X)

23 were present at the time of the historical measurements and contributed to the occurrence of the

24 observed sporadic, high NO_2 concentration events. With these and other potential influential

25 factors in mind, this suggests that certain conditions existing at an historical time may be

26 reasonably assumed to exist within our hypothetical air quality scenario of just meeting the

27 existing standard(s), possibly requiring additional complexity to the approach beyond a simple

28 proportional adjustment.

29   The selection of the adjustment approach should also consider particular elements of the

30 NO_2 standard (i.e., form and averaging time) and the risk output generated for this. In reviewing

31 Table 2-2 and Table 2-4, currently the 1-hour standard is the controlling standard in all potential

32 study areas for the 3-years where that 1-hour standard can be calculated. In this air quality

33 assessment, we are interested in calculating the number of times 1-hour concentrations are at or

34 above health effect benchmarks, the lowest of which is 100 ppb. Therefore, the ambient

35 concentrations that are at or above the 98th percentile DM1H (i.e., 100 ppb at the highest/design

36 monitor) are the most important feature of the concentration distribution to characterize well at

1 any monitor when adjusting concentrations to just meet the existing 1-hour standard (more so at

2 the highest design value monitor, though also of relative importance at the other monitors in the

3 CBSA).
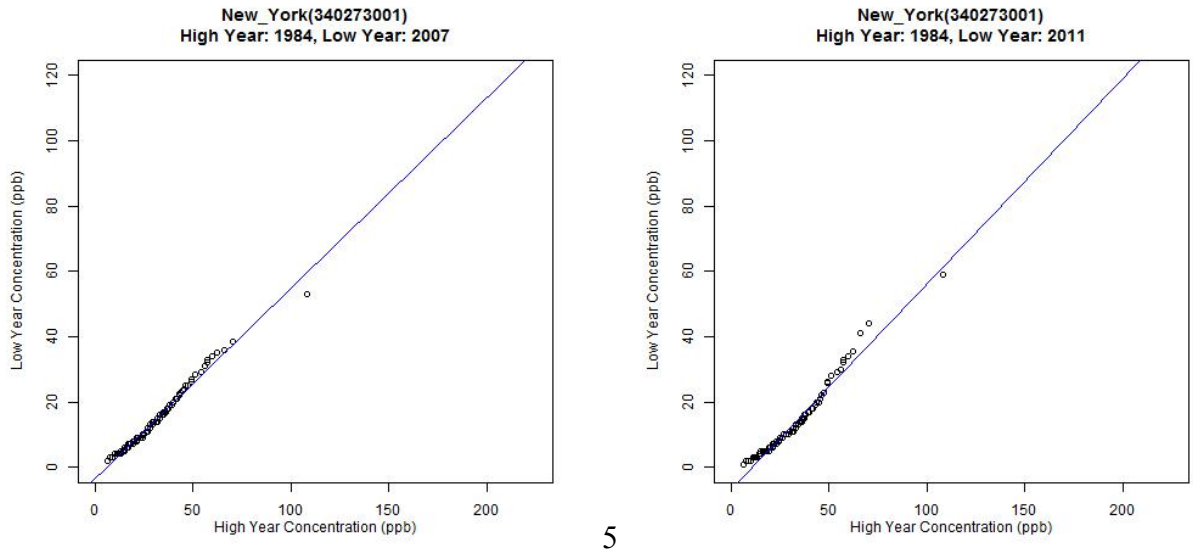
4   5

6 **Figure 2-2. Distribution of DM1H NO$_2$ concentrations (0 – 100th percentile) in the New**
7 **York CBSA for a high-concentration year (1984) versus a low-concentration year (2007)**
8 **adapted from Rizzo (2008) (left panel) and updated comparison with a recent low-**
9 **concentration year (2011) (right panel).**

10

11   With all of this considered, a two-step adjustment approach is proposed in this

12 assessment to adjust the recent ambient concentrations in a study area to just meet the existing

13 standards. For this two-step adjustment approach, a proportional approach is used, as was done

14 in the 2008 REA, though here only applied to concentrations up to and including the 98th

15 percentile DM1H (adjustment step 1). An additional modification to address the observed

16 deviations from linearity at the upper percentile concentrations is also proposed, particularly at

17 and above the 98th percentile (adjustment step 2). In this way, this two-step approach utilizes the

18 simplicity of the proportional approach used in the 2008 REA but addresses more fully, the

19 observed changes in peak concentration distributions to better capture the distribution of high

20 NO$_2$ concentrations when adjusting air quality to meet the existing standards. As noted above, it

21 is important to characterize well the adjusted concentrations that are at or above the 98th

22 percentile DM1H for the analyses we are proposing. The general details in the approach are as

23 follows and proposed to be applied to the selected study areas.

24   1. As was done in the 2008 REA for all selected study areas, compare historical high-
25   concentration year data (selected based on having concentrations close to or at
26   existing DM1H standard) with recent low-concentration year data within monitors

1   (for all possible monitors in a CBSA and considering similar years)[39], using the

2   distribution of DM1H concentrations.

3   a. Plot and evaluate for proportionality using visual inspection and linear

4   regression coefficient estimates and fit statistics, using both the full

5   distribution and those concentrations at or below the 98th percentile DM1H.

6   b. Regress all CBSA monitors against the CBSA design monitor DM1H also

7   across a several year period, noting variation (if any) in regression coefficients

8   and fit statistics.

9   c. Qualitatively discuss the potential impact to adjusted concentrations using the

10   above information and considering ambient monitor site attributes (i.e.,

11   section 2.3.1.2, distance from roads, the number/type/amount of emission

12   sources in close proximity to a monitor, concentration correlations).

13   2. Calculate design values for each monitor having recent (2010-2013) air quality,

14   considering both the 1-hour and annual standards, and identify the controlling

15   standard and design monitor.[40] Calculate the proportional adjustment factor needed to

16   just meet the existing standard by dividing the standard level by the design value.

17   3. Adjustment step 1: Adjust all DM1H concentrations proportionally, up to and

18   including the 98th percentile DM1H, at all monitors and for each single year in each

19   study area using the adjustment factor derived from the design monitor (and

20   considering the 3-year averaging period).

21   4. Adjustment step 2: For DM1H concentrations above the 98th percentile DM1H, rather

22   than use proportionally adjusted concentrations, calculate a set of ratios for each

23   monitor by dividing the DM1H concentrations that are above the 98th percentile

24   DM1H by the 98th percentile DM1H.[41] This set of ratios is proposed to be based on

25   the individual monitor and derived using a range of historical and recent ambient

26   monitor concentrations (e.g., 2000-2013) that meet completeness criteria. A mean or

27   median value across years is proposed as a reasonable approximation to use in

---

[39] An important limitation is that the monitors in operation can vary year-to-year, even more variable when considering the decades that exists between high-concentration and low-concentration years of interest, leading to fewer monitors available to evaluate in an area.

[40] See http://www.epa.gov/airtrends/values.html. To date, 2013 is most recent complete year for most monitor sites. The period 2010-2013 would include four annual design values and two 3-year averaged hourly design values.

[41] For a full year of ambient data, there would be 365 or 366 DM1H concentrations. Therefore, upwards to seven unique ratios could be calculated using these seven days having DM1H concentrations above the 98th percentile DM1H.

1   adjusting the upper percentile concentrations at each monitor. Where adjustment

2   factors cannot be calculated (i.e., the monitor is newly sited), adjustment factors from

3   the design monitor would be used.

4

5   One remaining issue in this adjustment approach regards that of the role of the near-road

6 data. To date, there are no near-road monitors that reported three years of complete data; most

7 CBSAs have only a single year of near-road monitor data, many of which are not considered as

8 having a complete year. Therefore, design values cannot necessarily be calculated using these

9 near-road monitors. We propose that, where simultaneous measurements are available at the

10 monitor having the highest design value for the year and hours that the near-road monitor active,

11 and where the near-road monitor is reporting greater concentrations than the monitor having the

12 highest design value, the proportional adjustment factor would then be calculated similar to what

13 is outlined above in step 2 (98th percentile DM1H) but would be based on the available near-road

14 monitor data. This proportional adjustment factor would then be applied to all years of air quality

15 evaluated in the particular CBSA. Otherwise, concentrations measured at the monitor having the

16 highest design value will be used to calculate the adjustment factor.

17 *2.3.1.3.2 Simulating air quality to represent on-road concentrations*

18   In the discussion that follows, we first briefly describe the approach used in the 2008

19 REA to simulate on-road NO$_2$ concentrations. This is followed by a consideration of the

20 information available in the current review that could further inform our understanding of factors

21 that contribute to variability in near-road and away-from-road NO$_2$ concentrations and their

22 relationships. This information includes the scientific evidence summarized in the 2nd draft ISA;

23 on-road, near-road and away-from-road measurement data and related analyses; and outputs

24 from recent and planned near-road modeling studies. Based on this available information, several

25 options are available in the current review to estimate on-road NO$_2$ concentrations. One options

26 would be to assume recent near-road ambient monitored NO$_2$ concentrations are a reasonable

27 approximation of on-road NO$_2$ concentrations, particularly for instances where monitors are sited

28 in close proximity to a major road (e.g., at or within 10 m or other proximal distance). In

29 addition, we could develop a set of on-road simulation factors using information from one or

30 more of the following sources: 1) ratios of on-road to away-from road concentrations, based on

31 available measurement data from research studies; 2) a statistical/fitted model using available

32 measurement-based near-road transect study data, or 3) air quality model-based on- and near-

33 road transect study concentrations.

34   As described earlier, the 2008 REA derived factors from exponential models that were

35 individually fitted to data obtained from eleven published studies having measured either on-road

1   (5 studies) or near-road (6 studies) NO_2 concentrations, along with having measured a number of
2   corresponding away-from-road NO_2 concentrations. All of the studies reported time-averaged
3   concentrations sampled over at least 1-2 week periods. Using the ratio of on-road and away-
4   from-road[42] concentrations estimated from the fitted exponential models, two empirical discrete
5   distributions of factors were generated and distinguished by one of either two seasons ("summer"
6   and "not summer") based on when the original study data were collected. To simulate hourly on-
7   road NO_2 concentrations in selected study areas analyzed in the 2008 REA air quality
8   characterization, the distributions of these factors were randomly sampled and applied to ambient
9   NO_2 concentrations measured at monitors sited ≥100 m from a major road and considering the
10   appropriate season. Using this approach, simulated on-road NO_2 concentrations in the 2008 REA
11   were, on-average, 80% higher than respective ambient levels at distances ≥100 m from a major
12   road (2008 REA, section 7.3.2).
13   The 2nd draft ISA (U.S. EPA, 2015) identifies a few studies that fit an
14   exponential/logarithmic function to the near-road and away-from-road NO_2 concentrations
15   (section 2.5.3.1), providing some support for using an exponential model in quantifying the
16   pattern of decreasing concentrations with increasing distance from a road, much like that used in
17   the 2008 REA (e.g., Cape et al., 2004; Gilbert et al., 2003). In summarizing the published
18   literature for where on-/near-road and away-from-road concentrations were measured (including
19   the same longer-term time-averaged study data used for the 2008 REA), the 2nd draft ISA states
20   "NO_2 concentrations measured from 0 to 20 m from the road range up to 20 ppb higher, or up to
21   100% higher than concentrations measured between 80 and 500 m from a road" (U.S. EPA,
22   2015). This value (up to 100%) is also generally consistent with the factors used to estimate on-
23   road concentrations in the 2008 REA. The 2nd draft ISA also evaluated studies having a more
24   intensive sampling regimen and, when restricted to measurements closest to the near-road
25   environments, data from these studies "suggest concentrations at 15 to 20 m [from a road]
26   average 20−40% higher than concentrations 80 m from the road." Further, the 2nd draft ISA
27   highlights greater differences in the near-road/away-from-road concentrations during daylight
28   hours, during summer hours, as well as noting the existence of an inverse relationship between
29   the difference and the concentration level.

---

[42] In the exponential model used in the 2008 REA, "background" concentrations are approximated and assumed to be not directly influenced by road emissions. These estimated concentrations served as the away-from-road concentrations in calculating the distribution of factors used to simulate on-road concentrations.

1   Absent from the 2nd draft ISA analysis however, is an assessment of the quantitative

2   relationship between on-road (i.e., at or within 1 m from a road) and immediately near-road (e.g.,

3   10 to 20 m away-from-road) measurement concentrations. For example, using data available

4   from three studies reporting concentrations measured ≤1 m from a road and at other immediately

5   near-road sample sites, the percent increase in on-road concentrations was calculated here (Table

6   2-6). The calculated percent increase was wide ranging, however, when considering urban roads

7   having the highest on-road concentrations, on average, the on-road concentrations could be about

8   15% to 35% higher than concentrations sited at about 10 to 20 meters away-from-roads. It is

9   worth noting that each of these three measurement studies reported concentrations that were

10   time-averaged over at least a one-week period.

11   We propose to include in this first proposed approach, data from any newly identified

12   research studies (e.g., the GMAP studies mentioned above) and additional studies identified in

13   the ISA to further characterize a quantitative relationship between on-road measurements and

14   concentrations measured immediately away-from-roads. In doing so, we will be cognizant of the

15   study time-averaging in summarizing the data, noting those having shorter-term average (e.g.,

16   hourly) and on-road measurement in particular, where data are available.

17   **Table 2-6. Percent increase of on-road compared to near-road using NO_2 concentrations in**
18   **three studies having both on-road and immediately near-road measurements.**

| Study Author	Road Description/ Season	On-Road/Near-Road Site Distance (m)	Mean On-Road/Near-Road Concentration (ppb)	Percent increase of on-road to nearest-road			Comments
				Mean	Median	Range (min to max)	
Bell and Ashenden (1997)	Rural/Summer	<1/20	16/9	88%	90%	27% to 183%	Summer data: May-September
	Rural/ Not Summer	<1/20	15/11	48%	34%	-12% to 143%	
Cape et al. (2004)	Two-way Trunk/Annual	1/10	19/15	35%	26%	21% to 65%	Site distances are variable based on actual road shoulder width
	Single-way Trunk/Annual	1/10	7/5	44%	46%	31% to 51%	
	Non-Trunk/ Annual	1/10	7/6	15%	21%	-7% to 30%	
Monn et al. (1997)	Urban/ Unknown	0/20	43/38	15%	19%	-3% to 24%	

19

20   A second proposed approach involves the development of a statistical model similar to

21   that developed in the 2008 REA, though using newly available near-road monitoring data

22   collected in Las Vegas, NV. Details of the statistical model development and on-road simulation

23   factor output is fully described in Appendix A, while information regarding the measurement

24   data collection are found in Kimbrough et al. (2013). Briefly, near-road measurements of air

25   quality, traffic, and meteorology were collected at a study area located adjacent to Interstate-15

1 (I-15) in Las Vegas NV during Dec. 2008 to Jan. 2010. Downwind sampling sites were located

2 approximately 20 m, 100 m, and 300 m east of the interstate. These measured 1-hour average

3 NO_2 concentrations were fitted to statistical models to generate factors for use in estimating on-

4 road NO_2 concentrations from concentrations measured at the newly sited near-road monitors.

5   Several statistical models were developed from this Las Vegas near-road monitoring data

6 set and evaluated to describe the pattern in the concentration decrease observed with increasing

7 distance from a major road. Based on model fits (R^2) and overall form, a logit-ln function was

8 determined as most appropriate, though from a practical perspective, the model fit is similar to

9 an exponential decay (e.g., see Figure 2-3). Considered also was the influence of local

10 meteorological conditions (e.g., wind direction and approximate mixing heights). The logit-ln

11 functions were then used to estimate on-road NO_2 concentrations and concentrations predicted at

12 varying distances from the road (i.e., 10, 20, 40, 50, 100, and 300 meters).

13   Using each hourly prediction, statistically modeled on-road NO_2 concentrations were

14 compared to the modeled near-road (e.g., 10 m) concentrations to calculate the percent increase

15 in on-road concentrations. The distributions of percent increases were then stratified by the near-

16 road concentration distribution quintiles and averaged for each; included also was the overall

17 average percent increase that was calculated using the entire distribution for each statistical

18 model developed. Given that for many hours of the day, meteorological data were available that

19 corresponded to concentration measurements, we evaluated five model scenarios: 1) all wind and

20 atmospheric stability conditions combined, 2) winds from the west (210°-330°, where the

21 monitors were downwind of the highway), 3) winds from the east (30°-150°, where the monitors

22 were upwind of the highway), 4) inversion conditions (convective mixing height less than 300

23 m), and 5) non-inversion conditions (convective mixing height greater than 300 m).

24
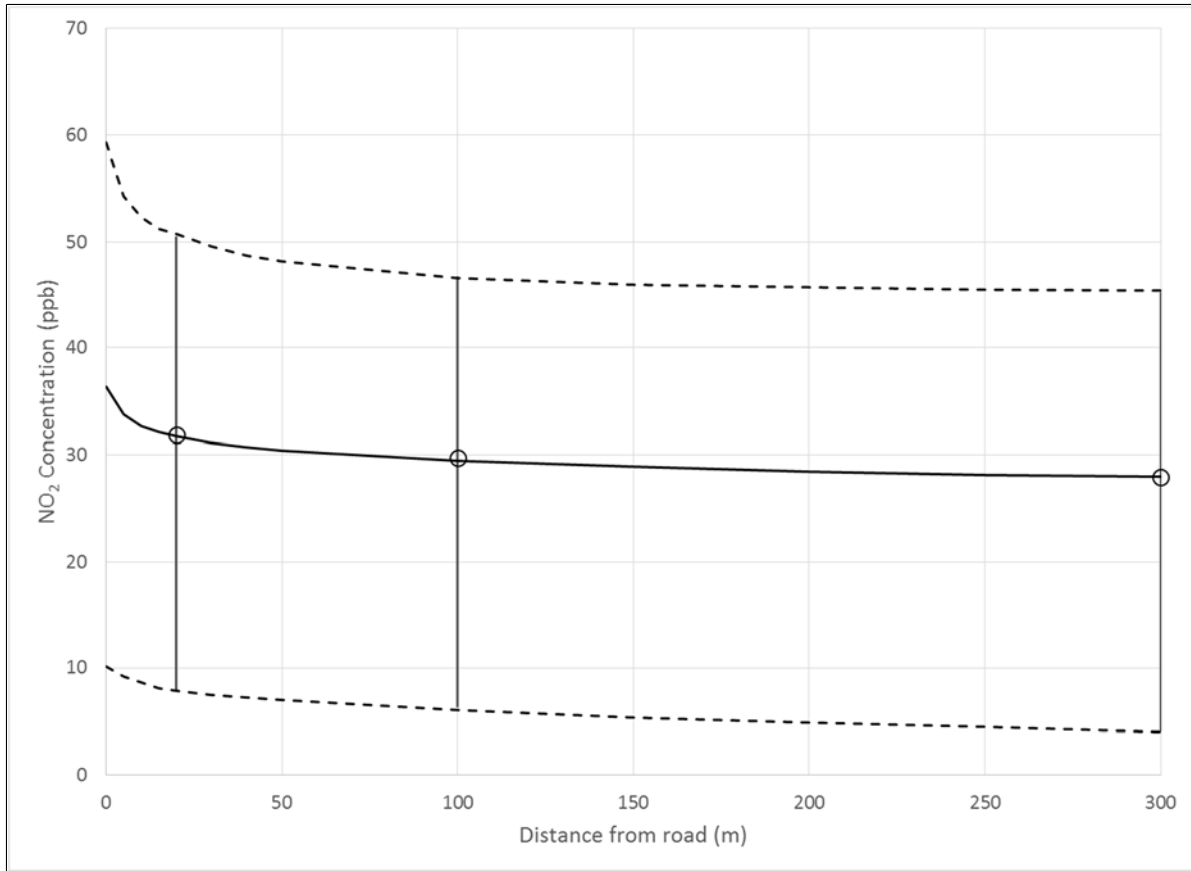
Figure 2-3. Predicted and observed NO$_2$ concentrations for winds from the west using based on data from a Las Vegas NV near-road measurement study. Predicted median (solid), predicted 98th and 2nd percentile (dotted), observed median (circles), and observed 98th and 2nd percentiles (error bars) are shown.

For this REA plan, we generated a set of factors to use in simulating on-road NO$_2$ concentrations, designed specifically for the measured concentrations at the new near-road monitors sited at or around 10 m and 20 m from a road.[43] These factors are provided in Table 2-7 and are stratified by the away-from-road concentration distribution quintiles and considering varying meteorological conditions. To a limited extent (and as discussed in the 2nd draft ISA), concentration level affects the value of the adjustment factor; in general, lower concentration quintiles have greater percent differences between on-road and away-from-road concentrations

---

[43] It is possible to generate on-road simulation factors to use for any monitor distance (e.g., 80 m, 200 m from the road), though the principal objective here was to use the new near-road monitor data.

1   than higher concentration quintiles. Meteorological conditions also affect the value of the

2   adjustment factor; conditions where winds were predominantly from the west (downwind) or

3   having a greater mixing height have greater percent difference between on-road and away-from-

4   road concentrations than when winds were from the east or atmospheric inversions were present.

5   In considering these factors and their influential variables, and seeking a generally conservative

6   though simple approach using these results to simulate on-road NO_2 concentrations, we propose

7   an increase in NO_2 concentrations by 15% and 20% could be applied where we have near-road

8   concentrations at monitors sited within 10 and 20 m from a major road, respectively. These

9   selected values are at the upper range of values provided in Table 2-7, particularly when

10   considering the highest concentrations values and associated high-concentration meteorological

11   conditions (and hence when benchmark exceedances would be expected to occur).

12   **Table 2-7. Potential factors that could be used to simulate on-road NO_2 concentrations**
13   **from near-road monitors sited at 10 or 20 meters from a major road, stratified by**
14   **concentration quintiles and meteorological conditions, based on analysis of Las Vegas, NV**
15   **near-road measurement study data (see Appendix A).**

Near-road distance	Concentration distribution quintile	Average upwards adjustment from near-road concentrations given meteorological condition				
		All	Westerly winds	Easterly winds	Atmospheric inversion	Non inversion
10 meters	1	13%	25%	10%	10%	7%
	2	19%	15%	12%	14%	20%
	3	14%	8%	10%	13%	22%
	4	10%	8%	9%	12%	21%
	5	11%	10%	9%	12%	15%
	overall	13%	13%	10%	12%	17%
20 meters	1	19%	34%	15%	13%	10%
	2	25%	22%	17%	19%	27%
	3	17%	11%	11%	17%	29%
	4	11%	10%	12%	15%	26%
	5	12%	11%	11%	14%	17%
	overall	17%	17%	13%	16%	22%

16

17   For comparison, these statistical model developed factors provided in Table 2-7 are

18   higher than those that could be approximated from a meta-analysis performed by Karner et al.

19   (2010). By using a similar collection of on- and near-road study data used by the 2008 REA,

20   Karner et al. (2010) applied a locally-weighted regression approach (LOESS) to roadway-edge

21   normalized concentrations and approximated the overall pattern in decreasing concentrations

22   with increasing distance from a road. Based on Figure 3 of that study (see the 2nd draft ISA

23   Figure 3-2) and approximating both the on-road concentrations and concentrations at a distance

1   of 10 m, 20 m, and 100 m from a road, we estimate the factors would be about 6%, 11%, and

2   33%, respectively. Note that many of the on-road NO_2 concentrations used in the Karner et al.

3   (2010) study were modeled using an exponential equation and, where measured concentrations

4   were available, were from longer-term sampling durations (i.e., time-averaged over at least a

5   week or more).

6   And finally, newly available data from air quality modeling analyses could also be used

7   to inform the development of an approach or factors to use in simulating on-road concentrations

8   from either the near-road monitor data, or from monitors sited at further distances from a road.

9   As an example, Figure 2-4 illustrates the modeled NO_2 concentration decline with distance from

10   a road in Ft. Lauderdale FL (Thurman, 2013). Hourly concentrations modeled at on-road

11   receptors could be used in a manner similar to that done above using the statistical model output,

12   and also using any away-from-road receptor distance of interest. For instance, maximum NO_2

13   concentration are about 325 and 295 ug/m^3 for the southbound and northbound traffic lanes,

14   respectively, modeled at a distance of about 5 m from the source emission release (Figure 2-4).

15   Assuming this receptor is representative of on-road concentrations, these on-road concentrations

16   represent an increase of approximately 35% that of the corresponding concentrations estimated at

17   the cross-road distance of 25 m, and considering either traffic direction.
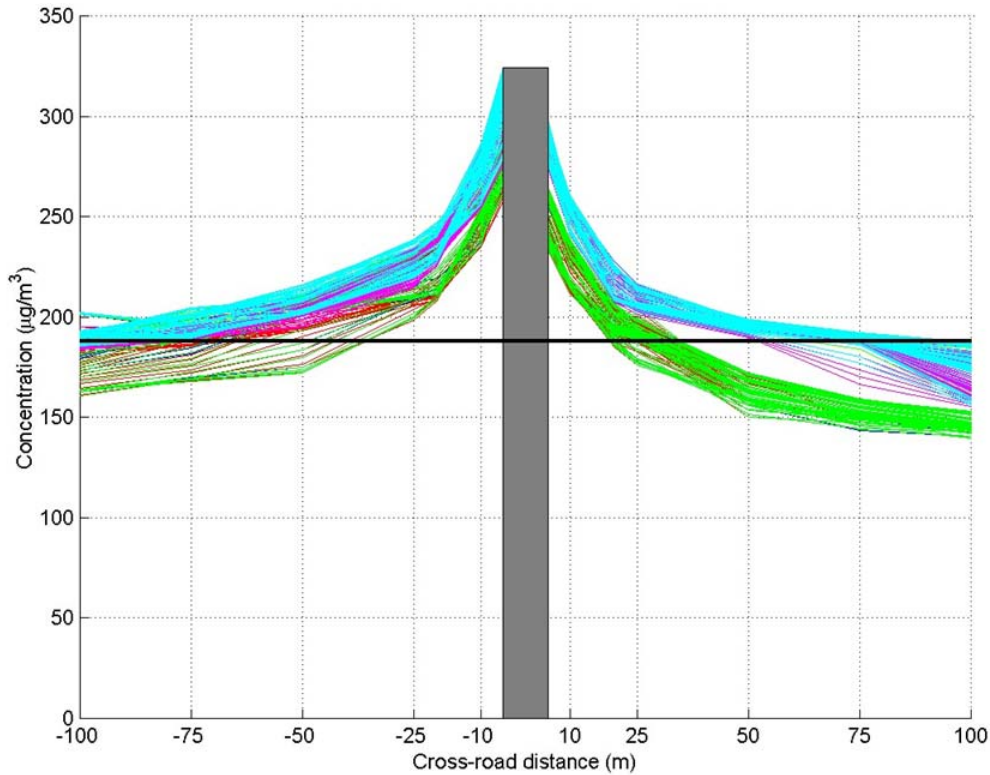
18

19   **Figure 2-4. AERMOD modeled maximum 1-hour NO_2 concentrations with increasing**
20   **distance from a major road in Ft. Lauderdale, FL.**

1   In summary, the newly deployed near-road monitors will better-characterize NO_2
2   concentrations occurring around roadways, compared to the monitoring information available in
3   the last review. Based on this newly available information, as well as information from the types
4   of monitoring and modeling analyses described above, staff will consider the extent to which it is
5   appropriate to apply on-road simulation factors to ambient NO_2 concentrations at sites of near-
6   road monitors. The appropriateness of applying such a factor to estimate on-road concentrations
7   in a given location will depend on the characteristics of the near-road site under evaluation,
8   including the proximity of the monitor to the road and the degree to which the near-road monitor
9   could also reflect NO_X emissions from nearby stationary sources.

10   To the extent it is judged appropriate to apply an on-road simulation factor, staff will
11   consider several potential approaches to doing so. As discussed above, studies indicate higher
12   on-road or curb-side NO_2 concentrations compared with concentrations measured at short
13   distances from the road along a transect (e.g., Bell et al., 1997; Cape et al., 2004; Monn et al.,
14   1997). Statistical models developed using near-road measurement data also indicate a pattern of
15   increasing concentrations in close proximity to the roadway, with maximum NO_2 concentrations
16   estimated to occur on-roads. The concentration change with distance to a road has been
17   described using an exponential/logarithmic decay equation (Cape et al., 2004; Gilbert et al.,
18   2003; U.S. EPA., 2008b; U.S. EPA, 2015) and more recently using LOESS smoothing (Karner et
19   al., 2010) and a logit-ln function (Appendix A). A recent emissions/dispersion modeling analysis
20   also indicates a similar pattern in decreasing concentration with increasing distance from road
21   (Thurman, 2013). Based on all of these analyses, on-road NO_2 concentrations could be simulated
22   by increasing NO_2 concentrations measured at near-road monitoring sites (i.e., within 10-20 m
23   from a major road) by about 6% to 35%, depending on near-road road distance, the approach
24   selected (i.e., factors-based using on-road/near-road measurement concentrations, statistical
25   model-based largely using near-road measurement data, or air quality model-based using
26   emissions and meteorological data), and the characterization of other important emissions
27   sources in the vicinity of near-road monitors that could influence measured concentrations.

28   *2.3.1.3.3 Characterization of uncertainty in estimated concentrations*

29   As was done in all of our recent REAs (U.S. EPA, 2008; U.S. EPA, 2009; U.S. EPA, 2010;
30   U.S. EPA, 2014b) a systematic approach adapted from WHO (2008) will be used here to succinctly
31   characterize uncertainties for each particular component comprising the assessment. First, staff
32   will identify, incorporate, and qualitatively describe any observed variability in input data sets,
33   influential attributes, overall composition of the knowledge-base, and estimated parameters within
34   the analyses performed to re-characterize (if needed) previously identified uncertainties in the 2008
35   REA (see Table 7-31 in U.S. EPA, 2008) and to identify additional uncertainties that were not

1   previously evaluated. In addition, and where possible, sensitivities of important variables anticipated

2   to significantly influence estimated concentrations (e.g., air quality adjusted to just meet the existing

3   standards, simulated on-road concentrations) will be evaluated to characterize the potential

4   magnitude and direction of influence the identified uncertainties may have on the estimated

5   concentrations.[44]

6 2.3.1.4 Calculating benchmark exceedances

7   As was discussed above in section 2.3.1.1, we have identified 1-hour concentrations

8   ranging from 100 to 400 ppb (in 100 ppb increments) as the benchmark levels to consider in the

9   air quality assessment. The complete set of DM1H concentrations will be used to calculate the

10   number of days per year the benchmark levels are exceeded at each monitor, for each year of air

11   quality, and within each study area. Each of the air quality scenarios considered in this

12   assessment (as is, existing standard, potential alternative standards (if any)) and adjusted high-

13   concentration environments (e.g., on-road) will be summarized using descriptive statistics such

14   as the mean and maximum number of exceedances per year for each individual CBSA.

15   Staff performed an analysis of the historical air quality to provide context for this

16   benchmark analysis, focusing here on instances where concentrations have been at or just around

17   the level of the existing 1-hour standard. It is worth noting that the historical data are

18   representative of real air quality scenarios that existed at the time of monitoring and that changes

19   in emissions control and atmospheric conditions that have occurred since that time would

20   preclude us from drawing complete conclusions about the number of exceedances associated

21   with a given 98th percentile DM1H concentration if attempting to use such information as a

22   prediction for future air quality. Nevertheless, using these unadjusted data are considered

23   informative given general consistencies in the overall concentration distribution over time at

24   each monitor and what would be expected regarding the number of exceedances given the form

25   of the existing 1-hour standard (i.e., for a complete year of data, there would be about 8 days

26   having concentrations at or above the 98th percentile DM1H value of 100 ppb).

27   We first calculated all rolling 3-year average 98th percentile DM1H values for all

28   individual monitors in operation from 1980-2014 that met the completeness criteria described

29   above (section 2.3.1.2). Staff then counted the number of days having exceedances of the 1-hour

30   benchmark levels (i.e., 100, 200, 300, and 400 ppb, if any) for each individual year within that 3-

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[44] A qualitative characterization of low, moderate, and high is assigned to the magnitude of influence and knowledge-base uncertainty descriptors, using quantitative observations relating to understanding the uncertainty, where possible.

1   year period and identified the maximum number of exceedances (thus, the highest observed

2   number of exceedances at that monitor for a single year) and calculated the mean number of

3   exceedances (thus, the average of the observed number of exceedances at that monitor across the

4   3-year averaging period) given the 3-year average 98th percentile DM1H for that monitor.

5   Results of this analysis are presented in Figure 2-5.

6   Based on the analysis of all available historical air quality and considering the form and

7   level of the existing 1-hour standard, the maximum number of days in a single year that the

8   DM1H was ≥ 100 ppb ranged from about 10 to 20 days (Figure 2-5, top left panel), while on

9   average across a 3-year period, the number of days/year having similar benchmark exceedances

10   ranged from about 6 to 13 days (Figure 2-5, top right panel). This mean number of days per year,

11   on average, corresponds well with general expectations described above (i.e., on average there

12   could be about 8 days/year with exceedances given the form of the standard). Furthermore, and

13   according to the analysis of all available historical air quality, exceeding a 1-hour benchmark

14   level of 200 ppb is a rare occurrence when considering the form and level of the existing 1-hour

15   standard. For example, of the 23 times a monitor had a 3-year average 98th percentile DM1H of

16   100 ppb, there were no exceedances of the 200 ppb benchmark on 19 of these occasions (Figure

17   2-5, bottom left panel). When averaging across the 3-year period, the mean number of days per

18   year having a similar benchmark exceedance drops to 1 or less, again with most monitors

19   recording no exceedances of the 200 ppb 1-hour benchmark. It should be noted that monitors in

20   California CBSAs (Los Angeles, San Francisco, etc.) constitute the bulk of the data where 3-year

21   average 98th percentile DM1H concentrations were at or above 100 ppb, though the results of

22   this analysis when excluding these areas are similar (data not shown), albeit with a tighter range

23   of values than when including the monitoring data in California (e.g., the mean number of

24   days/year having DM1H ≥ 100 ppb ranged from about 6 to 9).

25
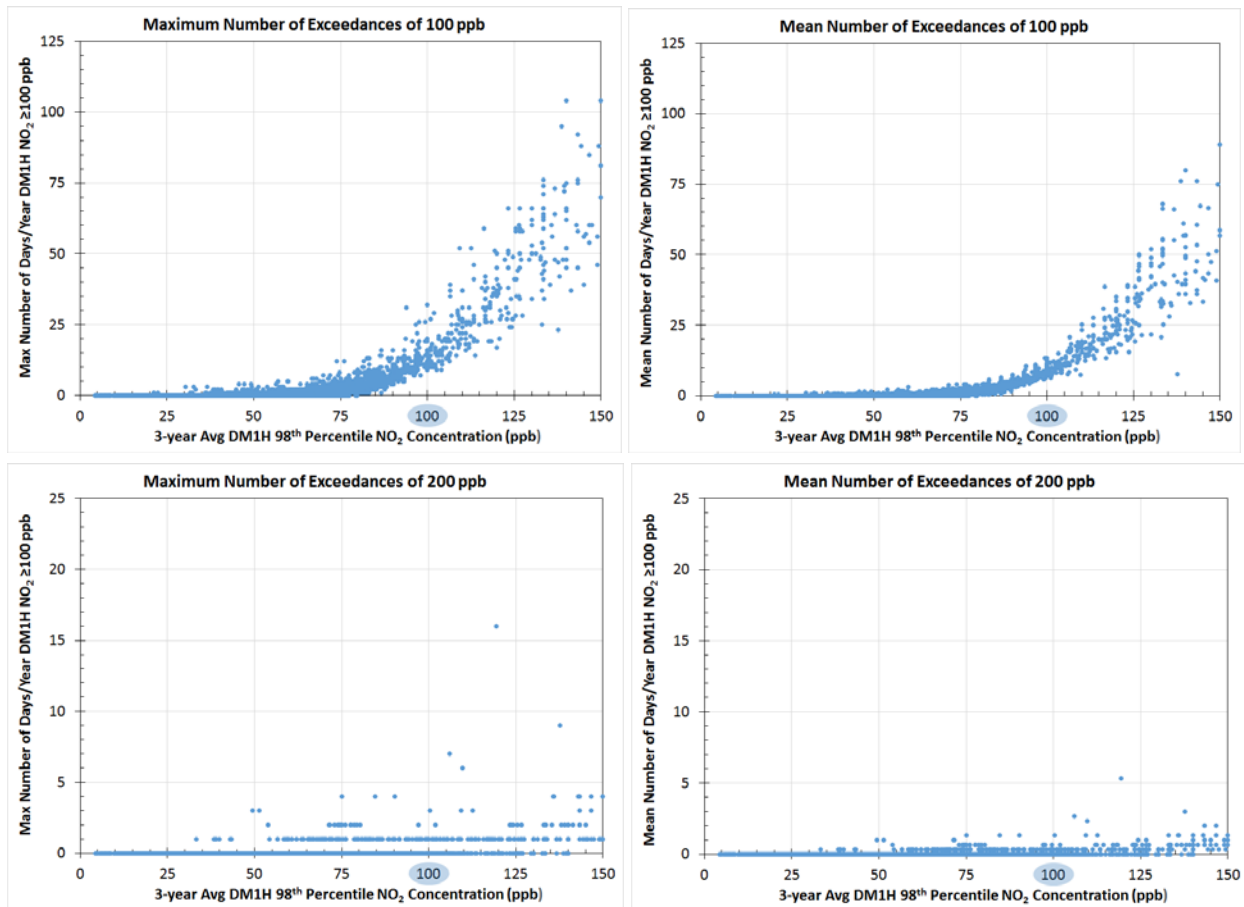
Figure 2-5. The maximum (left panel) and mean (right panel) number of days per year where DM1H NO₂ concentration was ≥ 100 ppb (top panel) and ≥ 200 ppb and associated with 3-year average 98th percentile DM1H NO₂ concentrations, using 1980-2014 ambient monitor data.

2.3.2 Illustrative Example: Characterizing Air Quality and Calculating Benchmark Exceedances in an Example Urban Study Area (Philadelphia)

This section presents results associated with use of the above proposed approaches to characterize air quality in an example study area, namely the Philadelphia CBSA. The Philadelphia CBSA was identified above as a strong candidate for selection in this assessment and was a study areas evaluated in the 2008 REA air quality characterization, thus providing comparative benchmark exceedance results for the two analysis periods. Further, there are a variety of monitoring locations included in the Philadelphia CBSA including industrial, residential (urban-core and suburban), and agricultural areas, as well as having a newly sited near-road monitor. In the first section describes the general attributes of the study area, focusing primarily the ambient monitoring attributes and NOₓ emissions information. Then, details are

1   provided regarding the adjustments made to simulate air quality that just meets the existing
2   standards and for estimating on-road concentrations from near-road concentrations. Finally,
3   exceedances of benchmark levels are calculated for two air quality scenarios (unadjusted ambient
4   concentrations and adjusted to just meeting the existing standard) and for the two distinct
5   concentration types (measured area-wide/near-road and simulated on-road) using the most
6   recently available monitoring data (2011-2014).

7   **2.3.2.1 Ambient monitor attributes**
8   Staff considered the complete set of all available monitors in the Philadelphia CBSA from
9   1980-2013 (Figure 2-6) to best characterize air quality when applying the analytical approaches
10 described above. Note that over this period of interest, monitors may either begin or cease
11 operation, and in any given year, monitoring data may not meet data completeness criteria. Thus
12 when considering the continuity of NO_2 measurements across the study area, on occasion, the
13 overall analytical data set may be fragmented. Regardless, the goal in this assessment for each
14 analytical approach is to take the necessary steps to ensure appropriate use of the data in an effort
15 to better understand their representativeness in describing the study area's air quality.
16   The siting of the ambient monitors is of particular importance, recognizing that the
17 purpose of the monitoring could have an influence on the measured NO_2 concentrations and
18 subsequent interpretation in the air quality characterization. Specific monitoring site attributes
19 available in EPA's Air Quality System (AQS) were summarized, including the monitoring
20 objective, measurement scale, and predominant land-use. Additional features such as monitor
21 proximity to NO_X stationary emission sources within 5 km and emission sources having at least
22 10 tons per year (tpy) were identified using each monitoring site and emission source geographic
23 coordinates and NO_X emissions estimates from the 2011 National Emissions Inventory (NEI).
24 Each of these attributes is summarized in Table 2-8 (active monitors) and Table 2-9 (inactive
25 monitors) to provide perspective on the representativeness of the ambient NO_2 monitoring
26 network in the Philadelphia CBSA.
27   The land-use field indicates the prevalent land use within ¼ mile of the monitoring site.
28 Most of the Philadelphia CBSA monitors are characterized as within residential and commercial
29 areas, with a few falling within industrial and agricultural areas. The measurement scale
30 represents the air volumes associated with the monitoring area dimensions. Most of the
31 Philadelphia CBSA monitors have measurement scales of neighborhood (500 m to 4 km), though
32 one was identified as urban scale (4 to 50 km) along with one microscale monitor (in close
33 proximity, up to 100 m from a source). The monitor objective describes the monitor in terms of
34 its attempt to generally characterize health effects, photochemical activity, transport, or welfare
35 effects. Monitors that would be useful for evaluating public health would be characterized as

1   having a monitor objective of population exposure and/or highest concentration, clearly the

2   intent of most of the monitors in the Philadelphia CBSA. The monitor indicated as having

3   source-oriented objective is the near-road monitor (monitor ID 421010075).

4   Mobile sources (e.g., automobiles) are the most significant contributors to NO_X emissions

5   in the U.S. (2nd draft ISA, section 2.3.1). Except for the new near-road monitor, distances of each

6   ambient monitor to major roads were obtained from the 2008 REA. The estimated distances of

7   the monitors to major roads ranged from a few meters to several hundred meters, although, on

8   average, most of the ambient monitors are placed at a distance of 100 meters or greater from a

9   major road (Table 2-8 and Table 2-9).

10   Stationary sources (e.g., power generating utilities combusting fossil fuels) can also

11   contribute significantly to NO_X emissions in the U.S (2nd draft ISA, section 2.3.1). When

12   considering the active monitors in the Philadelphia CBSA and the most recent NO_X emissions

13   categorized by source type, electric power generation, petroleum refineries, and municipal waste

14   combustion are important stationary sources potentially influencing NO_2 concentrations at a few

15   of the ambient monitors.
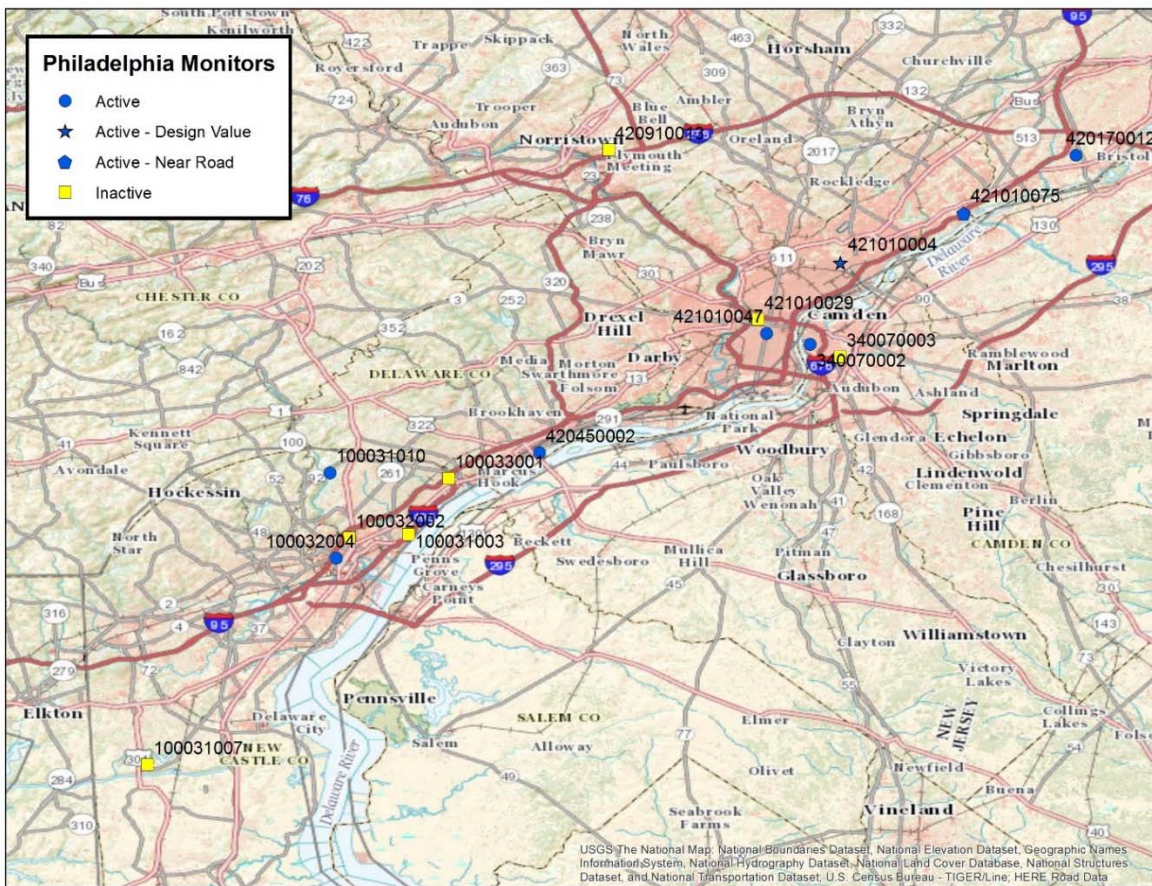
16

17 **Figure 2-6. Locations of the eight active and seven inactive ambient monitors in the**
18 **Philadelphia CBSA.**

Table 2-8. Attributes of active ambient monitors in the Philadelphia CBSA, as of 2014.

Attribute	Operational Monitor ID							
	100031010	100032004	340070002	420170012	420450002	421010004 (design)	421010047	421010075 (near-road)
Latitude	39.817222	39.739444	39.934446	40.107222	39.835556	40.008889	39.944651	40.054128
Longitude	75.563889	75.558056	75.125291	74.882222	75.3725	75.09778	75.165206	74.984802
Elevation (m)	0	0	4	12	3	22	21	9
Start year	2013	2001	2012	1974	1974	1977	1982	2014
End year	2014	2014	2014	2014	2014	2014	2014	2014
Land use	agricultural	commercial	industrial	residential	industrial	residential	residential	commercial
Measurement scale [a]		-	neighbor	neighbor	neighbor	urban	neighbor	microscale
Monitor objective 1 [b]		pop expos	pop expos	pop expos	pop expos	pop expos	pop expos	high conc
Monitor objective 2		high conc		-	-	-	-	source
Distance from roadway (m) [c]	na	82	na	393	413	45	66	12
NO$_X$ Emissions by Stationary Source Type (in total tons per year (tpy), summed for all sources within 5 Km of monitor that emit 10 or more tpy) [d]								
Electricity generation (combustion)		948	26	107	950		379	
Petroleum refinery		-	-	-	2146		1315	
Municipal waste combustor		-	297		1260		-	-
Chemical plant		13	14		275		14	
Pulp and paper plant		-	-	-	240	92	-	-
Wastewater treatment facility		-	14		56	-	14	-
Institutional (school, hosp., prison)		-	-	-	-	30	66	11
Automobile/truck/parts plant		11	-	-	-	-	-	-
Steam heating facility		-	24	-	-	-	24	
Military base		-	-	-	-	12		-
Petroleum storage facility		-	-	-	-	16		-
Source type not characterized	180	248	13	26	11	183		-

[a] Measurement scales: microscale (close proximity, up to 100 m from a source), neighborhood scale (0.5-4 km), and urban scale (4-50 km).
[b] Monitor objectives: population exposure, highest concentration, and source-oriented.
[c] All sites except for the 2014 near-road monitor used information generated from the 2008 REA (see footnote 22). Na is for where monitor not evaluated in 2008 REA.
[d] 2011 NO$_X$ emissions data were obtained from EPA's Emission Inventory System (EIS) Gateway located at https://eis.epa.gov/eis-system-web (2011 NEI version 2). Associated documentation is available at http://www.epa.gov/ttn/chief/net/2011inventory.html#inventorydoc.

Table 2-9. Attributes of inactive ambient monitors in the Philadelphia CBSA and used in analysis of historical NO₂ concentration trends.

Attribute	Historical Monitor ID						
	100031003	100031007	100032002	100033001	340070003	420910013	421010029
Latitude	39.761111	39.551111	39.757778	39.812222	39.923042	40.112222	39.957222
Longitude	75.491944	75.730833	75.546389	75.455556	75.097617	75.309167	75.173056
Elevation (m)	65	20	46	30	7.6	53	25
Start year	1992	1992	1978	1978	1979	1974	1975
End year	2000	1999	1992	1992	2008	2008	2005
Land use	residential	agricultural	commercial	residential	residential	residential	commercial
Measurement scale [a]		-	neighbor	neighbor	neighbor	neighbor	neighbor
Monitor objective 1 [b]	pop expos		high conc	high conc	pop expos	pop expos	high conc
Monitor objective 2	-	-			-	-	
Distance from roadway (m) [c]	189	144	na	na	405	630	103
NOₓ Emissions by Source Type (summed tons per year (tpy), sources within 5 Km emitting at least 10 tpy) [d]							
Electricity generation	948		948	963	26		353
Petroleum refinery	-	-	-	1490		-	-
Chemical plant	27		27		14		-
Automobile/truck/parts plant	-	-			-	-	
Municipal waste combustor	-	-		-	297	735	
Steel mill	-	-	-	166	-	85	
Steam heating facility	-	-			-	-	24
Wastewater treatment facility	-	-	-		14	-	-
Hot mix asphalt plant	-	-	-	-	-	16	
Pharmaceutical manufacturing	-	-			-	26	
Institutional	-	-	-		-	-	66
Type not characterized		-	199	54	13	53	

[a] Measurement scales: microscale (close proximity, up to 100 m from a source), neighborhood scale (0.5-4 km), and urban scale (4-50 km).

[b] Monitor objectives: population exposure, highest concentration, and source-oriented.

[c] All sites except for the 2014 near-road monitor used information generated from the 2008 REA (see footnote 23). Na is for where monitor not evaluated in 2008 REA.

[d] 2011 NOₓ emissions data were obtained from EPA's Emission Inventory System (EIS) Gateway located at https://eis.epa.gov/eis-system-web (2011 NEI version 2). Associated documentation is available at http://www.epa.gov/ttn/chief/net/2011inventory.html#inventorydoc. 2011 NEI may not represent actual emissions when monitor was in operation.

2.3.2.2 Air quality standard scenario adjustments

1
2   As described above in section 2.3.1.3.1, there are two steps to the approach used to adjust
3   ambient concentrations such that they just meet the existing standard(s). First, a proportional
4   factor is needed to adjust the ambient monitoring concentration up to and including the DM1H
5   98th percentile concentration. This proportional adjustment factor can be derived from the
6   currently available monitor design values. Because there are two standards (annual and hourly)
7   and two types of monitoring data considered (area-wide and near-road), there is additional
8   explanation needed. For the annual standard and each separate year of air quality, the result of
9   dividing the standard level (i.e., 53 ppb) by the monitor design value is used to estimate an
10   annual adjustment factor. For example, Table 2-2 indicates the maximum annual average
11   concentration in the Philadelphia CBSA for year 2011 is 20 ppb. Thus, 53/20 yields the
12   minimum proportional adjustment factor of 2.65 based on all available annual average
13   concentrations calculated for the Philadelphia monitors operating in 2011. When considering the
14   hourly standard, the maximum design value for the 3-year average DM1H 98th percentile for
15   2011-2013 is 61 ppb (Table 2-2). Dividing the DM1H standard level of 100 ppb by 61 ppb yields
16   a proportional adjustment factor of 1.64 for this three-year averaging period (2011-2013). For all
17   available area-wide monitors, design values were used in this manner to calculate all possible
18   proportional adjustment factors, with the minimum factors for each year or period summarized
19   below in Table 2-10.[45]
20   While the currently available near-road data do not constitute a complete year for most
21   monitoring sites and years (including the Philadelphia CBSA), the near-road data can still be
22   used to inform the proportional adjustment factor. Many near-road monitors will likely have a
23   complete year of data for 2014, though it is highly unlikely that any monitor would have a
24   complete set of three-year continuous monitoring available for this review. Therefore, the true
25   design values for the current DM1H standard cannot be calculated using the near-road monitor

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[45] Plots of the high-concentration and low-concentration year data for the Philadelphia CBSA (both using the same low- and high-concentration years as Rizzo (2008) and updated to include the most recent low-concentration year) are provided in Appendix B.

1   data. However, we feel it is informative to our estimates to calculate the similar standard

2   averaging time metrics for each type of monitor (near-road and area-wide) using year 2014

3   alone,[46] as well as for any other comparable years where a comparable statistic can be calculated

4   using the near-road data. In Philadelphia, while the near-road monitor (ID 421010075) collected

5   measurement data in 2014, the full year of data has not yet been uploaded to EPAs' Air Quality

6   System (AQS). Thus when using the available concentration data from the near-road monitor

7   (currently, the first three quarters of the year), an estimate of the annual average and single year

8   DM1H 98th percentile are used here to calculate the two potential adjustment factors. Similarly

9   and using the area-wide monitor concentration data available for 2014, two potential adjustment

10   factors are also calculated. Both the near-road and area-wide adjustment factors calculated using

11   the incomplete year data set are provided in Table 2-10.

12   As described above in screening the ambient monitoring data for areas having high

13   ambient concentrations, the hourly standard is the controlling standard for each of the years

14   considered in this analysis. When compared to the annual metric, the hourly metric has the

15   lowest estimated proportional adjustment factors in all comparable instances (or conversely, the

16   greatest relative monitor design value). In the Philadelphia CBSA, the monitor having the

17   highest design value for 2011-2013 (ID 421010004) yields the minimum adjustment factor of

18   1.64, while another area-wide monitor (ID 421010047) yields the minimum adjustment factor of

19   1.66 for 2014. Note that in the Philadelphia CBSA, the area-wide monitor adjustment factor is

20   less than the estimated adjustment factor of 1.78 when using the available 2014 near-road data

21   (Table 2-10). Therefore in this CBSA, the near-road monitor concentrations will not directly

22   inform the value of the proportional adjustment factor.

23   For all of the proposed study areas selected in this air quality assessment, a single

24   proportional factor, derived from one monitor in a CBSA, is used to adjust concentrations (up to

25   and including the DM1H 98th percentile) measured at all of the monitors in that CBSA. Thus, the

26   monitor having the highest design value will have adjusted concentrations that just meet the

27   existing hourly standard (a 3-year average DM1H 98th percentile of 100 ppb), while all other

28   monitors will have hourly design values less than that value. This assumption in applying the

29   single factor derived from one monitor to other monitors is reasonably justified by the following

30   analysis, where we compared DM1H concentrations at the monitor having the highest design

---

[46] The set of 3-year average DM1H 98th percentile design values for 2012-2014 will be calculated for the existing area-wide monitors when these data become available, however still, metrics calculated for the near-road data are most appropriately compared with area-wide monitor data on an individual year basis.

1   value (421010004) with concentrations measured at the other monitors in the CBSA having

2   measurement data for the same years.

3   **Table 2-10. Proportional adjustment factors calculated for the Philadelphia CBSA, 2011-**
4   **2014.**

Monitor Data Set	Standard Averaging Time	Year	Proportional Adjustment Factor
Area-Wide	Annual	2011	2.65
		2012	2.94
		2013	3.12
		2014	2.99
	DM1H	**2011-2013**	**1.64**
		2014[a]	**1.66**
Near-Road	Annual	2014	3.50
	DM1H	2014[a]	1.78

5   **Bold font** indicates the factor to be applied to proportionally adjust hourly ambient concentrations from the minimum through the
6   98th percentile DM1H.
7   [a] A single year is used for the DM1H for 2014 alone to calculate an adjustment factor because neither the near-road or area-wide
8   data set contained a complete year.
9

10   In considering the set of valid years of recent (2001-2013) ambient monitor data available,

11   we selected a number of years during which simultaneous measurements were made at the

12   monitor having the highest current design value and at the five Philadelphia CBSA monitors

13   having more than one year of monitor data over this period. We used a simple linear regression

14   of the DM1H concentrations (0 through 98th percentiles) for each year to generate regression

15   parameter estimates and standard errors, tests of significance (students-t), and linear model fit

16   statistics (R^2) (Table 2-11). In general, the monitor having the highest design value (for 2011-

17   2013) has an overall higher concentration distribution than the other monitors, even considering

18   the earliest years of data included in this analysis. More specifically, there is consistency in the

19   estimated regression slopes for each monitor over time, generally ranging from about 0.85-0.95,

20   and most regression intercepts are negative and generally range from -3 to -5 ppb. Model R^2 are

21   0.98 or better, indicating a strong linear fit for all regressions.

22   These linear regression results suggest similar changes in the overall ambient air quality

23   distribution have occurred with each monitor compared to the monitor having the highest design

24   value over this period of time, providing support to the use of a proportional adjustment factor

25   developed this single monitor and applied to the other monitors in the Philadelphia CBSA. The

26   consistent presence of small, though statistically significant negative intercepts could indicate the

27   contribution of either a constant source emission or a transformation process exclusively

28   influencing concentrations at the monitor having the highest design value. Note that when

29   adjusting this monitor to just meet the hourly standard, this factor (the value of the regression

1   intercept, or this potential exclusive source contribution/transformation process) would also

2   increase proportionally. However, the relationship between the other monitors and the monitor

3   with the highest design value after adjusting hourly concentrations remains preserved (i.e., the

4   concentration adjustment would not change the regression slope).

5   **Table 2-11. Slope and intercept parameter estimates regressing DM1H concentrations (0-**

6   **98th percentile) from the monitor having the highest design value (ID 421010004) on DM1H**

7   **concentrations measured at five Philadelphia CBSA area-wide monitors.**

	421010004	Intercept			Slope			
Monitor ID	Year	Estimate	SE	ProbT	Estimate	SE	ProbT	R^2
420450002	2001	-0.20	0.45	0.66	0.86	0.01	<0.01	0.9852
	2004	-3.47	0.39	<0.01	1.00	0.01	<0.01	0.9899
	2009	-3.18	0.15	<0.01	0.92	0.00	<0.01	0.9975
	2013	-5.19	0.19	<0.01	0.89	0.01	<0.01	0.9951
340070003	2001	-7.77	0.50	<0.01	1.11	0.01	<0.01	0.9893
	2004	-3.86	0.33	<0.01	1.01	0.01	<0.01	0.9929
	2007	1.13	0.29	<0.01	0.90	0.01	<0.01	0.9924
420170012	2001	-4.79	0.37	<0.01	0.97	0.01	<0.01	0.9921
	2004	-1.73	0.36	<0.01	0.85	0.01	<0.01	0.9885
	2009	-5.79	0.21	<0.01	0.91	0.01	<0.01	0.9953
	2013	-5.06	0.32	<0.01	0.83	0.01	<0.01	0.9841
420910013	2001	-5.84	0.42	<0.01	0.92	0.01	<0.01	0.9887
	2004	-6.05	0.21	<0.01	0.88	0.01	<0.01	0.9964
	2007	-1.94	0.24	<0.01	0.87	0.01	<0.01	0.9944
421010047	2001	4.34	0.29	<0.01	0.98	0.01	<0.01	0.9954
	2004	6.90	0.24	<0.01	0.85	0.01	<0.01	0.9949
	2009	2.98	0.35	<0.01	0.91	0.01	<0.01	0.9870
	2013	-0.19	0.27	0.49	0.96	0.01	<0.01	0.9913

8

9   The second step of the approach used to adjust ambient concentrations such that monitors

10   in the CBSA just meet the existing standard(s) involves the use of individual monitor-based

11   adjustment factors. This second set of adjustment factors is used to simulate the observed

12   deviations in linearity when comparing low-concentration to high-concentration years using the

13   same monitor. To derive this set of additional adjustment factors, we first evaluated changes (if

14   any) in the overall distribution of air quality that could have resulted from ambient air quality

15   rule-driven changes in emissions, with a particular focus on the upper percentile concentrations.

16   Staff used the complete historical ambient monitor concentration data set (1980-2013) and first

17   calculated daily maximum concentrations at each monitor having some measurement data across

1   the three-decade period,[47] then generated percentiles of the concentration distribution in 1

2   percent increments, and finally normalized this distribution to reflect a mean of zero and having

3   a standard deviation of one. Normalized DM1H concentrations for the upper 50th percentile (i.e.,

4   51-100 percentiles) were analyzed using principal components analysis (PCA)[48] to determine

5   whether there were multivariable features of the concentration distribution expressing

6   correlations with important explanatory variables such as monitor year or site ID.

7   The first three components output from the PCA comprised nearly 63% of the variance

8   within the data set, effectively reducing the original 50-variable data set to three new, though

9   consolidated variables. Results for the first principal component (PC1, explaining nearly 40% of

10   the variance) are provided in Figure 2-7. Much of the concentration distribution (upwards to

11   about the 90th percentile) is positively correlated with the first component (i.e., the PC1

12   loadings). The remaining upper percentile DM1H variables gradually shift to being negatively

13   correlated with the first component, beginning with the 95th percentile concentrations and

14   progressing to having maximum negative correlations at the 99th and 100th percentiles. The PC1

15   scores are calculated for each individual monitor site-year and are a linear combination of the

16   PC1 loadings and the normalized concentrations. When the PC1 scores are plotted against the

17   year the monitor data were collected, there is a general trend of increasing score with increasing

18   monitor year. The historical (1980s) monitoring data have mostly negative PC1 scores, then a

19   transition from negative to positive scores occurs at or around the mid to late 1990s, with recent

20   (2000s) monitoring data having mostly positive scores for the first principal component.

21   These observed changes in the overall NO_2 concentration distribution over time and occurring

22   across all monitors in the Philadelphia CBSA could correspond to the response to a final EPA

23   rule (the 1998 NO_X SIP Call). The rule required 22 states (including Pennsylvania) and the

24   District of Columbia to submit State implementation plans (SIPs) for reducing emissions of NO_X

25   with such emission reduction measures to be in place by May 1, 2003. Thus, when considering

26   the observed relationship between PC1 scores and PC1 loadings and the consistent relative

27   reductions in the upper percentile concentrations (95th and above) compared with other portions

---

[47] There were a total of six monitors in the Philadelphia CBSA having data spanning the 1980-2013 monitoring period: IDs 420450002, 420910013, 340070003, 420170012, 421010004 and 421010047.

[48] Principal components analysis can be thought of as developing a progressive series of individual linear correlations through a multidimensional variable space (i.e., in this illustrative example, 50 dimensions), with the first correlation/component explaining the greatest amount of variance existing across that multidimensional space (via a single variable comprised of some linear combination of the original 50 variables). Subsequent components explain progressively less variance than prior components, though are generated to again have maximum explanatory power given the unexplained variance that remains.

1   of the concentration distribution, and its correspondence with the timing of the NO_X emissions

2   reduction rule, we elected to use the recent ambient concentration data to best simulate deviation

3   from linearity at upper percentile concentrations that may exist when considering our

4   hypothetical high-concentration adjusted air quality scenario.
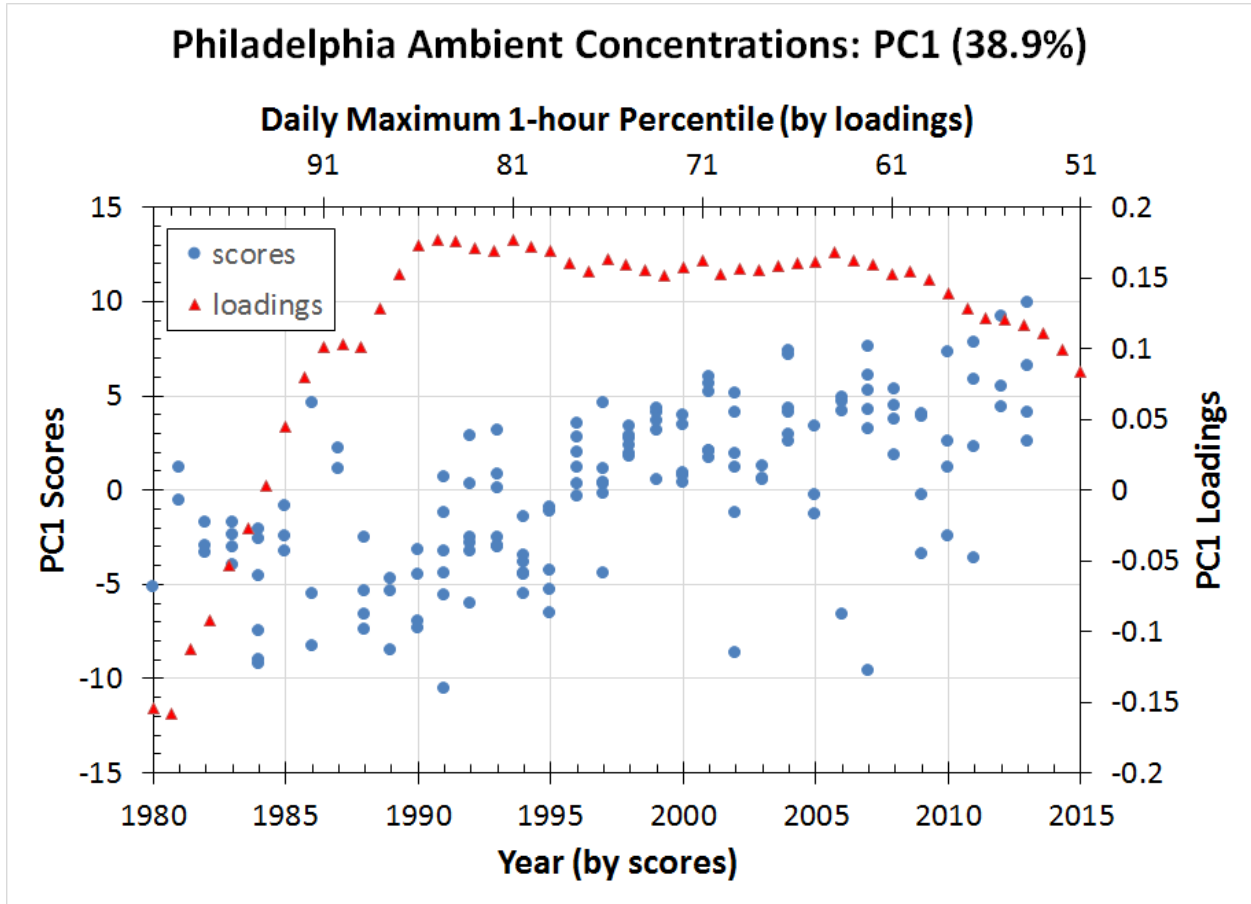
5

6   **Figure 2-7. Principal components (PC) monitor scores plotted by year along with PC**
7   **loadings plotted by DM1H percentile concentration value using the Philadelphia CBSA**
8   **NO_2 ambient concentrations (1980-2013). The scores and loadings plots are an overlay and**
9   **not intended to directly relate the values of the primary (year) and secondary (DM1H**
10   **percentile) X-axes.**

11   The second and third principal components explained 15.4% and 8.5% of the remaining

12   variance, respectively (Figure 2-8). The pattern in PC scores indicates a limited degree of

13   monitor-specific individuality for a few of the sites (IDs 42101004, 421010047, 420450002) and

14   site-years (most in the 1990s), as distinguished from scores at other monitor site years. Again,

15   upper percentile variables (largely the 90th and above) are strongly correlated with both PC2

16   (positive) and PC3 (negative) and largely discordant with the lowest percentile variables (50[th's],

17   PC2) and mid percentile variables (65-70[th's]). These results could indicate nuanced changes in

18   the NO_2 concentration distribution resulted from a localized emission change during that time

1   period, supporting the development of individual monitor-based adjustments to simulate upper

2   percentile concentrations.
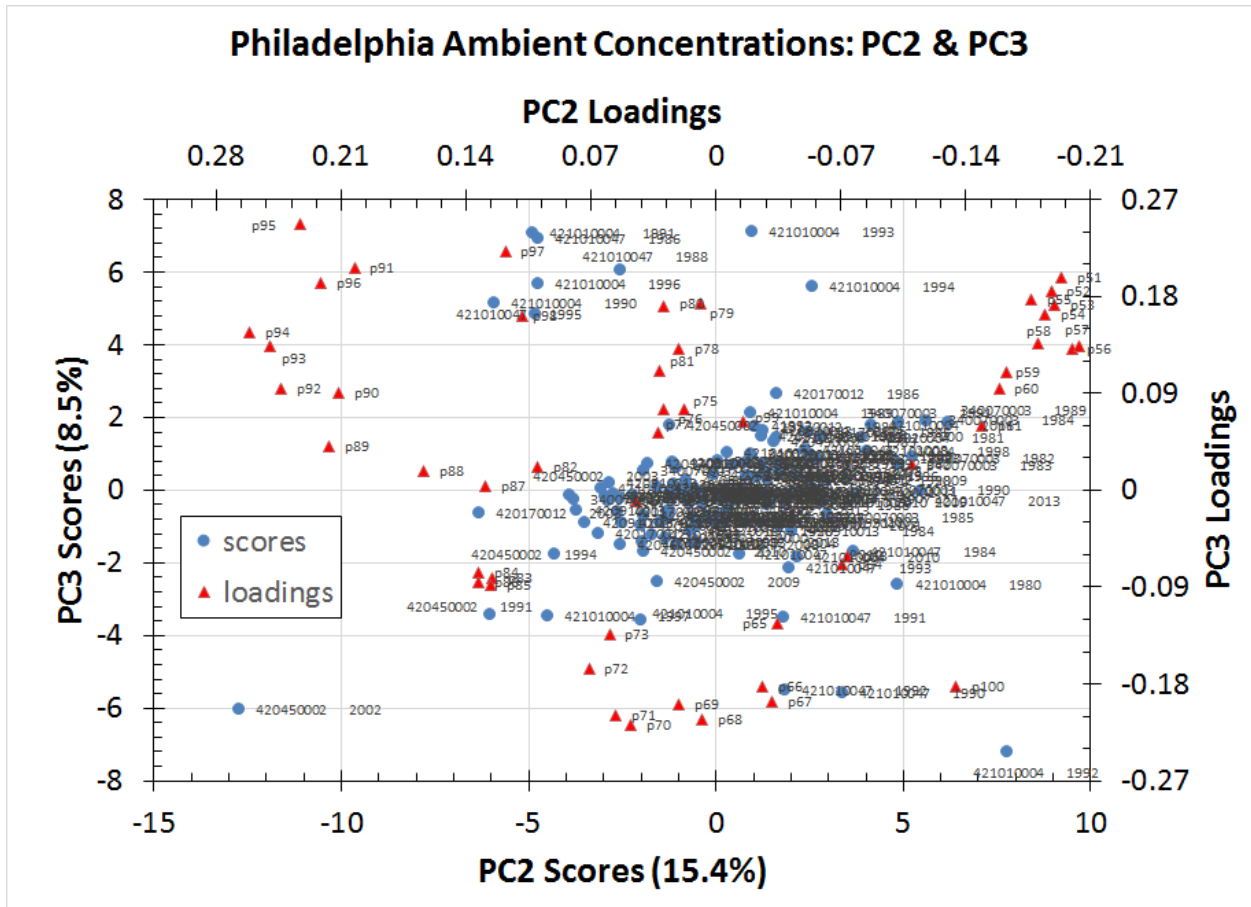
3

4   **Figure 2-8. Principal components scores plotted by loadings for each the second (PC2) and**
5   **third (PC3) components derived using the Philadelphia CBSA NO$_2$ ambient concentrations**
6   **(1980-2013). The scores and loadings plots are an overlay and not intended to directly**
7   **relate the values of the primary and secondary axes.**

8

9   Based on this analysis and the potential impact emission reductions have on

10   concentrations measured across study areas and at local monitoring sites, we calculated the

11   second set of adjustment factors using the 2003-2013 ambient monitoring data in the

12   Philadelphia CBSA. As described above, there are up to seven DM1H concentrations above the

13   98th percentile DM1H for a full year of ambient monitor data. At each monitor and for each year

14   of data, the seven highest DM1H concentrations were divided by their site-year's 98th percentile

15   DM1H concentration to generate a ratio. An adjustment factor for each monitor and DM1H was

16   calculated by averaging across the, at most, 11 ratio values (years) and are summarized in Table

17   2-12. There is general consistency in the adjustment factor across monitors upwards to about the

18   4th highest maximum, but beyond this point up to the maximum DM1H adjustment factor there is

1 increasing variability in the factor across the monitors. The maximum adjustment factor

2 exhibited the widest range of values; the minimum adjustment factor at monitor ID 340070002 is

3 about 21%, while at monitor ID 421010002, the adjustment factor to be applied to the 98th

4 DM1H to estimate the maximum DM1H is 56%.

5 **Table 2-12. Individual monitor-based factors calculated to adjust DM1H ambient NO$_2$**

6 **concentrations above the 98th percentile DM1H in the Philadelphia CBSA.**

Site ID	Adjustment factor derived from ratio of DM1H concentrations to 98th percentile DM1H, averaged across years 2003-2013						
	Max DM1H	2nd DM1H	3rd DM1H	4th DM1H	5th DM1H	6th DM1H	7th DM1H
421010075[a]	1.38	1.21	1.15	1.09	1.06	1.04	1.01
100031010[b]	1.44	1.22	1.17	1.10	1.07	1.05	1.02
100032004	1.53	1.22	1.15	1.12	1.08	1.02	-
340070002[c]	1.21	1.11	1.08	1.05	1.04	1.04	1.01
420170012	1.32	1.18	1.10	1.08	1.04	1.03	1.01
420450002	1.56	1.35	1.15	1.10	1.05	1.03	1.02
421010004	1.38	1.21	1.15	1.09	1.06	1.04	1.01
421010047	1.55	1.35	1.19	1.14	1.09	1.04	1.01

7 [a] The near-road monitor (421010075) uses the ratios derived from the monitor having the highest design value (42101004).
8 [b] Monitor ID 100031010 is newly sited (2013) and outside urban core of Philadelphia (outside Wilmington, DE). Data from a
9 similar monitor (420910013) located outside urban core of Philadelphia (Montgomery County PA operating 2003-2008) was
10 used to calculate ratios.
11 [c] Monitor ID 340070003 (operating during 2003-2008) is sited in close proximity to newly sited monitor ID 340070002
12 (operating during 2012-2013). The data from both monitors were combined to calculate ratios.
13

14   To estimate ambient concentrations that just meet the existing standard for years

15 evaluated in this illustrative example, for each year of ambient monitoring data and at all

16 monitors, the appropriate year proportional factor described above and found in Table 2-10 is

17 applied to all DM1H concentrations up to and including the 98th percentile DM1H. Then, the

18 remaining upper percentile concentrations above the DM1H 98th percentile are adjusted for each

19 of the eight monitors in the Philadelphia CBSA (the seven area-wide and one near-road monitor)

20 using the individual monitor-based adjustment factors provided in Table 2-12.

21   The results of applying these adjustments are illustrated in Figure 2-9 for one year of

22 concentrations (2011) measured at the monitor (ID 421010004) having the highest design value.

23 Plotted in this figure are the unadjusted DM1H concentrations, concentrations adjusted to just

24 meet the existing hourly standard using a proportional factor alone (and used in the 2008 REA),

25 and concentrations adjusted to just meet the existing hourly standard using a proportional factor

26 and additional factors for concentrations above the 98th percentile DM1H. Concentrations at or

27 above the DM1H 80th percentile are plotted to highlight this portion of the distribution. Using the

28 proportional adjustment alone for this year (an increase of 64%) appropriately increases the 80th

29 and 100th DM1H unadjusted concentrations of 44 and 88 ppb (gray line, Figure 2-9) to 72 and

1   144 ppb (red line, Figure 2-9). When addressing deviations from linearity above the 98th

2   percentile, upper percentile concentrations extend to somewhat higher concentrations (blue line,

3   Figure 2-9) when compared with that using a proportional factor alone to adjust all

4   concentrations. For example and by design, the proportionally adjusted 98th percentile DM1H

5   concentration of 125 ppb is used with the maximum DM1H adjustment factor of 1.38 to estimate

6   a maximum DM1H concentration of about 173 ppb.

7
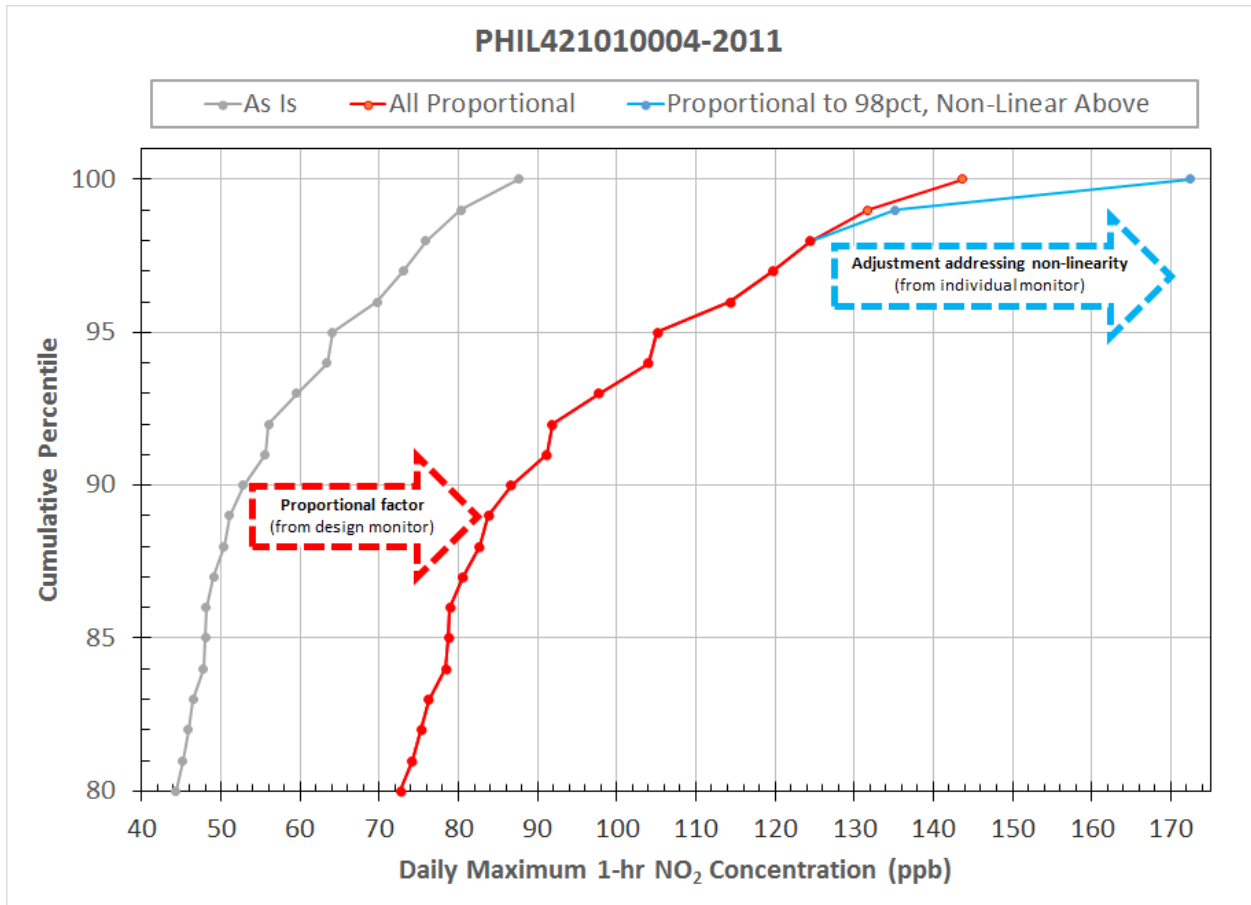
Figure 2-9. Distribution of unadjusted (as is) 2011 ambient NO$_2$ concentrations, that adjusted using a proportional factor alone (all proportional), and that adjusted using a combined proportional factor and ratio approach (proportional to 98th percentile, non-linear above) in the Philadelphia CBSA at monitor ID 421010004.

2.3.2.3 Simulated on-road concentrations

13   In this example calculation, we applied results from one of the proposed approaches

14   discussed in section 2.3.1.3.2 to simulate on-road concentrations for two air quality scenarios;

15   the first scenario using the unadjusted near-road concentrations (monitor ID 421010075) and the

16   second scenario following an adjustment to just meet the existing standard at this same near-road

17   monitor. A concentration increase of 15% developed from the Las Vegas study was universally

1   applied to the available 2014 near-road monitor DM1H NO$_2$ concentrations[49] to estimate the

2   distribution of on-road NO$_2$ concentrations (see section 2.3.1.3.2 and Table 2-7).

2.3.2.4 Number of benchmark exceedances

4   A total of two air quality scenarios were evaluated in the Philadelphia CBSA (i.e.,

5   unadjusted concentrations, those adjusted to just meet the existing hourly standard), and for both

6   of these air quality scenarios, we simulated on-road NO$_2$ concentrations based on the available

7   2014 near-road monitoring data. Staff counted the number of days per monitor site-year the

8   DM1H ambient concentrations exceeded the 1-hour benchmark levels of 100, 200, 300, and 400

9   ppb, separately considering the area-wide, near-road and on-road concentrations. Presented are

10   the mean (averaged across all monitors for the area-wide) and maximum (maximum at a single

11   monitor) number of benchmark exceedances for each year.

12   Benchmark exceedance results using the unadjusted 2011-2014 air quality are presented

13   in Table 2-13. There were no exceedances of any benchmark when considering unadjusted air

14   quality measurements made at either the area-wide or near-road monitors. When simulating on-

15   road concentrations using the unadjusted 2014 near-road concentrations, there were no

16   exceedances of the 100 ppb 1-hour benchmark level per year (and hence no exceedances of any

17   of the higher benchmark levels). As a reminder, a complete year of data is not yet available for

18   2014, thus it is still possible that there could be an occurrence of a benchmark exceedance for

19   this year.

20   Benchmark exceedance results using the unadjusted 2011-2014 air quality adjusted to just

21   meet the existing 1-hour standard are presented in Table 2-14. On average, there are a handful of

22   exceedances per year of the 100 ppb 1-hour benchmark level considering the area-wide monitors,

23   though the maximum estimated number of days with an exceedance of that same benchmark in a

24   single year could be just over 20. Note that because the 2014 monitoring data were not yet fully

25   complete, it is possible there could be additional days having benchmark exceedances. However,

26   the actual estimated number of benchmark exceedances when the monitoring set is complete is

27   likely to fall within the range already estimated using the other years of air quality. When

28   simulating on-road NO$_2$ concentrations using the 2014 near-road monitor data adjusted to just

29   meet the existing 1-hour standard, there is only one additional day having an exceedance of the

30   100 ppb benchmark level compared with that estimated at that near-road monitor. If

---

[49] The Philadelphia near-road monitor 421010075 is sited 12 m from the target roadway.

1 extrapolating this data simply to represent a full year, it is possible that there could be as many as

2 8 days per year where the 100 ppb 1-hour benchmark level is exceeded. For all study area

3 locations (area-wide, near-road, and on-road) there were no exceedances of 1-hour benchmark

4 levels at or above 200 ppb.

5 **Table 2-13. Mean and maximum number of days per year ambient monitor NO$_2$**
6 **concentrations (area-wide, near-road, simulated on-road) are at or above selected 1-hour**
7 **benchmark levels in Philadelphia CBSA, unadjusted air quality.**

Study Area Location	Year	DM1H ≥ 100 ppb		DM1H ≥ 200 ppb		DM1H ≥ 300 ppb		DM1H ≥ 400 ppb	
		Mean	Max	Mean	Max	Mean	Max	Mean	Max
Area-Wide	2011	0	0	0	0	0	0	0	0
	2012	0	0	0	0	0	0	0	0
	2013	0	0	0	0	0	0	0	0
	2014[a]	0	0	0	0	0	0	0	0
Near-Road[b]	2014[a]	-	0	-	0	-	0	-	0
On-Road[b]	2014[a]	-	0	-	0	-	0	-	0

8 [a] The monitoring data for available for 2014 are not for a full year (e.g., the near-road monitor has data for quarters 1 through 3).
9 [b] There is only one near-road monitor in the Philadelphia CBSA, therefore means are not calculated.
10

11 **Table 2-14. Mean and maximum number of days per year ambient monitor NO$_2$**
12 **concentrations (area-wide, near-road, simulated on-road) are at or above selected 1-hour**
13 **benchmark levels in Philadelphia CBSA, air quality adjusted to just meet the existing**
14 **standard.**

Study Area Location	Year	DM1H ≥ 100 ppb		DM1H ≥ 200 ppb		DM1H ≥ 300 ppb		DM1H ≥ 400 ppb	
		Mean	Max	Mean	Max	Mean	Max	Mean	Max
Area-Wide	2011	6	23	0	0	0	0	0	0
	2012	2	4	0	0	0	0	0	0
	2013	1	2	0	0	0	0	0	0
	2014[a]	3	6	0	0	0	0	0	0
Near-Road[b]	2014[a]	-	5	-	0	-	0	-	0
On-Road[b]	2014[a]	-	6	-	0	-	0	-	0

15 [a] The monitoring data for available for 2014 are not for a full year (e.g., the near-road monitor has data for quarters 1 through 3).
16 [b] There is only one near-road monitor in the Philadelphia CBSA, therefore means are not calculated.
17

18 For comparison, the 2008 REA calculated the number of exceedances for two of the

19 benchmarks (i.e., 100 and 200 ppb) using 2001-2003 ambient air quality adjusted to just meet the

20 now existing standard, as well as using a probabilistic adjustment factor to simulate on-road NO$_2$

21 concentrations (Table 2-15). Results for the 2008 REA were summarized by site-year rather than

22 by year alone and the available area-wide monitors were separated by two distance from road

23 categories; nevertheless the overall comparison to the current analysis remains meaningful to a

24 certain extent. There is consistency between the 2008 REA estimates and the current calculations

25 when considering the area-wide monitors adjusted to just meet the existing standards, as both

26 indicate on average approximately 3-6 days per year where the 100 ppb 1-hour benchmark is

1   exceeded. Both analyses also predict, on average, no exceedances of the 200 ppb 1-hour

2   benchmark at the area-wide monitors with air quality adjusted to just meet the existing standards.

3   There are however large differences in the number of days per year having exceedances

4   when considering the simulated on-road concentrations. In the 2008 REA, on average, over 100

5   days per year were estimated to have an exceedance of the 100 ppb 1-hour benchmark, along

6   with an upper percentile estimate of nearly 300 days per year. This is likely the result of the

7   selected monitor data used in the 2008 REA, whereby monitors sited at least 100 m from a major

8   road were used to simulate the on-road NO_2 concentrations. The on-road simulation approach

9   assumed monitors sited 100 m or greater from a road was a reasonable distance to not have a

10   direct influence from road emissions, an important uncertainty identified in that assessment

11   (2008 REA, section 7.4.6). In reviewing the current monitor attribute data provided in Table 2-8

12   and Table 2-9, while it is possible that at that distance from a road these monitors would have

13   limited direct contribution from roadway emissions, there remains the potential for other source

14   emissions to substantially influence concentrations measured at the monitor that were not

15   accounted for before simulating the on-road concentrations. For example, NO_2 concentrations at

16   monitor ID 420450002 (located 413 m from a major road) could be largely influenced by several

17   stationary sources, including NO_X emissions from electricity generation (via combustion),

18   petroleum refineries, and municipal waste combustion (Table 2-8). Using this generally high-

19   concentration monitor (see 2008 REA Appendix A, Table A-54 and Figure A-54) and other

20   monitors having similar stationary source emissions in close proximity (e.g., monitor

21   421010029) to simulate the on-road NO_2 concentrations would likely overestimate the number of

22   benchmark exceedances.

23   **Table 2-15. Mean and upper percentile number of days per year ambient monitor NO_2**
24   **concentrations (area-wide and on-road) are at or above selected 1-hour benchmark levels**
25   **in the 2008 REA Philadelphia CBSA, 2001-2003 air quality adjusted to just meet the**
26   **existing standard.**

Area-wide Monitor distance from road	Site-years [a] (n)	DM1H ≥ 100 ppb		DM1H ≥ 200 ppb	
		Mean	P99	Mean	P99
>20m-<100m[b]	7	5	11	0	0
≥100m[c]	14	3	15	0	1
On-Road[d]	1400	116	294	7	68

27   [a] For area-wide monitors, this is the number of measurement monitor site years available (e.g., two monitors for two years with 3
28   monitors for one year equals 7 site-years of data). The on-road site years are from 1,000 simulations using the 14 site-years of
29   data from monitors sited ≥100m from a major road).
30   [b] From Table 7-23 (U.S. EPA, 2008a).
31   [c] From Table 7-24 (U.S. EPA, 2008a).
32   [d] From Table 7-28 (U.S. EPA, 2008a).

3 HUMAN EXPOSURE ASSESSMENT

In the last review, in addition to analyses of NO_2 air quality, the EPA used an exposure model to generate estimates of 1-hour personal NO_2 exposures in an urban study area. These modeled 1-hour personal exposures were compared to 1-hour health effect benchmarks ranging from 100 to 300 ppb.[50] In the current review, staff will use results of the updated comparison of NO_2 air quality with health effect benchmarks (Chapter 2, above) to consider the potential utility of performing an updated assessment of personal NO_2 exposures. To the extent the air quality assessment indicates little potential for the occurrence of ambient NO_2 concentrations at or above the various 1-hour health effect benchmark levels (i.e., indicating little potential for NO_2 exposures of public health concern), the added value of more refined estimates of personal NO_2 exposures would be limited. Alternatively, to the extent the air quality assessment indicates the potential for NO_2 exposures of public health concern, more refined estimates of NO_2 exposures will be considered in the current review.

In this latter scenario, conclusions on whether to conduct an updated NO_2 exposure assessment will also be informed by 1) the extent to which important uncertainties identified in the last review have been addressed by newly available information, approaches, and tools, and 2) judgments as to the likelihood that an updated quantitative assessment would substantially add to our understanding of NO_2 exposures, beyond the insights gained from the exposure assessment conducted in the last review. This chapter provides overviews of the exposure assessment conducted in the last review (section 3.1) and the new information, approaches and tools available in the current review that could potentially inform an updated exposure assessment, should one be supported (section 3.2).

3.1 OVERVIEW OF EXPOSURE ASSESSMENT IN THE LAST REVIEW

In the 2008 REA, population exposures to ambient NO_2 were simulated using the Air Pollutants Exposure (APEX) model (U.S. EPA, 2008a, Chapter 8). The APEX model simulates the movement of individuals through time and space across a user-defined modeling domain and estimates their exposures to pollutants within indoor, outdoor, and in-vehicle

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[50] As described above for the air quality assessment (section 2.1), 1-hour health effect benchmark concentrations were based on a meta-analysis of individual-level data from controlled human exposure studies of NO_2-induced airway responsiveness (U.S. EPA, 2008b, Table 3.1-3).

1   microenvironments. APEX takes into account the most critical factors that contribute to total

2   human exposure to ambient NO_2, including the temporal and spatial distributions of people and

3   the NO_2 concentrations they experience as they travel throughout an urban study area (including

4   on- and near-roads), and the variation of NO_2 concentrations within various microenvironments

5   (U.S. EPA, 2008a, section 8.2).

6   The ambient concentrations for the exposure assessment in the last review were generated

7   using the EPA's air dispersion model, AERMOD. AERMOD was used to simulate hourly NO_2

8   concentrations at a census block-level and at link-based roadway receptors, considering

9   emissions from stationary, area-wide, and on-road mobile sources (U.S. EPA 2008a, section 8.4).

10   Multiple air quality scenarios were evaluated, including air quality adjusted to just meet the

11   existing annual standard and air quality adjusted to just meet alternative 1-hour standards with

12   levels ranging from 50 to 200 ppb (U.S. EPA, 2008a, section 8.9.4).[51] Given the resource-

13   intensive nature of this approach, only one study area was selected for analysis (four counties

14   comprising the core urban area in Atlanta, GA). The exposure assessment served to complement

15   the results of the broader, less resource-intensive NO_2 air quality characterization described

16   above that largely relies on existing ambient monitor concentrations (section 2.1).

17   Exposure estimates focused on people with asthma, based on the evidence for potentially

18   increased health risks following NO_2 exposures (U.S. EPA, 2008b). Exposures were estimated

19   for all people with asthma (0 to 99 years of age) and in school-age children with asthma (5 to 17

20   years of age). The 2008 REA noted that children spend more time engaged in outdoor activities,

21   possibly increasing their NO_2 exposures. The 2008 REA compared the estimated 1-hour personal

22   exposures to health effect benchmarks ranging from 100 to 300 ppb; these benchmarks were

23   based on the 2008 ISA's assessment of controlled human exposure studies that evaluated airway

24   responsiveness following NO_2 exposures (see above, sections 2.1 and 2.2.2) (U.S. EPA, 2008a,

25   section 8.9.4).

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[51] Adjusted air quality was based on the years 2001 to 2003. This three-year period was selected to encompass the most recent year of NO_X emissions data available (i.e., 2002) at the time the exposure assessment was conducted (U.S. EPA, 2008a).

3.1.1 Key Results

The 2008 REA presented a number of results from the exposure assessment in Atlanta (U.S. EPA, 2008a, section 8.9.4 and Appendix B), including the following:

- Roadway-related exposures accounted for more than 99% of exposures to NO_2 concentrations at or above 1-hour health effect benchmarks in Atlanta (U.S. EPA, 2008a, Figures 8-17 and 8-18). Of these roadway-related exposures, approximately 70% were estimated to occur in vehicles, with the remainder estimated to occur outdoors near roads.

- When air quality was adjusted to just meet the existing annual standard in Atlanta, almost all people with asthma (i.e., 88% to 99%, depending on the year) were estimated to experience 1-hour exposures to NO_2 concentrations at or above 300 ppb at least six times per year.[52] For 1-hour health effect benchmarks of 100 or 200 ppb, all people with asthma were estimated to experience at least six exposures to these concentrations per year (U.S. EPA, 2008a, section 8.9.4.3; Appendix B, Tables B-43 to B-54).

- Compared to the annual standard, when air quality was adjusted to just meet 1-hour standards with levels of 100 or 50 ppb, there were substantial reductions in the number of people estimated to experience six or more exposures per year to 1-hour NO_2 concentrations at or above 300 ppb. Reductions were more modest for lower health effect benchmarks (i.e., 100 to 200 ppb) and for smaller numbers of occurrences (i.e., one or more, two or more, etc.). Reductions were also more modest for 1-hour standards with levels of 150 or 200 ppb (U.S. EPA, 2008a, Appendix B, Tables B-43 to B-54).

3.1.2 Uncertainties and Limitations

Important uncertainties identified for the 2008 REA NO_2 exposure estimates included some of the same uncertainties identified for the air quality assessment (section 2.1.2, above). For example, uncertainties in the approach used to adjust air quality and uncertainties in health effect benchmarks were also important for the 2008 exposure assessment. The 2008 REA also identified uncertainties specifically associated with the exposure model inputs, approach, and

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[52] Six or more days per year was the largest number of occurrences that was specifically reported in the 2008 REA. Similar results were obtained for all people with asthma and children with asthma, though estimated exposures were somewhat higher for children (U.S. EPA, 2008a, section 8.9.4.3; Appendix B, Tables B-43 to B-54).

1 estimated benchmark exceedances. These uncertainties, and their potential implications, are
2 discussed in detail in the 2008 REA (U.S. EPA, 2008a, section 8.12).
3   Table 3-1 (below) provides the qualitative summary of the key uncertainties related to the
4 exposure modeling and evaluated in the 2008 REA (U.S. EPA, 2008a, Table 8-17). While our
5 approach to evaluating uncertainties has evolved since the time of the 2008 REA,[53] the 2008
6 REA characterization remains a reasonable starting point for the discussion that follows,
7 focusing on the key uncertainties identified in the 2008 REA and any newly identified potential
8 elements here. As such, this current evaluation is not intended to serve as a re-characterization of
9 all previously identified uncertainties nor does it serve to fully characterize uncertainty in any
10 newly identified elements. If a new exposure assessment is performed using a similar modeling
11 approach, each element listed here (and any newly identified elements) would be newly
12 evaluated and characterized. The following discussion focusses on a few of the most important
13 uncertainties identified and evaluated in the 2008 REA[54] along with expanded context for
14 particular elements not discussed at that time.
15   Regarding AERMOD inputs and algorithms, one important uncertainty identified as
16 specific to the 2008 REA exposure assessment was the AERMOD estimated concentrations used
17 to represent the air quality surface across the Atlanta study area (U.S. EPA 2008b, section 8.12).
18 A performance evaluation using limited ambient measurement data suggested a potential bias
19 towards overestimating ambient concentrations, potentially attributable to uncertainty in mobile
20 source emissions and/or diurnal profiles used as inputs (among other sources of uncertainty)
21 (U.S. EPA, 2008a, section 8.4.8). Given the few monitors available for the evaluation, and the
22 overall strong confidence in the AERMOD system and other input data, the 2008 REA did not
23 adjust all estimated concentrations across the entire 4-county modeling domain based on the

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[53] Included in the most recent uncertainty characterization approach used for the O_3 REA are implicit evaluations of the magnitude and direction of influence the uncertain element has on exposure results as well as an evaluation of the degree of uncertainty in the knowledge base used to inform the characterization. The 2008 REA approach only characterized the magnitude and direction of influence.

[54] The three categorizations used in the 2008 REA to characterize uncertainty (low, medium, and high) were intended to indicate the impact [magnitude] the type [element] of uncertainty potentially has on the estimated exposures and the potential direction of bias [influence] (under- or over-estimate, or unknown). A range in the magnitude is possible when considering multiple influential components exist within each of the identified elements. For example, regarding the 'low – high' range reported for the CHAD database, it was judged that there is a small (low) impact on exposure estimates when using CHAD diary data from selected studies rather than using CHAD as a whole. Also, the CHAD diaries used in the exposure modeling (generally nationally representative data as a whole) does not directly account for longer drive times observed for Atlanta commuters, potentially leading to large (high) exposure underestimations (see 2008 REA. Section 8.12.2.2). Details as to how particular judgments were made regarding each element in Table 3-1 here are provided in section 8.12 of the 2008 REA.

1   differing concentrations observed at the few monitor locations. Nevertheless, ensuring the proper

2   characterization of the hourly ambient concentrations input to APEX in estimating exposure is

3   important and, in the absence of having spatially and temporally robust ambient measurement

4   data, this could remain as a key uncertainty.

5   **Table 3-1. Summary of 2008 REA qualitative uncertainty analysis for the exposure**

6   **assessment.**

Source	Type [Element]	Concentration/ Exceedance Bias [Influence] Direction	Characterization of Uncertainty [Magnitude]
AERMOD Inputs and Algorithms	AERMOD formulations for mobile sources	unknown	Low
	On-road emissions	over	Low – Medium
	O$_3$ monitoring data	over	Low
	Use of unadjusted NO$_2$ concentrations	unknown	Low – Medium
	Meteorological data	unknown	Low – Medium
APEX Inputs and Algorithms	Population data base	both	Low
	Commuting data base	both	Low – Medium
	CHAD data base	under	Low – High
	Meteorological data	both	Low
	Air exchange rates	unknown	Medium
	A/C prevalence	none	Low
	Indoor sources not modeled	under	Medium
	Indoor decay distribution	under	Low – Medium
	Indoor concentration distribution	under	Medium – High
	Longitudinal profile	both	Low

7   Brackets [ ] in column heading indicate current/evolved terms used in characterizing uncertainty. Our current approach would
8   consider uncertainty in the knowledge-base, characterized using three categorical ratings.

9

10   Regarding APEX inputs and algorithms, the commuting database was identified as a

11   potentially important influential source of uncertainty. While there is limited uncertainty in the

12   U.S. Census-derived database itself *per se*, the potentially influential elements of uncertainty

13   considered here are the overall use of it in our exposure modeling approach and what may not be

14   properly accounted for in moving simulated individuals across the exposure modeling domain.

15   For example, the commuting option is only applied to employed individuals; when a work event

16   occurs, the individual travels to a probabilistically determined work tract and then returns to the

17   home tract when a home event occurs. Ambient concentrations used for estimating

18   microenvironmental concentrations associated with other travel events (e.g., trips to school, a

19   park, or grocery store) are generally limited to either the domain average of all ambient

1 concentrations (which was the approach used in the 2008 REA) or a random selection of a user-

2 defined number of receptors.

3   The Consolidated Human Activity Database (CHAD) serves as a fundamental component

4 in the exposure model that connects simulated individuals with the estimated

5 microenvironmental concentrations. There were several components of CHAD that were

6 considered in the 2008 REA as potentially influential and contributing to uncertainty in exposure

7 estimates including a qualitative evaluation of survey year and methodology used, however

8 given new consideration here, a few additional components can be identified. For example and

9 related to the commuting issue identified above, an important element was mentioned therein

10 regarding the representativeness of CHAD diary drive durations (largely from national surveys)

11 for the people simulated in the Atlanta model domain. In addition, the 2008 REA also considered

12 the potential impact on exposures from using diaries dating back to 1984 and whether these

13 could appropriately capture commute times for the simulated years (2001-2003). Not discussed

14 in the 2008 REA uncertainty characterization however, though related to both commuting and

15 the CHAD diaries used in the exposure assessment, is the lack of linking the Consolidated

16 Human Activity Database (CHAD) diary driving event durations used by APEX with each

17 simulated individual's commute distance. Also not directly identified in the 2008 REA as an

18 uncertainty, though relevant to an evaluation of the CHAD database, were the limited number of

19 diaries available for use by APEX (approximately 15,000 total), particularly those from school-

20 age children (approximately 3,000 diary days).

21   In addition, though not specifically identified as an exposure uncertainty in the 2008 REA,

22 there was uncertainty associated with the factors approach used to adjust 1-hour AERMOD

23 ambient concentrations to predict on- and near-roadway concentrations. While AERMOD

24 predicted 1-hour NO_2 concentrations occurring at roadway link-based receptors, these estimated

25 on-road NO_2 concentrations could not be used directly as an input to APEX based on its existing

26 configuration. Thus, a distribution of factors were developed from the AERMOD predicted on-

27 road and census tract level concentrations (U.S. EPA, 2008a, section 8.7.2.5). These differently

28 derived, though related, on-road factors used by APEX, along with the number of estimated on-

29 road peak concentrations, were compared with those used for the air quality characterization

30 (U.S. EPA, 2008a, section 8.4.8.3). The two similar, though independently developed,

31 distributions of factors used to simulate on-road concentrations were found to be comparable

32 across a wide range of estimated values, though they diverged at upper percentiles of the two

33 concentration distributions.

34   A few remaining important uncertainties identified in the 2008 REA were related to

35 estimation of indoor exposures (i.e., air exchange rates developed from a North Carolina research

36 study and applied to Atlanta), including the added exposure concentration contribution

1   originating from a single indoor source (i.e., gas stoves). Indoor source emissions will increase
2   the frequency of peak NO_2 exposure concentrations, particularly given their short-term event
3   durations (e.g., cooking with a gas stove). However, substantial uncertainty likely remains in
4   generating reasonable emission rates for all indoor sources (e.g., gas stoves, heaters, fireplaces)
5   and in appropriately simulating these types of events as they occur throughout the day. These
6   general uncertainties regarding accurately estimating the source contribution to indoor
7   concentrations, coupled with their limited relevance to improving our understanding of the
8   relationship between ambient air quality and ambient-related exposures, and recognizing the
9   aforementioned important uncertainties in simulating movement of individuals across the
10   modeling domain and estimating certain microenvironmental concentrations, likely preclude the
11   need for a focused consideration in this assessment.
12   Finally, broad context was added to the Atlanta study area by evaluating influential
13   attributes that may affect the estimated exposures (i.e., population density near roads, air
14   exchange rates/air conditioning prevalence, roads per capita, and daily vehicle miles travelled-
15   DVMT). We compared the values determined for Atlanta with those in other potential study
16   areas (e.g., Boston, Los Angeles, etc.) to evaluate the overall representativeness of the Atlanta
17   study area. For a few elements Atlanta was similar to other urban areas (e.g., population per
18   total roadway miles), potentially indicating some degree of representativeness of the Atlanta
19   study area exposure results for other urban areas. For other elements Atlanta had lower values
20   (e.g., roads per capita, DVMT), indicating a lack of representativeness. Regardless of the
21   outcome, having one study area with refined exposure estimates in the 2008 REA presents an
22   issue regarding representativeness and is an uncertainty.

23   ## 3.2   CONSIDERATION OF NEWLY AVAILABLE INFORMATION

24   ### 3.2.1 Emissions Inventory

25   The 2008 REA document relied on emissions from the 2002 NEI, which was the most
26   current inventory available at that time. The NEI is compiled, in detail for all sectors every three
27   years, such that since then there are data for NEI years 2005, 2008, and the most current, 2011.
28   The NEI's NO_X emissions trend has continued to show reductions, due both to actual decreases
29   and to methodological updates that improve estimation accuracy. Notable methods changes
30   include the evolution of EPA's on-road emissions models from the Mobile model, used in the
31   2002 NEI, to todays' Motor Vehicle Emission Simulator (MOVES). The more recent model
32   results for on-road, the sector with the largest NO_X emissions at about 35%, estimates slightly
33   lower overall on-road emissions and a redistribution among motor vehicle types. The following
34   graphs show NO_X trends from all sectors and from the largest contributing sectors (mobile and
35   stationary fuel combustion, which account for approximately 75% of emissions), respectively.

1

2

3
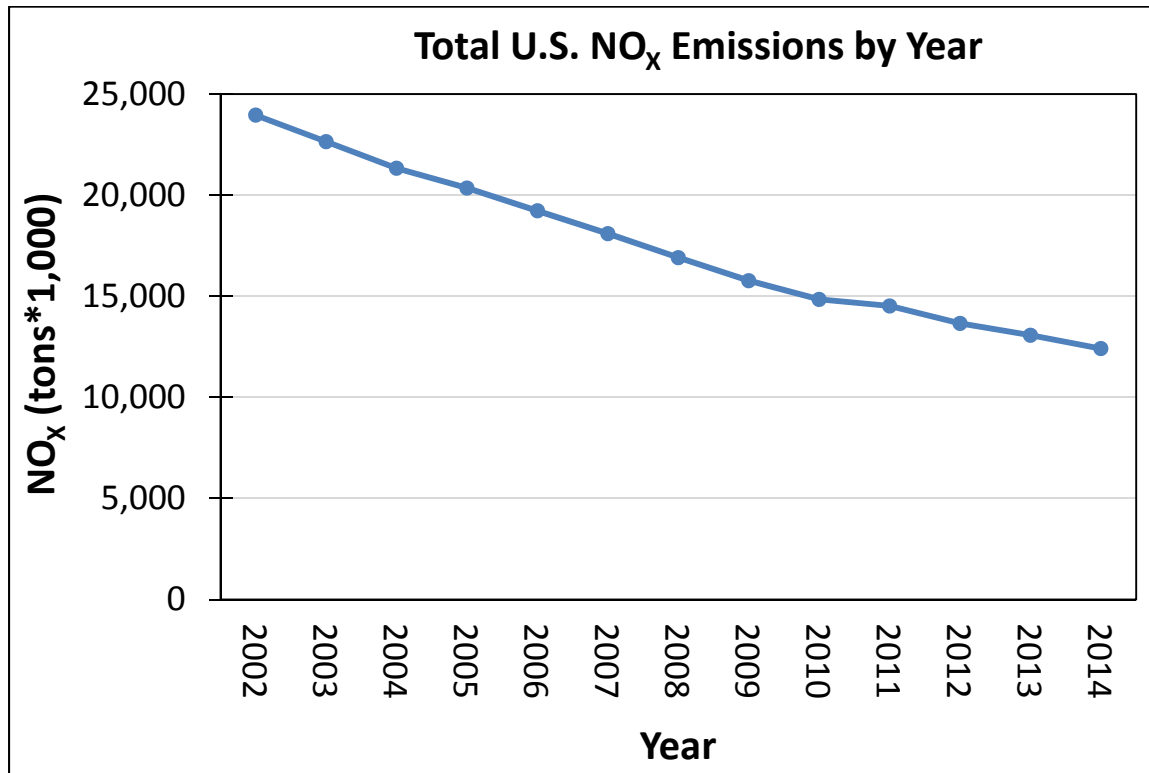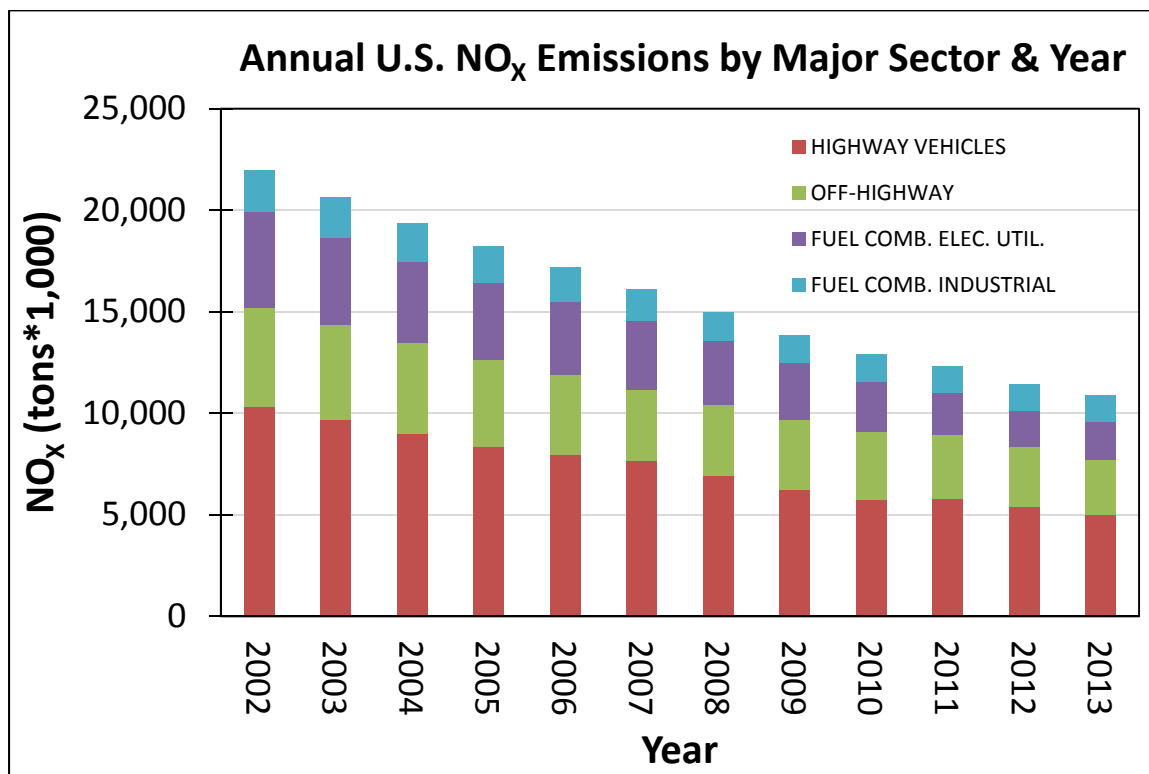
Figure 3-1. Total annual NO$_X$ emissions (top) and annual emissions stratified by top four
sectors (bottom), 2002-2013.

1   For mobile source modeling, the 2008 REA used traffic demand modeling conducted by
2   the Atlanta Regional Commission, which resulted in four periods of a temporal profile,
3   "morning, afternoon, evening, and nighttime." However, the most up to date version of the
4   mobile model (currently MOVES 2014) includes emission factors that have temperature
5   sensitivity as well as the ability to model with more specific temporal profiles. As part of the
6   recent NEI process (2011 NEIv2), EPA has collected regional specific temporal profiles either
7   from individual states submissions or by analyzing traffic count data from the Vehicle Travel
8   Information System (VTRIS). These temporal profiles vary by geography as well as having
9   distinct profiles by vehicle type and road type for day of the week and diurnal profiles (weekday
10   vs Saturday vs Sunday). These resulting profiles are expected to be more temporally
11   representative than those used previously. In addition to updating the profiles, the impact
12   temperature has on emissions could be captured by using hourly and region specific meteorology
13   (e.g., from Weather Research and Forecast modeling or from detailed ambient monitor data).
14   Temporal variability in meteorology could either increase or decrease emissions at any specific
15   hour by accounting for events such as increased emissions from air conditioning use or from cold
16   starts.

3.2.2 Air Quality Modeling

18   The AERMOD modeling system, including the AERMOD dispersion model and its
19   meteorological preprocessor, AERMET, have had 8 major update cycles since the last
20   NO_2 REA was conducted. These updates include major revisions to the NO_2 chemistry
21   options used to estimate NO/NO_2 partitioning (i.e., the Ozone Limiting Method and
22   Plume Volume Molar Ratio Method), which have resulted in more accurate estimates of
23   NO_2 concentrations from stationary and mobile sources. Newer versions of AERMOD
24   have also incorporated new options for applying background data (NO_2 and ozone for
25   NO/NO_2 conversion) into the modeling scenario based on the wind direction and the
26   location of available monitoring data. Additionally, AERMET has been updated to
27   incorporate high-resolution meteorological data in order to provide more representative
28   and accurate inputs to the AERMOD model.

3.2.3 Exposure Modeling

30   There have been a number of updates made to the APEX model and many of the input
31   data sets used. Table 3-1 above highlights several important elements of the model and its inputs
32   and serves to inform a basic structure to the discussion that follows.

3.2.3.1 US population and commuting database

APEX currently uses the 2010 census information, including an updated commuting database. This update does not necessarily confer an automatic reduction to the already limited degree of uncertainty, however it is a notable model improvement. APEX now links the CHAD diary drive times with the US census derived commute distances, a significant improvement in better estimating interpersonal variability in exposures occurring while inside a motor vehicle.

3.2.3.2 CHAD activity pattern database

The 2008 REA used a CHAD database containing approximately 23,000 diary days. The most recent version of CHAD master now has well over 50,000 diary days, including a significantly greater number of diaries for children (nearly 18,000 diary days) and the majority of the data are from year 2000 and beyond. Further, additional evaluations performed on the diary data from CHAD and other activity pattern survey data, in particular time spent outdoors and the frequency of outdoor event participation for both healthy individuals and those having asthma, have improved our understanding of the representativeness of the database in capturing the activities that people perform and the locations they visit. See U.S. EPA (2014b), Chapter 5.

3.2.3.3 Air exchange rates

Air exchange rate (AER) data used for estimating indoor residential and other building exposures in the 2008 REA were re-evaluated and updated to a limited degree using recently available residential AER data from the Detroit Exposure and Aerosol Research Study (DEARS) (Williams et al., 2008) and from a field study of 37 small and medium commercial buildings throughout California conducted in 2008 to 2010 (Bennett et al., 2011). See U.S. EPA (2014b), Appendix 5E.

3.2.3.4 Microenvironmental concentrations

The APEX model has additional options available to allow for estimation of more spatially variable microenvironmental concentrations. First, microenvironment location codes are now able to be linked with spatially varying ambient concentration locations (e.g., census tracts, grid points) in the modeling domain. This mapping indicates which set of ambient concentrations

1   are to be used by APEX in calculating microenvironmental concentrations.[55] As part of this

2   significant model improvement, the road concentrations can now be input as a separate ambient

3   air quality file rather than estimated using the ambient concentrations and a distribution of

4   factors to simulate the on-road concentrations.

5   **3.2.3.5 Asthma prevalence**

6   Asthma prevalence has been updated to reflect recently collected data (2006-2010) which

7   are now stratified by U.S. Census tracts using the most recent Census data. First, prevalence data

8   were obtained from the Center for Disease Control (CDC) and Prevention's National Health

9   Interview Survey (NHIS). Briefly, years 2006-2010 NHIS survey data were combined to

10   calculate asthma prevalence, defined as the probability of a "Yes" response to the question "do

11   you still have asthma?" among those that responded "Yes" to the question "has a doctor ever

12   diagnosed you with asthma?" The asthma prevalence was first stratified by NHIS defined

13   regions (Midwest, Northeast, South, and West), sex, age (single years for ages 0-17) or age

14   groups (ages ≥ 18), and a family income/poverty ratio. These new asthma prevalence estimates

15   were then linked to 2010 US Census tract-level poverty ratio probabilities, also stratified by age

16   and age groups, to generate a final database consisting of U.S. Census tract-level asthma

17   prevalence for the entire U.S. See U.S. EPA (2014b) Appendix 5C.

18   **3.3   SUMMARY AND CONCLUSIONS**

19   As described in this chapter, the decision of whether to conduct an updated model-based

20   exposure assessment will be informed by consideration of several factors. The first of these

21   factors will be the results of updated analyses comparing NO_2 air quality with health effect

22   benchmarks (Chapter 2, above). To the extent these analyses indicate little potential for the

23   occurrence of ambient NO_2 concentrations at or above the various 1-hour health effect

24   benchmarks (i.e., indicating little potential for NO_2 exposures of public health concern), there

25   would be limited value to having more refined estimates of personal NO_2 exposures.

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[55] Seven locations are used; "Home" (H), "Work" (W), "Other" (O), "Roadway" (R), "Road near Work" (RW), "Near Home" (NH), and "Near Work" (NW). An 'either' location, "Last" (L), draws from either Near Home or Near Work, depending on the last location the individual was in. A person who is not employed has identical work and home locations. The H and W concentrations are calculated from the air quality data in a person's home and work sectors, respectively. The concentrations in the O location are calculated from a composite of set of air districts. By default, APEX uses the city-average air concentration to calculate O concentrations. If the user specifies roadway air quality districts, then APEX will use these AQ data to determine microenvironmental concentrations for R and RW locations. R is drawn from road concentrations near the Home location, while RW is drawn from road concentrations near the work location. NH is randomly sampled from a tract within a given distance from the H location, while NW is sampled near the work location.

1   Alternatively, to the extent the air quality assessment indicates the potential for NO_2 exposures

2   of public health concern, generating more refined estimates of NO_2 exposures will be considered

3   in the current review. In this latter scenario, conclusions on whether to conduct an updated NO_2

4   exposure assessment will also be informed by 1) the extent to which important uncertainties

5   identified in the last review have been addressed by newly available information, approaches,

6   and tools, and 2) judgments as to the likelihood that an updated quantitative assessment would

7   substantially add to our understanding of NO_2 exposures beyond the insights gained from the

8   exposure assessment conducted in the last review.

9   As discussed in sections 3.2.1 to 3.2.3 above, there have been a number of improvements

10 to key exposure modeling inputs and approaches since the last review (sections 3.2.1 to 3.2.3,

11 above). This new information is substantially different from that used in the 2008 REA and

12 would be likely to appreciably reduce the uncertainties and limitations of the last assessment.

13 Therefore, to the extent health effect benchmark comparisons indicate the potential for the

14 current NAAQS to allow NO_2 exposures of public health concern, we reach the preliminary

15 conclusion that more refined model-based estimates of NO_2 exposures would be supported in the

16 current review.

4 HUMAN HEALTH RISK ASSESSMENT

For some pollutants and health endpoints, there is sufficient scientific evidence and information available to support the development of quantitative estimates of pollutant-related health risks. Depending on the evidence and information available, health risk assessments can be based on information from controlled human exposure studies or on information from epidemiologic studies. In the last review, the 2008 REA conducted a NO_2 human health risk assessment based on information from an epidemiology study.

This chapter presents staff's considerations and preliminary conclusions regarding the information that could inform a potential updated human health risk assessment in the current review. Section 4.1 discusses the extent to which the available evidence and information could support a quantitative risk assessment based on information from controlled human exposure studies. Section 4.2 discusses the extent to which the available evidence and information could support an updated quantitative risk assessment based on information from epidemiology studies.

4.1 RISK ASSESSMENT BASED ON INFORMATION FROM CONTROLLED HUMAN EXPOSURE STUDIES

In some cases, population-level health risks can be estimated using information on exposure-response relationships from controlled human exposure studies, combined with modeled or monitored estimates of personal exposures. This type of risk assessment requires a robust evidence base comprised of controlled human exposure studies that are similar in design (e.g., exposure methods, health effect measurements, study subject characteristics[56]) and that allow for quantification of an exposure-response function (e.g., see REA for ozone, U.S. EPA, 2014).

In the last review of the primary NO_2 NAAQS, a meta-analysis of information from available controlled human exposure studies indicated that exposures to NO_2 concentrations from 100 to 300 ppb could increase airway responsiveness in people with asthma, although the magnitude of that increase was not quantified. As discussed in the 2008 ISA (U.S. EPA, 2008b, section 3.1.3.2), there was considerable variability in methods and results across these studies, and they did not provide a basis for deriving an exposure-response function. A human health risk

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[56] Generalizability of results from controlled human exposure studies to at-risk populations can be limited because the most sensitive individuals (e.g., children, people with severe asthma) are often excluded.

1 assessment based on information from these controlled human exposure studies was not

2 conducted in the 2008 REA (U.S. EPA, 2008a).

3 As discussed above (section 2.2.2), the evidence from controlled human exposure studies

4 has not changed substantially since the last review. The 2nd draft ISA (U.S. EPA, 2015) includes

5 updated meta-analyses of individual-level data from controlled human exposure studies that

6 evaluated the occurrence of NO_2-induced increases in airway responsiveness in people with

7 asthma. In addition to assessing the direction of NO_2-induced changes in airway responsiveness

8 as was done during the last review, the updated meta-analyses also showed that about a quarter

9 of the individuals with asthma exposed at rest to NO_2 experienced a clinically relevant increase

10 in airway responsiveness. These meta-analyses provide evidence supporting the occurrence of

11 increased airway responsiveness in people with asthma following exposures to NO_2

12 concentrations at or above 100 ppb.

13 However, considerable variability in methods and results across these studies precludes

14 their use in deriving an exposure-response function for NO_2-induced changes in airway

15 responsiveness. Specifically, these studies varied in their exposure protocols (e.g., exercise

16 versus rest, exposure durations ranged from 20 minutes to 3 hours), in the approaches used to

17 measure airway responsiveness (e.g., 20% reduction in forced expiratory volume in 1 second

18 (FEV_1), 100% increase in specific airway resistance (sRaw)), in the time of measurement post-

19 exposure (e.g., immediately up to several hours), in their methods for administering

20 bronchoconstricting agents, and in the types of challenges used to induce airway responsiveness

21 (i.e., specific versus non-specific challenge) (U.S. EPA, 2015, section 5.2.2.1). Even within

22 studies that used either specific or non-specific challenge agents, there was considerable

23 variability. Non-specific airway responsiveness was evaluated using carbachol, methacholine,

24 histamine, SO_2, or cold air as challenge agents. Specific airway responsiveness was evaluated

25 using ragweed, house dust mite, birch, timothy, or cat allergen as challenge agents (U.S. EPA,

26 2015, Tables 5-2 and 5-3).

27 Results are highly variable across these studies. The available information does not

28 demonstrate an exposure-dependent response and, therefore, this information is not sufficient to

29 support the derivation of an exposure-response function for use in quantitative estimates of NO_2

30 health risks. Goodman et al. (2009) reached a similar conclusion, based on meta-analyses and

31 meta-regressions of information from studies of NO_2-induced specific and non-specific airway

32 responsiveness. In addition, there is not strong evidence of an exposure-response relationship in

33 individual studies that evaluated exposures to multiple NO_2 concentrations (Bylin et al., 1988;

34 Orehek et al., 1976). Therefore, while the available information is sufficient to support the

35 identification of health effect benchmarks for NO_2, as described above (section 2.2.2), we reach

36 the preliminary conclusion that a quantitative risk assessment based on information from

1 controlled human exposure studies is not supported by the evidence available in the current
2 review.

4.2 RISK ASSESSMENT BASED ON INFORMATION FROM EPIDEMIOLOGY STUDIES

5   Risk estimates based on epidemiologic studies have the potential to provide perspective
6 on the most serious pollutant-associated public health risks (e.g., hospital admissions, emergency
7 department (ED) visits, premature mortality) in populations that often include at-risk groups.
8 However, the amount of emphasis given to such quantitative risk estimates depends on the extent
9 to which the underlying epidemiologic studies address key uncertainties, including the potential
10 for confounding by co-occurring pollutants. This section describes the epidemiology-based risk
11 assessment conducted in the 2008 REA (section 4.2.1) and staff's consideration of the relevant
12 evidence and information that is available in the current review (section 4.2.2).

4.2.1 Overview of the Assessment in the Last Review

14   In the last review of the primary NO_2 NAAQS, respiratory-related ED visits in the
15 Atlanta MSA were estimated as a function of short-term ambient NO_2 concentrations, based on
16 concentration-response relationships from an epidemiologic study by Tolbert et al. (2007) (U.S.
17 EPA, 2008a, Chapter 9).[57] Specifically, the 2008 REA modeled respiratory-related ED visits
18 (including asthma, chronic obstructive pulmonary disease (COPD), upper respiratory illness,
19 pneumonia and bronchiolitis) for individuals of all ages based on a 3-day moving average of the
20 daily maximum 1-hour NO_2 concentrations measured at a single central-site monitor.[58] The
21 selection of the Tolbert et al. (2007) study as the basis for risk modeling reflected an emphasis on
22 (1) studies conducted within the U.S.; (2) studies of ambient NO_2 exposure (rather than indoor
23 exposure); (3) studies of respiratory-related ED visits or hospital admissions (given the clear
24 public health significance of this endpoint compared to symptoms and the degree of supporting

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[57] As discussed above (section 1.2), the strongest evidence in the last review was for respiratory effects attributable to short-term NO_2 exposures. The study by Tolbert et al. (2007) was a key study supporting the relationship between NO_2 and respiratory effects (U.S. EPA, 2008b, Chapter 5).

[58] The monitor used in generating risk estimates (monitor id 131210048) matches that used in the Tolbert et al. (2007) study.

1   evidence presented in the 2008 ISA); and (4) studies that provided both single- and co-pollutant

2   concentration-response functions.[59]

3   The REA presented incidence estimates associated with NO_2 concentrations adjusted to

4   just meet the annual NO_2 standard with its level of 53 ppb, NO_2 concentrations adjusted to just

5   meet potential alternative 1-hour NO_2 standards with levels ranging from 50 to 200 ppb (98th and

6   99th percentile forms),[60] and unadjusted NO_2 concentrations. As discussed above (section 2.1),

7   air quality adjustments were based on a proportional roll up of monitored NO_2 concentrations.

8   **4.2.1.1 Summary of the 2008 REA epidemiology-based risk assessment results**

9   The 2008 REA presented risk estimates based on single-pollutant models and co-pollutant

10   models (U.S. EPA, 2008a, Chapter 9). For adjusted air quality, the 2008 REA noted the

11   following:

12   • When air quality was adjusted to simulate just meeting the existing annual standard, about 8
13   to 9% of respiratory-related ED visits in the Atlanta MSA were estimated to be attributable to
14   short-term NO_2 exposures, based on a single-pollutant model. Risk estimates based on co-
15   pollutant models remained positive, though they were smaller and confidence intervals were
16   wider than estimates based on the single pollutant model. Co-pollutant models that included
17   another roadway-associated pollutant (i.e., CO) resulted in modest reductions in NO_2 risk
18   estimates (i.e., about 7 to 8% of respiratory ED visits estimated to be associated with NO_2).
19   The smallest risks were estimated with a co-pollutant model that also included PM_{10} (i.e.,
20   about 3% of respiratory ED visits estimated to be associated with NO_2) and with a multi-
21   pollutant model that included both PM_{10} and O_3 (U.S. EPA, 2008a, Tables 9-3 and 9-4).

22   • When air quality was adjusted to simulate just meeting potential alternative standards with 1-
23   hour averaging times, standards with levels of 50, 100, and 150 ppb reduced estimated NO_2-
24   associated risks compared to the annual standard alone. When air quality was adjusted to
25   simulate just meeting a potential alternative standard with a 1-hour averaging time and a
26   level of 200 ppb, estimated risks were similar to those estimated for the annual standard (U.S.
27   EPA, 2008a, Tables 9-3 and 9-4).

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[59] The 2008 REA reflected the ISA conclusion that an important uncertainty in the NO_2 epidemiologic evidence is the extent to
which NO_2 is independently associated with [short-term] respiratory effects or if NO_2 is a marker for the effects of another
traffic-related pollutant or mix of pollutants (U.S. EPA, 2008b, section 4.2.8, p. 37). This highlights the importance of including
both single- and co-pollutant functions in modeling emergency department visits for NO_2.

[60] These were the same air quality scenarios evaluated for the air quality and exposure assessments (sections 2.1 and 3.1, above).

4.2.1.2 Uncertainties and limitations

In the last review, the 2008 REA noted that a number of key uncertainties should be considered when interpreting these results with regard to decisions on the standard. These included the following (U.S. EPA, 2008a, section 9.6):

- Uncertainties in the estimates of NO_2 coefficients in concentration-response functions used in the assessment.
- Uncertainties concerning the specification of the concentration-response model (including the shape of the relationships) and whether or not a population threshold exists within the range of concentrations examined in the studies.
- Uncertainty concerning possible confounding by co-occurring pollutants.
- Uncertainty in the adjustment of air quality distributions to simulate just meeting various standards. The REA recognized this as an important uncertainty, especially for scenarios where considerable upward adjustment was required to simulate just meeting some of the standards.

4.2.2 Consideration of Newly Available Information

The decision whether to conduct an updated epidemiology-based risk assessment in the current review will be informed by staff's conclusions regarding the extent to which such an updated assessment would reduce uncertainties and substantially improve our understanding of NO_2-attributable health risks, beyond the insights gained from the risk assessment conducted in the last review. As discussed above (section 1.3), the strongest evidence in the current review, based on the weight-of-evidence determinations in the 2nd draft ISA, is for respiratory effects attributable to either short- or long-term NO_2 exposures (U.S. EPA, 2015, section 1.5.1). In this section, staff considers the evidence that could inform an assessment of effects attributable to short-term (4.2.2.1) or long-term (4.2.2.2) NO_2 exposures.

4.2.2.1 Health effects associated with short-term exposures

For short-term NO_2 exposures, we first consider the overall strength of the scientific evidence for NO_2-attributable health endpoints and at-risk populations compared to the evidence in the last review, as presented in the 2nd draft ISA (U.S. EPA, 2015). As discussed above (section 1.3), the 2nd draft ISA concludes that the evidence supports "a causal relationship between short-term NO_2 exposure and respiratory effects" and that the "strongest evidence is for effects on asthma exacerbation" (U.S. EPA, 2015, Table 1-1). These conclusions are based on a number of epidemiologic, controlled human exposure, and animal toxicological studies, which together describe a "coherent and biologically plausible pathway by which NO_2 exposure can trigger an asthma exacerbation" (U.S. EPA, 2015, p. 1-17).

In comparing the evidence for NO_2-attributable respiratory effects available in the current review to that available in the last review, the ISA notes the following (U.S. EPA, 2015, p. 1-19):

1   Much of the evidence from epidemiologic and experimental studies was available
2   in the 2008 ISA. However, compared to the 2008 ISA, this ISA more explicitly
3   evaluates the coherence and biological plausibility for specific respiratory
4   outcome groups. Rather than new evidence, the integration of epidemiologic and
5   experimental evidence for asthma exacerbation—uptake of NO_2 in the respiratory
6   tract and reactions to form reactive oxidation products, allergic inflammation,
7   airway responsiveness, asthma symptoms, and hospital admissions and ED visits
8   for asthma, associations with NO_2 measured in people's locations, which may
9   better represent exposure, associations with adjustment for another traffic-related
10   pollutant—describes a coherent, biologically plausible pathway to support a
11   causal relationship between short-term NO_2 exposure and respiratory effects.

12 Thus, the change in the causality determination[61] for respiratory effects attributable to short-term

13 NO_2 exposures is largely due to the evolution of the ISA's approach to assessing the evidence,

14 rather than the availability of substantially different evidence in the current review.

15   The evidence that has become available since the last review has not substantially

16 changed our understanding of health effects attributable to short-term NO_2 exposures or of the

17 populations potentially at increased risk of such effects.[62] Updated risk estimates based on

18 information from epidemiology studies in the current review would be subject to the same

19 uncertainties identified in the 2008 REA.

20   In particular, recent studies do not provide an improved basis, compared to the last

21 review, for quantifying NO_2-attributable risks independent of other roadway-associated

22 pollutants (e.g., carbon monoxide, particulate matter, elemental carbon, and volatile organic

23 compounds). Table 1-1 of the 2nd draft ISA (U.S. EPA, 2015) concludes that an important

24 uncertainty in the current review continues to be the "[s]trength of inference from co-pollutant

25 models about independent associations of NO_2, especially with pollutants measured at central

26 site monitors" (U.S. EPA, 2015). In particular, of the key studies supporting the causal

27 relationship with respiratory effects (U.S. EPA, 2015, Table 5-45), two U.S. studies evaluating

28 asthma-related hospital admissions or emergency department visits have become available since

[61] In the previous review, the ISA concluded that the evidence was "sufficient to infer a likely causal relationship between short-term NO_2 exposure and adverse effects on the respiratory system" (U.S. EPA, 2008b, section 5.3.2.1).

[62] Though the ISA's framework for identifying at-risk populations (U.S. EPA, 2015, section 1.6.5) was developed since the last review of the NO_2 NAAQS, the 2008 ISA identified people with asthma, children, and older adults as populations potentially at increased risk for NO_2-related effects (U.S. EPA, 2008b, section 4.3.1 and 4.3.2). As discussed above (section 1.3), in the current review the 2nd draft ISA concludes that "there is adequate evidence that people with asthma, children, and older adults are at increased risk for NO_2-related health effects" (U.S. EPA, 2015, Table 7-26; section 1.6.5).

1   the last review (Strickland et al., 2010; Li et al., 2012). Neither of these studies reported NO_2

2   health effect associations in co-pollutant models that included other roadway-related pollutants.

3   Based on the above considerations, an updated epidemiology-based risk assessment

4   estimating respiratory-related endpoints attributable to short-term NO_2 exposures would be

5   subject to uncertainties that are essentially the same as those identified in the 2008 REA (U.S.

6   EPA, 2008). We reach the preliminary conclusion that such an updated epidemiology-based risk

7   assessment in the current review would not appreciably reduce uncertainties and limitations from

8   the assessment conducting in the last review and would be unlikely to substantially improve our

9   understanding of NO_2-attributable health risks or increase our confidence in risk estimates

10   beyond the assessment from the last review.

11   **4.2.2.2 Health effects associated with long-term exposures**

12   As discussed above (section 1.3), the 2nd draft ISA concludes that the evidence "indicates

13   there is likely to be a causal relationship between long-term NO_2 exposure and respiratory

14   effects" (U.S. EPA, 2015, section 1.5.1, pp. 1-19 and 1-21) and the "strongest evidence is for

15   effects on asthma development" (U.S. EPA, 2015, Table 1-1). This contrasts with the conclusion

16   from the 2008 ISA that the evidence at that time was "suggestive but not sufficient to infer a

17   causal relationship" between long-term NO_2 exposures and respiratory effects. Key evidence

18   supporting the change to this causal determination comes from recent epidemiologic cohort

19   studies reporting associations between long-term ambient NO_2 concentrations (i.e., averaged

20   over 1−10 years) and development of asthma in children. There is some support for the

21   biological plausibility of effects attributable to long-term exposures provided by "a small body of

22   experimental studies" (U.S. EPA, 2015, Table 1-1).

23   As for short-term NO_2 exposures, an important issue in considering a potential

24   quantitative risk assessment is the extent to which available epidemiologic studies report health

25   effect associations with long-term NO_2 in co-pollutant models, specifically for traffic-related

26   pollutants. This is an even more important issue for long-term NO_2 exposures, given the higher

27   correlations between long-term NO_2 concentrations and other pollutants reported in many

28   epidemiologic studies (U.S. EPA, 2015, Table 6-1).[63]

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[63] In studies of NO_2-associated asthma development, correlations with co-occurring pollutants were most often reported for PM. NO_2 and PM were often highly correlated in these studies, particularly in studies that used land use regression or dispersion modeling to estimate long-term NO_2 exposures (e.g., r values were greater than 0.9 in several studies) (U.S. EPA, 2015, Table 6-1).

1   Of the key studies evaluating associations between long-term NO_2 and the development
2   of asthma (U.S. EPA, 2015, Table 6-5), none evaluated associations in co-pollutant models for
3   traffic-related pollutants. Table 6-5 of the 2nd draft ISA notes that an important remaining
4   uncertainty is that "regarding potential confounding by traffic-related co-pollutants" (U.S. EPA,
5   2015). In particular, the 2nd draft ISA notes that "[w]hen reported, correlations with $PM_{2.5}$ and
6   EC often were high ($r = 0.7-0.96$)" and that "[n]o co-pollutant models [were] analyzed" (U.S.
7   EPA, 2015, Table 6-5).
8   Although there are new epidemiologic studies available in the current review supporting
9   a relationship between long-term NO_2 exposures and development of asthma in children, a
10   quantitative risk assessment based on information from such studies would be subject to
11   considerable uncertainty due to the inability to distinguish the contributions of NO_2 from the
12   contributions of other highly correlated pollutants. Given these limitations, we reach the
13   preliminary conclusion that such a risk assessment would not substantially add to our
14   understanding of NO_2-attributable health risks and would therefore be of limited value in
15   informing decisions in the current review.

5 SUMMARY OF CONCLUSIONS AND NEXT STEPS

The preliminary conclusions presented in chapters 2 through 4 of this planning document reflect the EPA staff's preliminary assessment of the degree to which updated quantitative analyses in the current review of the primary NO_2 NAAQS are likely to substantially add to our understanding of NO_2 exposures or health risks. In developing these preliminary conclusions, staff considered a variety of factors (Figure 1-3) including the availability of appropriate health evidence; the availability of technical information, tools, and methods; and judgments as to the potential for particular quantitative assessments to provide important insights into exposures or health risks, beyond the insights gained from previous assessments. This chapter summarizes staff's preliminary conclusions (section 5.1) and discusses the next steps in the review of the primary NO_2 NAAQS (section 5.2).

5.1 SUMMARY OF PRELIMINARY CONCLUSIONS

Summaries of staff's preliminary conclusions are as follows:

- Air quality comparison to health benchmarks: New information from the NO_2 monitoring network and from available research studies has the potential to substantially improve our understanding of NO_2 concentrations around major roads in the current review. Staff concludes that updated analyses comparing ambient NO_2 concentrations (i.e., as surrogates for potential exposure concentrations) to health effect benchmarks would better characterize a key uncertainty from the last review (i.e., uncertainty in ambient NO_2 concentrations on- or near-roads).

- Exposure assessment: While modeling tools have been updated since the last review, staff reaches the preliminary conclusion that an updated exposure assessment would be warranted only if the air quality assessment discussed in Chapter 2 indicates the potential for NO_2 exposures that could be of public health concern (i.e., based on comparisons of ambient NO_2 concentrations with health effect benchmarks). If the air quality assessment indicates little potential for such exposures, including on or near major roads, staff reaches the preliminary conclusion that an updated assessment of population exposures would be of limited use in informing decisions in the current review.

- Risk assessment based on information from controlled human exposure studies: Based on the evidence assessed in the 2nd draft ISA, staff reaches the preliminary conclusion that available studies do not provide information to support the identification of an NO_2 exposure-response relationship with relevant health endpoints and at relevant NO_2 concentrations. Therefore, as in the last review, staff reaches the preliminary conclusion that the available evidence in the current review is not sufficient to support a risk assessment based on exposure-response information from controlled human exposure studies.

- Risk assessment based on information from epidemiologic studies of health effects associated with short-term NO_2 exposure: Evidence that has become available since the

1   last review has not substantially changed our understanding of the health effects
2   attributable to short-term NO_2 exposures or our understanding of the populations at
3   increased risk from such exposures. Recent U.S. epidemiology studies of asthma-related
4   hospital admissions or emergency department visits also have not provided information
5   on NO_2 effects, independent of other traffic-related pollutants. Therefore, an updated
6   epidemiology-based risk assessment estimating respiratory-related endpoints attributable
7   to short-term NO_2 exposures would be subject to uncertainties that are essentially the
8   same as those identified in the 2008 REA. We reach the preliminary conclusion that such
9   an updated epidemiology-based risk assessment in the current review would be unlikely
10   to substantially improve our understanding of NO_2-attributable health risks or increase
11   our confidence in risk estimates, beyond the assessment from the last review.

12   •   Risk assessment based on information from epidemiologic studies of health effects
13   associated with long-term NO_2 exposure: Key U.S. epidemiology studies of long-term
14   NO_2 and asthma incidence do not present analyses with co-pollutant models that include
15   highly correlated traffic-related pollutants. A risk assessment quantifying the
16   development of asthma attributable to long-term NO_2 exposures would be subject to
17   considerable uncertainty due to the inability to distinguish the contributions of NO_2 from
18   the contributions of other pollutants. Therefore, we reach the preliminary conclusion that
19   such a risk assessment would be of limited value in informing decisions in the current
20   review.

5.2 NEXT STEPS

22   Given the preliminary conclusions summarized above, the next step in the current review

23 will be for staff to conduct the analyses comparing NO_2 air quality concentrations to health effect

24 benchmarks, as described in Chapter 2 of this planning document. Based on the results of these

25 analyses, staff will consider the potential for populations in U.S. urban areas just meeting the

26 existing primary standards to experience NO_2 exposures that may be of concern in people with

27 asthma. In doing so, staff's considerations will focus on the frequency with which ambient NO_2

28 concentrations in the U.S. could be at or above various health effect benchmarks.

29   The results of these analyses will inform the subsequent steps appropriate for the current

30 review. If results indicate that the existing standards allow the potential for exposures to NO_2

31 concentrations that may be of concern for public health, more refined quantitative analyses of

32 personal NO_2 exposures will be considered. If results indicate limited potential for such NO_2

33 exposures, there will be less support for a more refined quantitative assessment of personal NO_2

34 exposures.

35   To the extent analyses are limited to comparisons between ambient NO_2 concentrations

36 and health effect benchmarks, we anticipate that these analyses, as well as the evidence and

37 rationale supporting a conclusion not to conduct more refined analyses of personal NO_2

38 exposures, will be incorporated into the first draft of the Policy Assessment (PA). Under this

39 potential scenario, a separate REA will not be generated and CASAC will review the analyses

40 comparing ambient NO_2 concentrations to health effect benchmarks, including staff's

1 interpretation of the results of these analyses, as part of its review of the PA. Based on this

2 potential scenario, Table 5-1 presents a tentative schedule for the documents remaining to be

3 produced for this review. To the extent the tentative schedule presented in Table 5-1 requires

4 modification (e.g., to accommodate an exposure assessment), a revised schedule will be

5 communicated to CASAC and the public.

6 **Table 5-1. Tentative schedule for next steps in the review of the primary NO$_2$ NAAQS.**

Stage of Review	Major Milestone	Target Date
Integrated Review Plan (IRP)	Final IRP	June 2014
Integrated Science Assessment (ISA)	1st draft ISA	November 2013
	CASAC public meeting for review of the 1st draft ISA	March 12-13, 2014
	2nd draft ISA	January 2015
	CASAC review of the 2nd draft ISA	June 2-3, 2015
	Final ISA	Fall 2015
Risk/Exposure Assessment (REA)	REA Planning Document	May 4, 2015
	CASAC review of REA Planning Document	June 2-3, 2015
Policy Assessment (PA) including quantitative analyses comparing ambient NO$_2$ concentrations to health effect benchmarks	1st draft PA	Spring/Summer 2016

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APPENDIX A: ANALYSIS OF LAS VEGAS NEAR-ROAD NO$_2$ MEASUREMENT DATA AND LOGIT MODEL DEVELOPED TO SIMULATE NO$_2$ ON-ROAD CONCENTRATIONS

This appendix provides the memorandum from Jennifer Richmond-Bryant to Stephen Graham regarding an approach to use in simulating on-road NO$_2$ concentrations from near-road NO$_2$ concentrations.

List of Figures

United States Environmental Protection Agency
National Center for Environmental Assessment
Research Triangle Park, North Carolina 27711

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5   **MEMORANDUM**

6

To: Stephen Graham – Physical Scientist, Risk and Benefits Group, OAQPS

From: Jennifer Richmond-Bryant – Senior Physical Scientist, National Center
               for Environmental Assessment, ORD

Date: May 4, 2015

Subject: Analysis of Las Vegas Near-Road NO_2 Measurement Data and Logit
               Model Developed Simulate NO_2 On-Road Concentrations.

7

8   **A-1.   Overview**
9   This memo summarizes analyses performed on recently collected near-road measurement data in Las
10 Vegas, NV. Several statistical models were evaluated to describe the pattern in the concentration
11 reduction observed with increasing distance from a major road. Based on model fits (R^2) and overall
12 form, a logit function was determined most appropriate. Proposed is an approach to use in estimating
13 on-road NO_2 concentrations when having NO_2 concentration measurements at a distance from a road,
14 particularly at locations in close proximity to a road (i.e., 10 m and 20 m).

15

16 **A-2.   Methods**
17 *Study Area*

18 Near-road measurements of air quality, traffic, and meteorology were obtained at sites in Las Vegas, NV.
19 The near road monitoring study area was chosen where 1) the Annual Average Daily Traffic (AADT)
20 exceeded 150,000 vehicles per day, 2) airflow downwind of the highway was not restricted by natural or
21 human-made structures, and 3) state and local governments permitted sampling sites within 300 meters
22 (m) of the road to be established. The study area and sampling sites are shown in Figure A-1.

1

Figure A-1. The Las Vegas study area. Interstate-15 runs north-south, and the monitoring sites follow along a northwest-to-southeast transect along a road crossing.

The Las Vegas study area was located adjacent to Interstate-15 (I-15). Along this segment of the road, AADT is approximately 206,000 vehicles per day, with 10% of those characterized as heavy-duty diesel trucks (Nevada DOT, 2006). At this location, I-15 runs in the north-south direction, and the highway sits below grade with walls sloping upwards from the road at 20° angles. The terrain above the embankment is flat within a 10 km radius of the road. Downwind sampling sites were located approximately 20 m, 100 m, and 300 m east of the highway, and an upwind site was placed approximately 100 m west of the road. Meteorology at this study area is generally characterized as arid, with hot summers and sunshine throughout the year. With mountain ranges surrounding the Las Vegas metropolitan area, the area is subject to atmospheric inversions. A detailed description of this study area is provided in Kimbrough et al. (2013).

Data Collection

Nitric oxide (NO) and total oxides of nitrogen (NO_X) were monitored continuously by chemiluminescence with a trace oxides of nitrogen analyzer (Ecotech, Model EC 9841 B, Knoxfield, VIC, Australia) with measurements averaged every five minutes, and NO_2 was estimated via differencing. Multipoint calibration was performed at the beginning of the study, and zero and span checks were performed

A-2

1   nightly for each of the gaseous monitors. Inlets for each of these monitors were placed approximately 3
2   m above ground, and air pollutant concentrations were measured at the 20 m, 100 m, and 300 m
3   downwind sites.

4   Surface meteorological parameters monitored included wind speed, wind direction, air temperature,
5   relative humidity, precipitation, and solar radiation. All meteorological parameters were measured at
6   the Las Vegas airport (LAS), as part of the standard meteorological measurements made by the National
7   Weather Service (NWS) at most major airports. The LAS meteorological station, which is approximately
8   1.5 km from the Las Vegas near-road site, is part of the Automated Surface Observing Systems (ASOS)
9   and has a 1-minute temporal resolution for wind speed and direction. Upper air data from the Universal
10   Rawinsonde Observation (RAOB) station in Mercury/Desert Rock, NV (KDRA, elevation 1006 m) was
11   used as the primary upper air station. 1-Minute ASOS wind data were processed for input to AERMET
12   using AERMINUTE for the period 12/1/2008 through 2/28/2010. Upper air and surface data were
13   processed through AERMET to obtain hourly averages of surface and upper air meteorological
14   parameters.

15   Data were collected at the Las Vegas site between December 12, 2008 and January 21, 2010. There were
16   8,466 complete hours (81.8%) of data (based on the presence of NO_2 data at all three downwind sites)
17   available for that time period and used in the analyses presented here.

18   *On-Road Concentration Estimation*

19   In order to estimate the on-road concentration of NO_2 at each hour where measurement data were
20   complete, a statistical distribution was fit to the concentrations measured at the three downwind
21   monitoring locations. All models were of the basic form:

22   f(C) = m*g(x) + b   (equation 1)

23   where f(C) = a statistical distribution fit to the concentration data across the three monitoring sites, g(x)
24   = a statistical distribution fit to the location of the monitors across the three sites, m = the model
25   estimated slope, and b = the model estimated intercept. The slope for any given assumption is the first
26   derivative of the concentration function with respect to space, m = df(C)/dg(x). Linear, ln-ln, ln-linear,
27   and logit-ln distributions were fit to the concentration data across the three monitoring sites. Table A-1
28   presents the forms of f(C) and g(x) for different assumptions about the model distribution. The
29   dependent and independent terms increase in complexity. A lognormal distribution was fit to the
30   distance data in some of the models to linearize that term, since the monitor sites were not evenly
31   spaced relative to one another. Likewise, a lognormal distribution was fit to the concentrations in some
32   of the models to linearize that term for the same reason. If a ln-ln model were used, then the model
33   could not be solved for an on-road concentration (at a distance of x = 0), because the declining on-road
34   concentration would produce a negative slope, leading to a solution of infinite concentration when
35   integrating the derivative to solve for f(C) at x = 0. A logit function was fit to the concentrations for the
36   logit-ln model to test if the concentration distribution approximated an S-shaped curve. Use of the logit-
37   ln model was most physically sensible, because turbulent mixing related to traffic in some instances
38   could cause concentration levels to plateau near the road and then gradually drop off in a manner

1   similar to a Gaussian distribution (centerline to lateral extrema). Note that the reference concentration

2   was C(300), because the reference point had to be along the data distribution for the functional fit to

3   apply. The reference was considered a point downstream where concentration returned to background

4   levels. Using the 100 m upstream site as the reference point would have produced a mathematical

5   instability.

6   **Table A-1. Concentration vs. distance-from-road model formulations.**

MODEL FORM	f(C)	g(x)	C(x)
linear	C	x	b
ln-linear	ln(C)	x	exp(b)*exp(m*x)
ln-ln	ln(C)	ln(x)	exp(b)*x^m
logit-ln	e$^{C(x)}$/[e$^{C(x)}$+e$^{C(ref)}$]	ln(x)	C(ref)+ln{b+mln(x)}-ln{1- [b+mln(x)]}

7

8   When evaluating the performance of the selected models, the logit-ln formulation had the highest

9   median R^2 of all model types (R^2 = 94.7% compared with 84.6%, 93.7%, and 88.2% for the linear, ln-ln,

10   and ln-linear models, respectively). The median, average, and percentile statistics were calculated for R^2

11   across the data, since there were 8,466 curve fits corresponding to each complete time period. These

12   results support use of the logit-ln model. In conjunction with the physical rationale described above, the

13   logit-ln model was used to estimate on-road concentrations.

14

15   ## A-3.   Results and Discussion

16   The on-road NO_2 concentrations were estimated for five scenarios: 1) all wind and stability conditions

17   combined, 2) winds from the west (210°-330°, where the monitors were downwind of the highway), 3)

18   winds from the east (30°-150°, where the monitors were upwind of the highway), 4) inversion

19   conditions (convective mixing height less than 300 m), and 5) non-inversion conditions (convective

20   mixing height greater than 300 m). Summary statistics for each scenario are provided in Table A-2

21   through Table A-6, and predicted and observed data distributions are displayed in Figure A-2 through

22   Figure A-6. In addition, limited results for NO_X concentrations are included in the text for added context

23   and discussion clarity. For each scenario, estimated NO_2 concentrations derived from the logit-ln model

24   are presented for the on-road, 10 m, 20 m, 30 m, 40 m, 50 m, 100 m, and 300 m sites. Observation data

25   are also presented so that estimates for the 20 m, 100 m, and 300 m sites could be compared with

26   observations for validation. Model estimates were within 1% of observations. Good agreement between

27   observations and predictions is also indicated when examining the figures for each scenario, because

28   the median and range of the observations (given by the second to ninety-eighth percentile of the data)

29   coincide with the median and range of the model predictions.

30   For all study area conditions combined, the average and maximum on-road NO_2 concentrations were 31

31   ppb and 104 ppb, respectively (Table A-2). Despite the use of the logit function to represent the

1   concentration variable, Figure A-2 (as well as the other scenarios, Figure A-3 through Figure A-6)

2   illustrates that the concentration plateau with decreasing distance to the road is very limited in size.

3   Overall, the concentration trend with increasing distance from the roads is consistent with

4   exponential/logarithmic decay functions described in other similar measurement studies (section

5   2.5.3.1, US EPA, 2015). NO_2 concentrations declined from on-road to the 10 m site by 13%, on average,

6   and from on-road to the 20 m site by an average of 17% for (Table A-7). A reduction in NO_X at the 20 m

7   site on average was 25% (not shown). When stratifying the percent change in concentration by quintile

8   of concentration at the 10 m or 20 m location, the highest average difference between on-road and the

9   10 m or 20 m site is 19% and 25%, respectively (Table A-7), and occurs in the second quintile (i.e. at low

10   NO_2 concentrations). Some predictions in the first quintile are negative at the 10 m site or produce

11   higher values at the 10 m or 20 m sites compared with the on-road value (perhaps driven by inclusion of

12   concentration data for where predominant winds were from the east). This effectively reduced the

13   average difference between on-road and away from road concentrations considering the lowest

14   concentration quintile. At higher NO_2 concentrations (i.e., quintile 4 and 5), the percent change

15   decreased to 10-11% for the 10 m site and 11-12% for the 20 m site, on average (Table A-7).

16   When segregating the study area data set by wind direction, on-road concentration estimates were

17   generally higher for winds from the west (Table A-3, average NO_2: 36 ppb, max NO_2: 74 ppb, average

18   NO_X: 86 ppb, max NO_X: 264 ppb) compared with concentrations when winds were from the east (Table

19   A-4, average NO_2: 23 ppb, max NO_2: 75 ppb, average NO_X: 38 ppb, max NO_X: 215 ppb). The estimated

20   near-road gradient was slightly sharper for winds from the west, with an average on-road to 20 m

21   difference of 17% for NO_2 (Table A-8) and 28% for NO_X (not shown) for winds from the west and an

22   average difference of 13% for NO_2 (Table A-9) and 14% for NO_X (not shown) for winds from the east.

23   Interestingly, the average estimated NO_X gradient (not shown) was twice as large for winds from the

24   west compared with winds from the east, but the difference in gradient was much smaller for NO_2. In

25   general, the NO gradient tends to be sharper than the NO_2 gradient (Karner et al., 2010). This finding

26   suggests that when the winds from the west disperse the NO away from the roadway, photochemistry

27   occurs over shorter time and length scales such that the NO disappears faster than the NO_2 compared

28   with the case where the monitors are on the upwind side of the road. At lower NO_2 concentrations, the

29   difference between on-road and the 10 m or 20 m sites was much larger when winds were from the

30   west compared with the difference for winds from the east. For example, at 10 m, the difference was

31   10% when winds were from the east and 25% when winds were from the west (see Tables A-8 and A-9).

32   For the three highest concentration quintiles the percent difference was comparable regardless of wind

33   direction (8-10% at the 10 m site, 10-12% at the 20 m site).

34   When segregating the study area data set by stability conditions, concentrations were on average higher

35   during an inversion (Table A-5, average NO_2: 38 ppb, max NO_2: 79 ppb, average NO_X: 86 ppb, max NO_X:

36   264 ppb) compared with non-inversion conditions (Table A-6, average NO_2: 23 ppb, max NO_2: 82 ppb,

37   average NO_X: 102 ppb, max NO_X: 225 ppb). The estimated near-road gradient was slightly sharper for

38   non-inversion conditions, with an average on-road to 20 m difference of 22% for NO_2 (Table A-11) and

39   26% for NO_X (not shown) for non-inversion conditions and an average difference of 16% for NO_2 (Table

40   A-10) and 17% for NO_X (not shown) for inversion conditions. Across the second to fifth quintile for

1   concentration, the differences were roughly 20-30% higher for the non-inversion conditions compared
2   with the inversion conditions when considering the on-road to 20 m comparison. This makes sense given
3   that inversion quells convective mixing in the atmosphere. At the lowest concentrations, the
4   relationship between differences and inversion conditions seems to reverse.

5   This analysis is limited by the assumption that the logit model is appropriate for every atmospheric
6   condition included in the data set. Model performance, as measured by R^2, varies somewhat by
7   atmospheric condition but is reasonably high. For example, median R^2 decreases slightly from 96% to
8   92% when stratifying by winds from the west vs. winds from the east. Median R^2 declines from 93% to
9   83% when stratifying by non-inversion vs. inversion conditions. Furthermore, model evaluations
10   (predicted vs. observed concentrations) did not vary by atmospheric condition. Overall, the logit model
11   still appears to be a reasonable choice to fit the data. Additionally, discrepancies observed between the
12   summary statistics overall and when considering particular atmospheric conditions might be attributed
13   to an absence of meteorological data at some times of day when concentrations were measured. For
14   example, only 907 hours of concentration data were used in developing the model for inversion
15   conditions, and 4,444 hours of concentration data were used in developing the model for non-inversion
16   conditions compared with 8,466 hours of data overall. More specifically in this instance, the maximum
17   on-road NO_2 estimate of 104 ppb in the overall dataset is not found in any of the stratifications. Another
18   limitation of this work is that this analysis was performed only for the Las Vegas study area. This study
19   area had limited influence from sources other than those originating from the roadway. This relationship
20   would not necessarily be representative for many urban sites with multiple sources including, for
21   example, emissions from additional arterial roads or combustion-related power plants. However, this
22   work provides important insight about NO_2 concentration changes from a single highway.

23   From a practical perspective, this analysis can shed light on how well the existing near-road monitoring
24   network may be useful in understanding on-road NO_2 concentrations. Greater similarity between the
25   percent differences during inversion conditions and differences across all wind directions suggests that
26   inversions are a prevalent feature of the meteorology in Las Vegas. Hence, estimates of 10-15%
27   reductions in NO_2 concentration at the 10 m site and 10-20% at the 20 m site might be reasonable in
28   many cases for regions where inversions tend to occur. Where away from road concentrations are low
29   (i.e., lower percentiles of the overall concentration distribution), reductions of 15-25% at the 10 m site
30   and 20-35% at the 20 m site might be more reasonable. In areas where inversions do not tend to occur,
31   the non-inversion conditions may be more typical, with differences of 15-20% at the 10 m site and
32   differences of 20-30% at the 20 m site, though keep in mind, NO_2 concentrations are generally lower for
33   this scenario. The selection of a higher gradient and applied equally to all possible atmospheric
34   conditions or concentration levels would tend to produce a more conservative estimate of on-road NO_2
35   concentrations. However, since the maximum estimated on-road NO_2 concentration was 104 ppb, if
36   assuming a steeper gradient produced NO_2 concentrations above this value, then the modeler would
37   have to question the validity of the gradient assumption.

38

A-4.   References

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http://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=288043.

Table A-1. Summary statistics for observed and predicted concentrations for all wind and stability conditions combined.

Distance	OBSERVATIONS				MODEL PREDICTED						
	20m	100m	300m		10m	20m	30m	40m	50m	100m	300m
NO$_2$ (ppb)											
Avg	24.86	22.65	19.91		26.66	25.33	24.55	24.00	23.57	22.24	20.14
Stdev	12.27	12.22	12.35		12.48	12.24	12.16	12.12	12.11	12.14	12.42
5%	7.20	5.98	4.44		7.78	7.44	7.19	6.93	6.76	6.06	4.48
25%	14.29	11.46	8.40		16.18	14.80	14.00	13.34	12.86	11.26	8.49
50%	23.74	21.48	17.46		26.43	24.54	23.44	22.81	22.22	20.57	17.94
75%	34.85	32.95	30.94		36.46	35.24	34.59	34.16	33.80	32.72	31.23
95%	44.94	42.46	40.10		47.10	45.33	44.47	43.90	43.46	42.17	40.40
98%	49.20	46.08	44.11		51.73	49.57	48.45	47.77	47.27	46.00	44.26

Table A-2. Summary statistics for observed and predicted concentrations, westerly winds (210°-330°).

Distance	OBSERVATIONS				MODEL PREDICTED						
	20m	100m	300m		10m	20m	30m	40m	50m	100m	300m
NO$_2$ (ppb)											
Avg	29.93	27.16	24.65		31.28	29.94	29.16	28.61	28.18	26.84	24.73
Stdev	11.40	11.73	12.54		11.32	11.37	11.45	11.53	11.60	11.89	12.54
5%	10.88	7.95	5.11		12.28	10.96	10.19	9.63	9.15	7.69	5.14
25%	20.62	16.12	12.05		22.55	20.64	19.36	18.33	17.61	15.40	12.13
50%	31.82	29.76	27.90		32.65	31.77	31.08	30.68	30.41	29.46	27.94
75%	38.64	36.37	34.72		39.70	38.61	37.90	37.45	37.15	36.27	34.84
95%	46.70	43.49	42.04		48.46	46.95	46.03	45.48	44.98	43.71	42.26
98%	50.49	46.75	45.28		52.35	50.76	49.55	48.75	48.18	46.58	45.41

Table A-3. Summary statistics for observed and predicted concentrations, easterly winds (30°-150°).

Distance	OBSERVATIONS				MODEL PREDICTED						
	20m	100m	300m		10m	20m	30m	40m	50m	100m	300m
NO$_2$ (ppb)											
Avg	19.31	18.11	16.59	22.95	20.49	19.74	19.31	19.00	18.77	18.02	16.85
Stdev	11.52	11.17	10.78	13.24	12.01	11.72	11.57	11.47	11.40	11.21	11.01
5%	5.66	4.90	4.03	6.63	6.11	5.93	5.75	5.66	5.54	5.11	4.09
25%	9.96	8.83	7.59	12.44	10.77	10.15	9.72	9.48	9.37	8.69	7.66
50%	16.50	15.64	13.97	19.88	17.57	16.93	16.45	16.18	16.03	15.29	14.18
75%	26.49	25.22	23.66	31.04	27.98	27.08	26.57	26.30	25.92	25.16	24.14
95%	41.33	39.40	36.64	47.77	43.34	41.70	41.13	40.61	40.34	39.07	37.26
98%	47.61	44.75	42.65	55.33	49.92	48.58	47.97	47.31	46.80	45.47	43.35

Table A-4. Summary statistics for observed and predicted concentrations for inversion conditions (convective mixing height less than 300 m).

	OBSERVATIONS				MODEL						
Distance	20m	100m	300m		10m	20m	30m	40m	50m	100m	300m
NO$_2$ (ppb)											
Avg	31.72	28.14	25.96		33.33	31.80	30.91	30.28	29.78	28.26	25.84
Stdev	12.14	11.54	11.16		12.60	12.19	11.98	11.84	11.75	11.52	11.32
5%	11.85	9.65	8.50		12.41	11.90	11.45	11.19	10.95	10.00	8.41
25%	22.79	18.83	16.95		23.71	22.73	21.78	21.19	20.67	19.04	16.47
50%	32.58	28.50	26.53		33.97	32.77	31.84	31.10	30.52	28.67	26.05
75%	40.89	37.07	33.99		42.91	41.03	39.96	39.28	38.61	36.89	34.12
95%	50.90	46.09	43.95		53.02	50.95	49.79	48.77	48.12	46.48	43.94
98%	53.85	49.89	46.89		56.92	53.78	52.70	52.08	51.48	49.36	47.04

Table A-5. Summary statistics for observed and predicted concentrations for non-inversion conditions (convective mixing height greater than 300 m).

	OBSERVATIONS				MODEL						
Distance	20m	100m	300m		10m	20m	30m	40m	50m	100m	300m
NO$_2$ (ppb)											
Avg	16.85	13.89	11.56	22.81	18.25	16.88	16.07	15.50	15.06	13.69	11.51
Stdev	8.95	8.55	8.47	11.77	9.42	8.94	8.72	8.60	8.53	8.40	8.54
5%	5.95	4.87	3.64	6.88	6.42	6.08	5.94	5.81	5.70	5.16	3.63
25%	10.30	7.97	5.70	13.97	11.23	10.32	9.75	9.36	9.02	7.95	5.63
50%	15.04	10.79	7.98	21.32	16.53	15.03	14.13	13.38	12.85	11.02	8.17
75%	21.24	16.09	13.71	30.08	23.16	21.21	19.99	19.17	18.50	16.82	14.90
95%	34.92	31.04	28.89	43.79	36.71	34.94	34.10	33.49	33.07	31.41	29.62
98%	40.15	35.89	34.13	50.39	42.40	40.27	39.12	38.55	38.38	36.84	35.10

Table A-6. Summary statistics for percent change in NO$_2$ concentration from modeled on-road to concentrations at distances of 10 m and 20 m away from roads for all atmospheric conditions combined, by quintile of NO$_2$ predicted concentrations at 10 m and 20 m away from the roads.

	10 m to on-road comparison		20 m to on-road comparison	
Quintile	Conc range (ppb)	Average % change	Conc range (ppb)	Average % change
1	-1.27 to 15.16	13%	1.32 to 13.79	19%
2	15.17 to 23.78	19%	13.80 to 22.21	25%
3	23.78 to 33.24	14%	22.21 to 31.95	17%
4	33.24 to 42.55	10%	31.95 to 41.12	11%
5	42.55 to 69.01	11%	41.13 to 68.60	12%
overall	-1.27 to 69.01	13%	1.32 to 68.60	17%

Table A-7. Summary statistics for percent change in NO$_2$ concentration from modeled on-road to concentrations at distances of 10 m and 20 m away from roads for winds from the west (210°-330°), by quintile of NO$_2$ predicted concentrations at 10 m and 20 m away from roads.

	10 m to on-road comparison		20 m to on-road comparison	
Quintile	Conc range (ppb)	Average % change	Conc range (ppb)	Average % change
1	0.50 to 20.69	25%	1.32 to 18.73	34%
2	20.70 to 30.15	15%	18.77 to 28.94	22%
3	30.15 to 36.48	8%	28.94 to 35.50	11%
4	36.49 to 42.85	8%	35.50 to 41.59	10%
5	42.89 to 64.84	10%	41.60 to 62.56	11%
overall	0.50 to 64.84	13%	1.32 to 62.56	17%

Table A-8. Summary statistics for percent change in NO$_2$ concentration from modeled on-road to concentrations at distances of 10 m and 20 m away from roads for winds from the east (30°-150°), by quintile of NO$_2$ predicted concentrations at 10 m and 20 m away from roads.

	10 m to on-road comparison		20 m to on-road comparison	
Quintile	Conc range (ppb)	Average % change	Conc range (ppb)	Average % change
1	-1.27 to 10.23	10%	2.31 to 9.70	15%
2	10.23 to 16.01	12%	9.70 to 15.26	17%
3	16.02 to 24.25	10%	15.26 to 23.50	11%
4	24.26 to 38.08	9%	23.50 to 36.95	12%
5	38.10 to 69.01	9%	36.96 to 68.60	11%
overall	-1.27 to 69.01	10%	2.31 to 68.60	13%

Table A-9. Summary statistics for percent change in NO$_2$ concentration from modeled on-road to concentrations at distances of 10 m and 20 m away from roads for inversion conditions (mixing height < 300 m), by quintile of NO$_2$ predicted concentrations at 10 m and 20 m away from roads.

	10 m to on-road comparison		20 m to on-road comparison	
Quintile	Conc range (ppb)	Average % change	Conc range (ppb)	Average % change
1	4.60 to 22.07	10%	5.17 to 20.55	13%
2	22.08 to 30.35	14%	20.61 to 28.66	19%
3	30.45 to 38.40	13%	28.69 to 36.72	17%
4	38.44 to 46.29	12%	36.84 to 44.57	15%
5	46.37 to 69.01	12%	44.61 to 68.60	14%
overall	4.60 to 69.01	12%	5.17 to 68.60	16%

Table A-10. Summary statistics for percent change in NO$_2$ concentration from modeled on-road to concentrations at distances of 10 m and 20 m away from roads for non-inversion conditions (mixing height > 300 m), by quintile of NO$_2$ predicted concentrations at 10 m and 20 m away from roads.

Quintile	10 m to on-road comparison		20 m to on-road comparison	
	Conc range (ppb)	Average % change	Conc range (ppb)	Average % change
1	-1.27 to 10.36	7%	2.31 to 9.63	10%
2	10.36 to 15.01	20%	9.64 to 13.53	27%
3	15.01 to 19.78	22%	13.54 to 17.94	29%
4	19.79 to 27.27	21%	17.94 to 25.14	26%
5	27.29 to 67.78	15%	25.15 to 66.94	17%
overall	-1.27 to 67.78	17%	2.31 to 66.94	22%
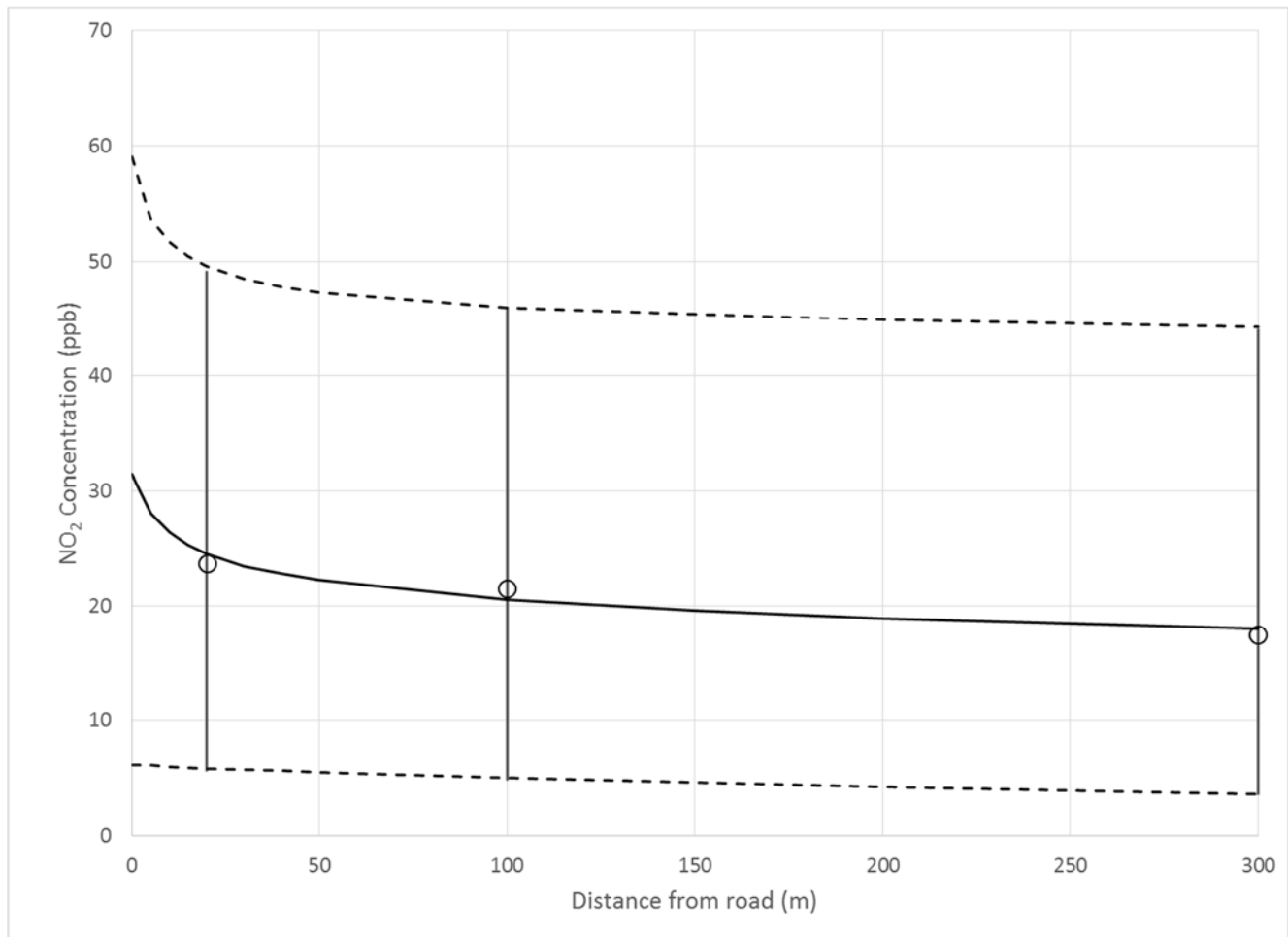
Figure A-1. Predicted and observed NO$_2$ concentrations for all wind and stability conditions combined. Predicted median (solid), predicted 98th and 2nd percentile (dotted), observed median (circles), and observed 98th and 2nd percentiles (error bars) are shown.
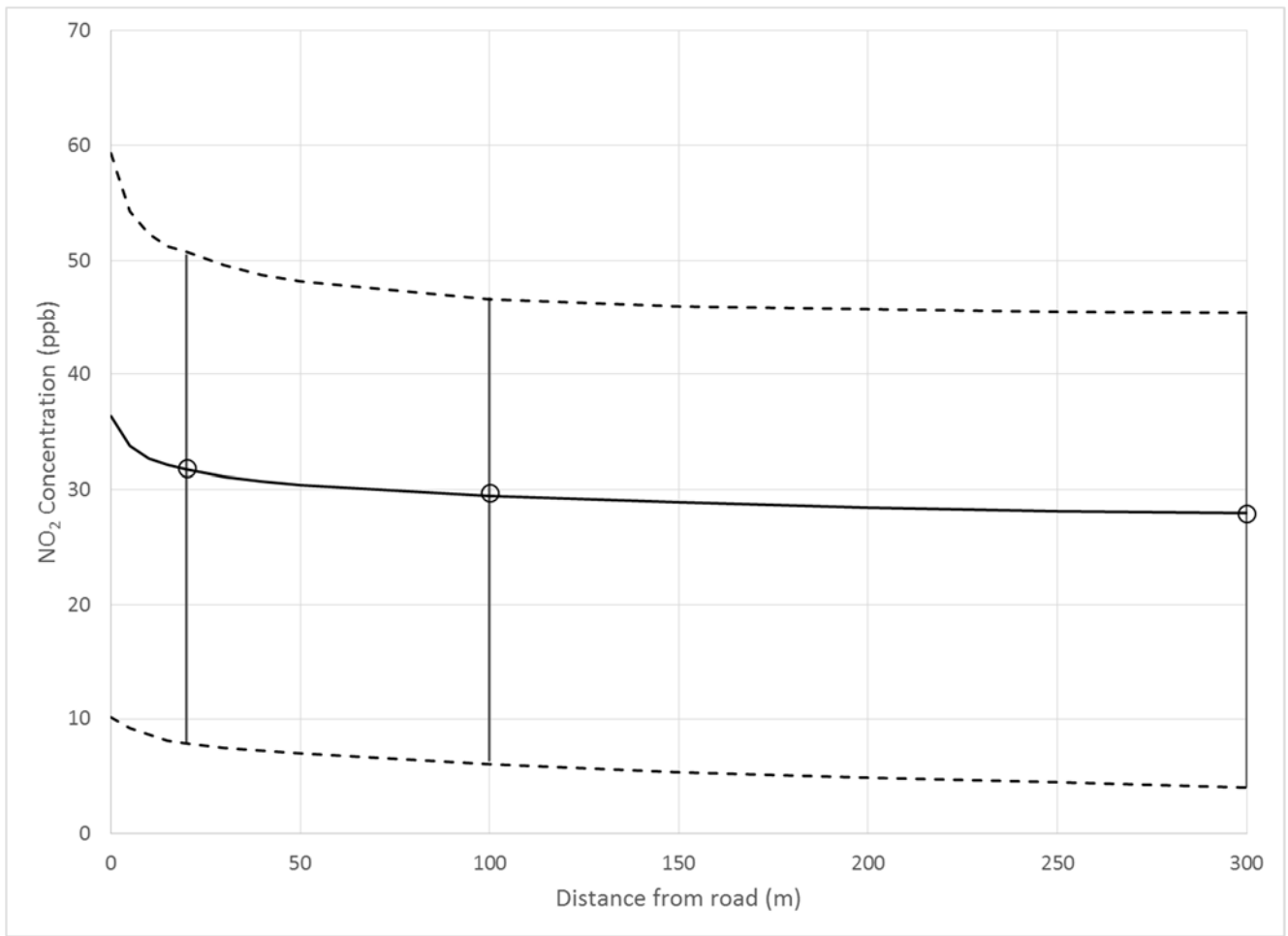
Figure A-2. Predicted and observed NO$_2$ concentrations for winds from the west. Predicted median (solid), predicted 98th and 2nd percentile (dotted), observed median (circles), and observed 98th and 2nd percentiles (error bars) are shown.
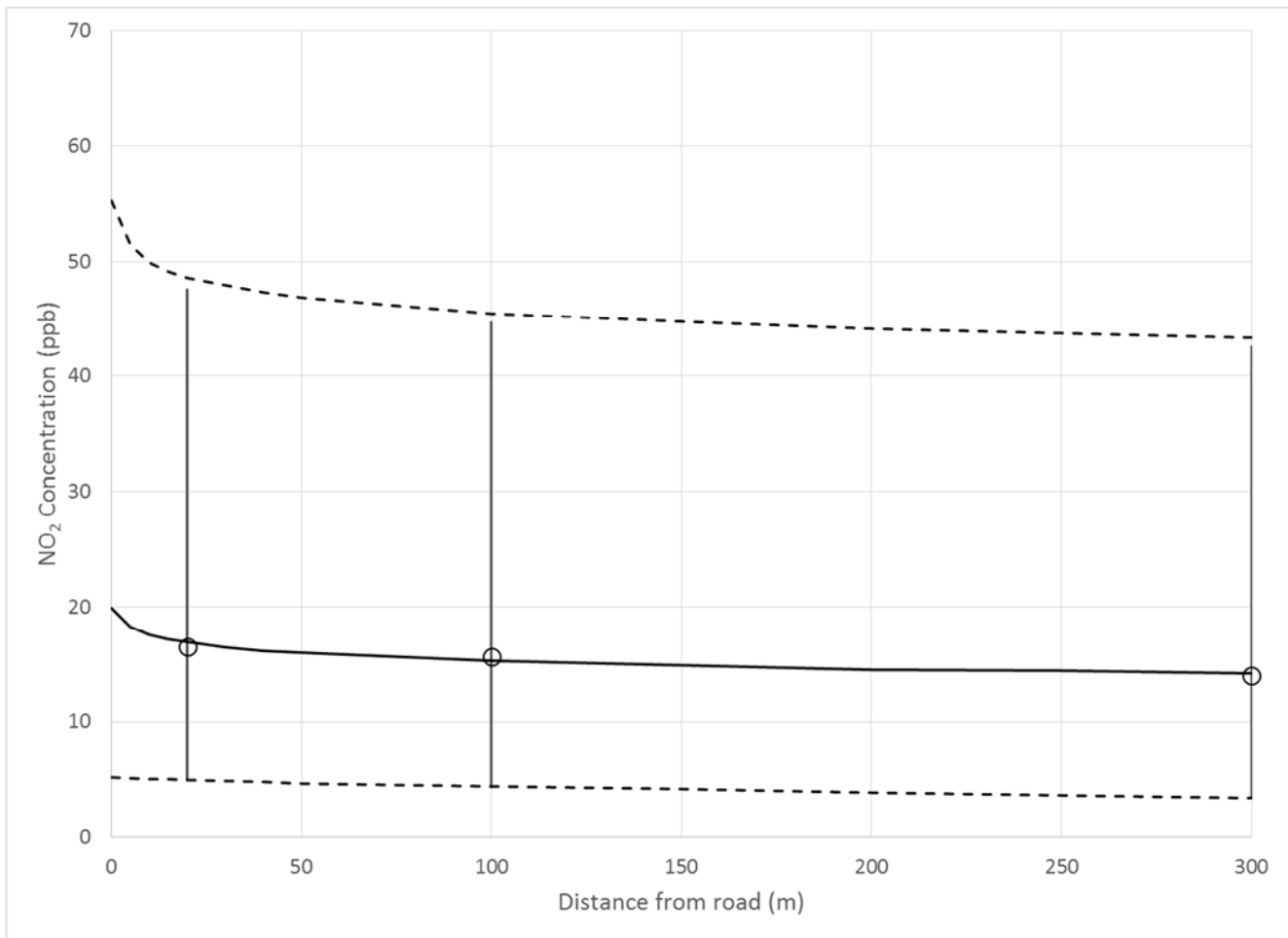
Figure A-3. Predicted and observed NO$_2$ concentrations for winds from the east. Predicted median (solid), predicted 98th and 2nd percentile (dotted), observed median (circles), and observed 98th and 2nd percentiles (error bars) are shown.
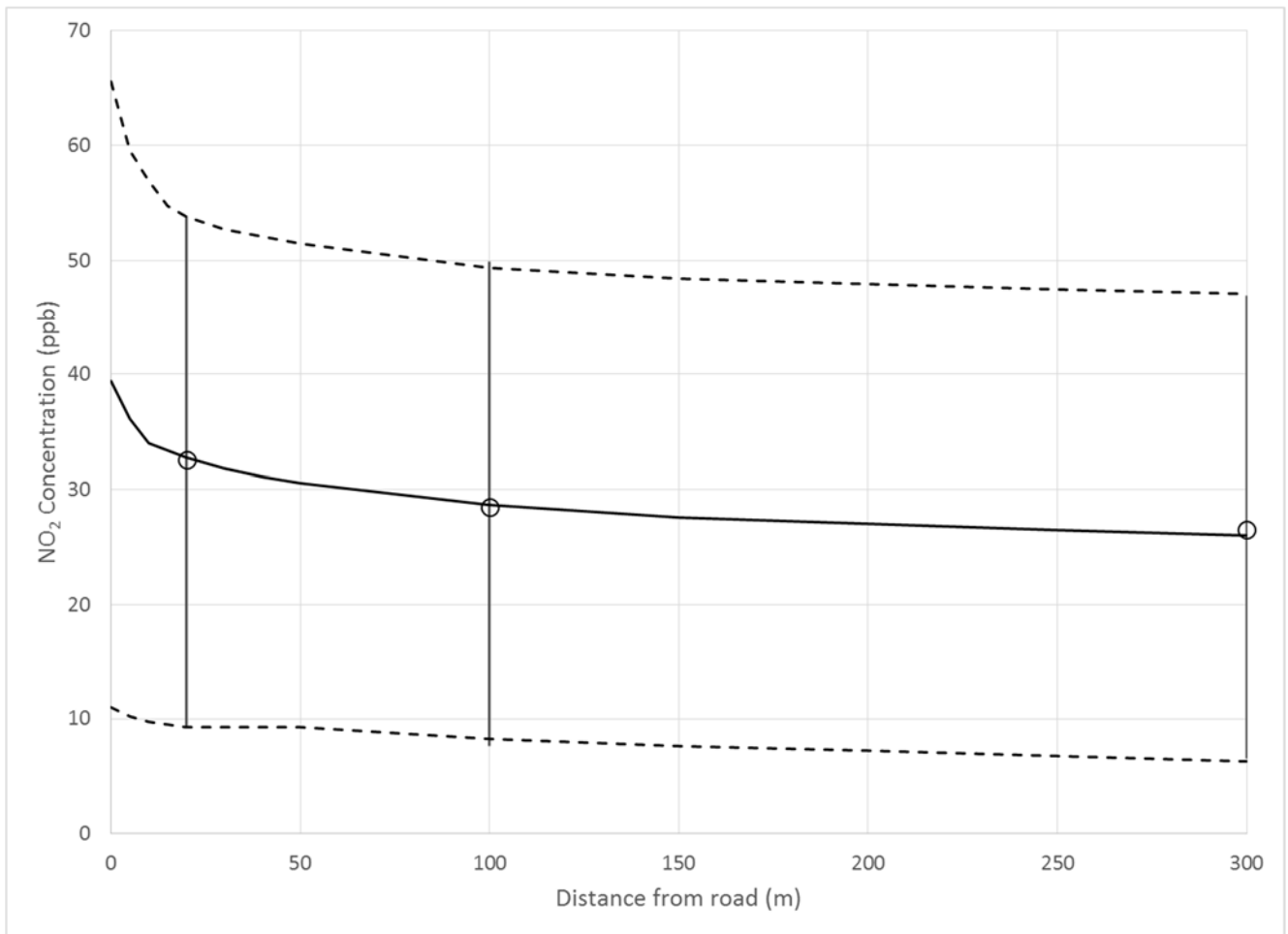
Figure A-4. Predicted and observed NO$_2$ concentrations for inversion conditions (convective mixing height less than 300 m). Predicted median (solid), predicted 98th and 2nd percentile (dotted), observed median (circles), and observed 98th and 2nd percentiles (error bars) are shown.
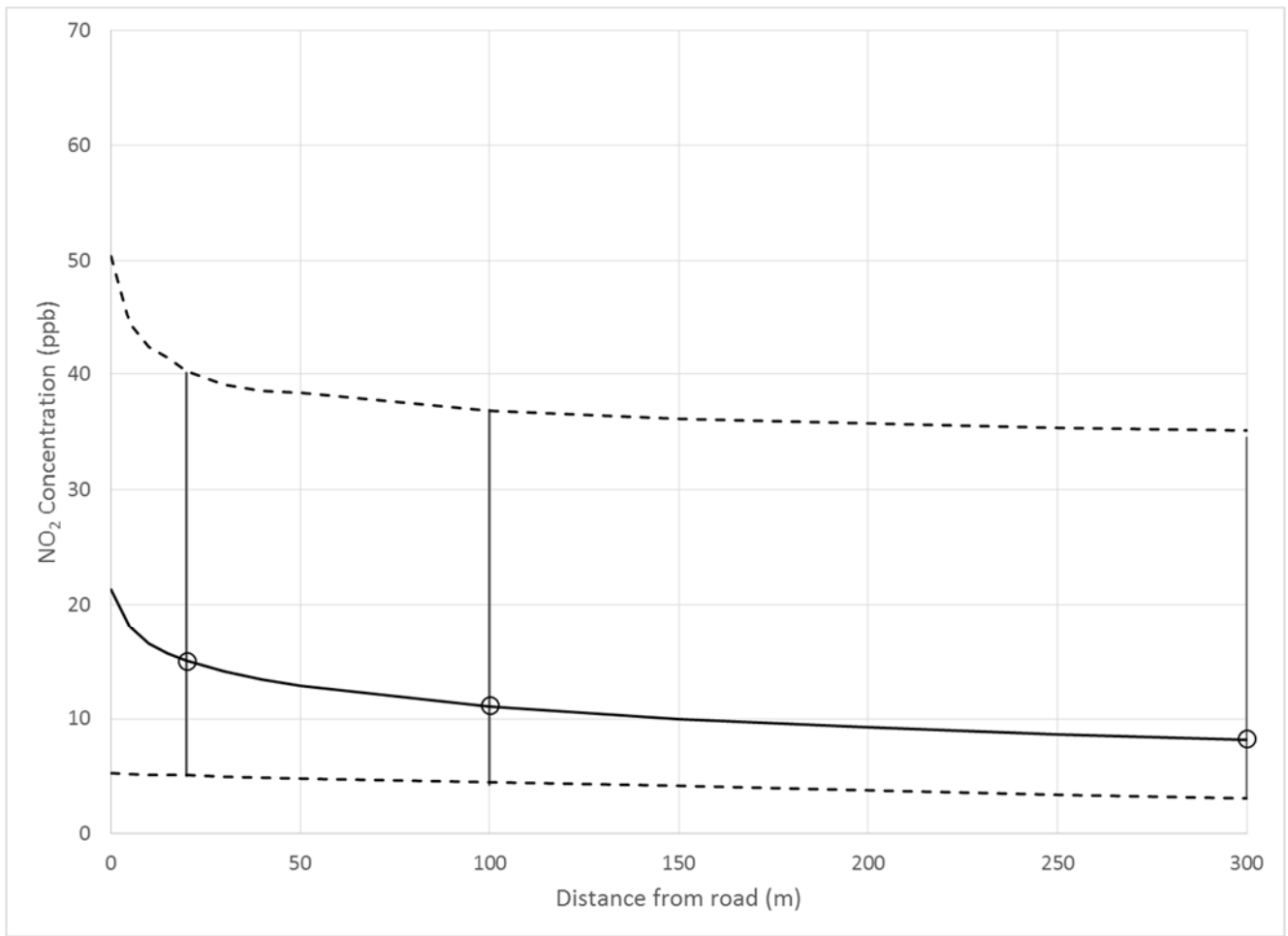
Figure A-5. Predicted and observed NO$_2$ concentrations for non-inversion conditions (convective mixing height greater than 300 m). Predicted median (solid), predicted 98th and 2nd percentile (dotted), observed median (circles), and observed 98th and 2nd percentiles (error bars) are shown.

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APPENDIX B: CONCENTRATION PLOTS AND MAPS OF SELECTED PHILADELPHIA CBSA MONITORS

This appendix provides plots of low- and high-concentration year ambient monitoring data and maps showing the local built environment and natural features surrounding monitors. All high-concentration year data used were from 1984. Two sets of low concentration year data were used; the first was 2007, the same year used in the 2008 REA (Rizzo, 2008), the second low concentration year was from 2013 where available, otherwise, the most recent year of air quality data available.

List of Figures

Reference

Rizzo M. (2008). Investigation of how distributions of hourly nitrogen dioxide concentrations have changed over time in six cities. Nitrogen Dioxide NAAQS Review Docket (EPA–HQ–OAR–2006–0922). Available at: http://www.epa.gov/ttn/naaqs/standards/nox/s_nox_cr_rea.html.

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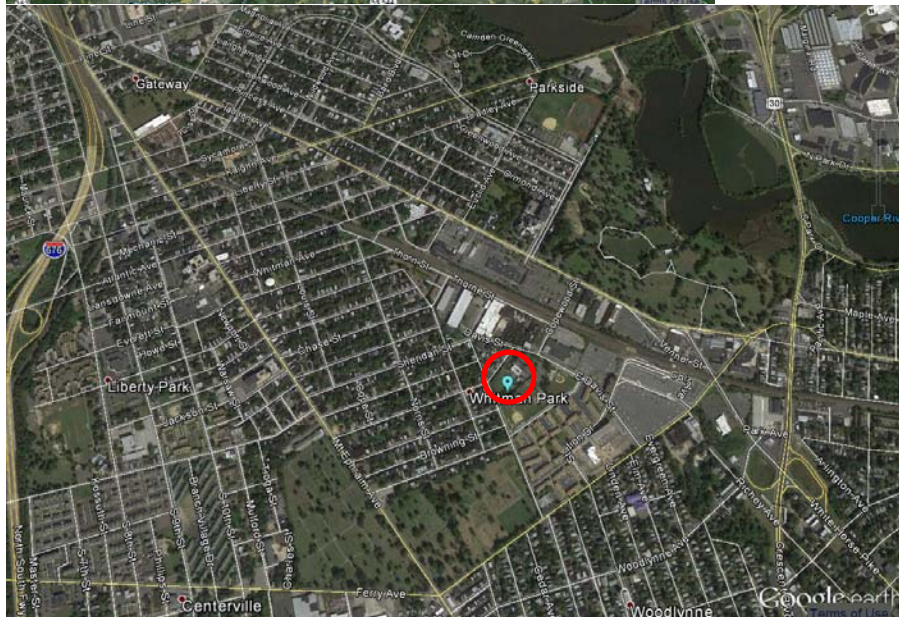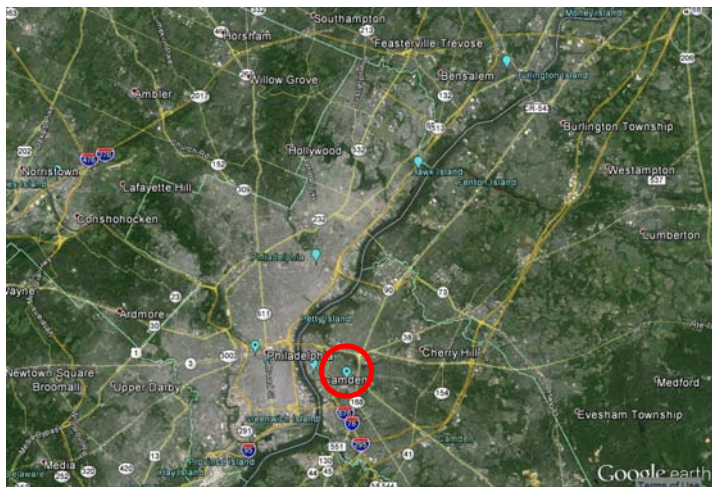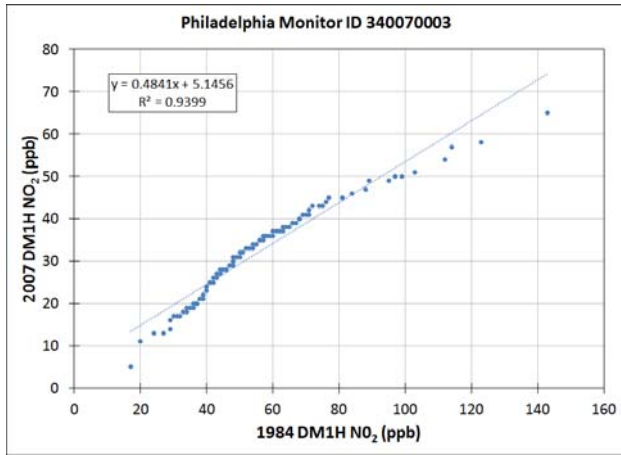
Figure B-1. Comparison of low- (2007, y-axis top left panel) and high- (1984, x-axis top panel) concentration years for Philadelphia CBSA monitor ID 340070003. Map indicating the monitor location within CBSA (middle panel) and expanded view (bottom panel) to show the local built-environment and natural features proximal to the monitor.
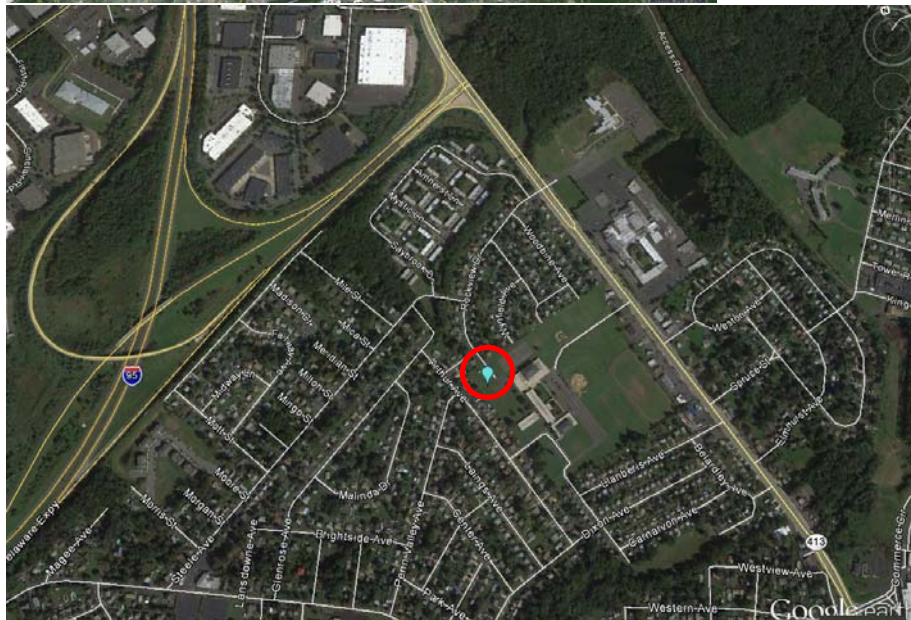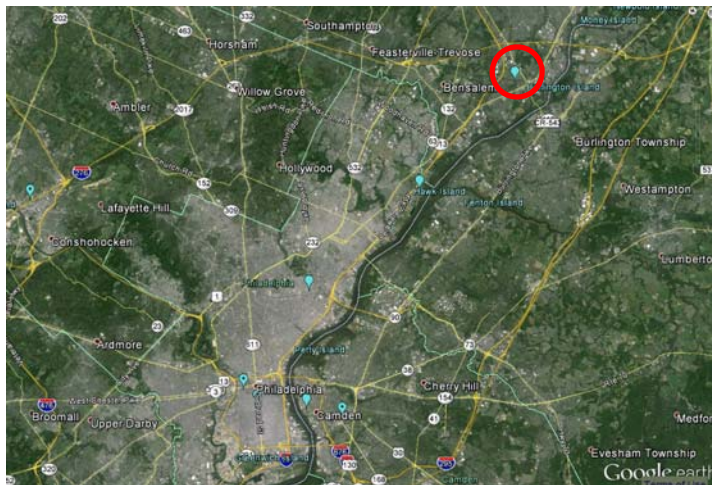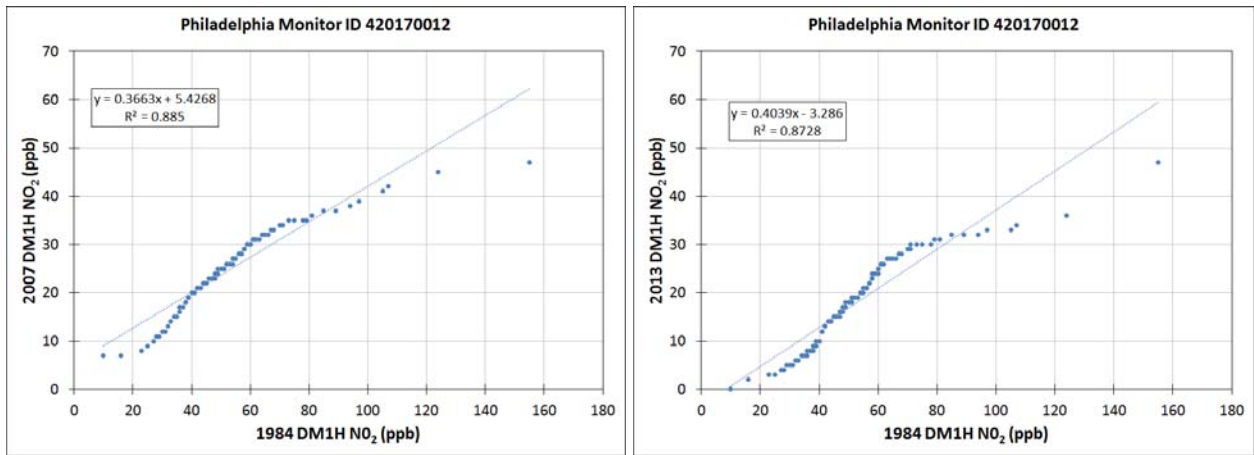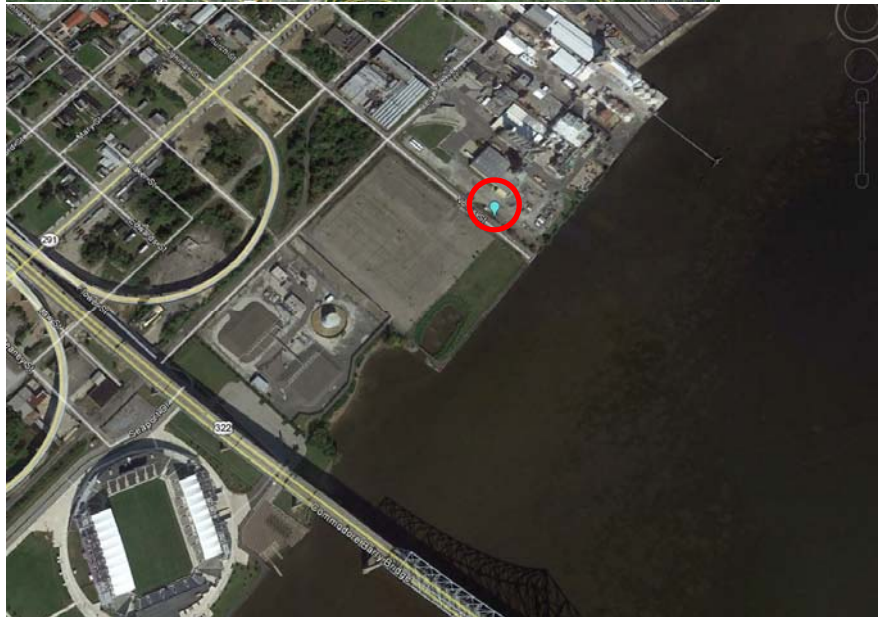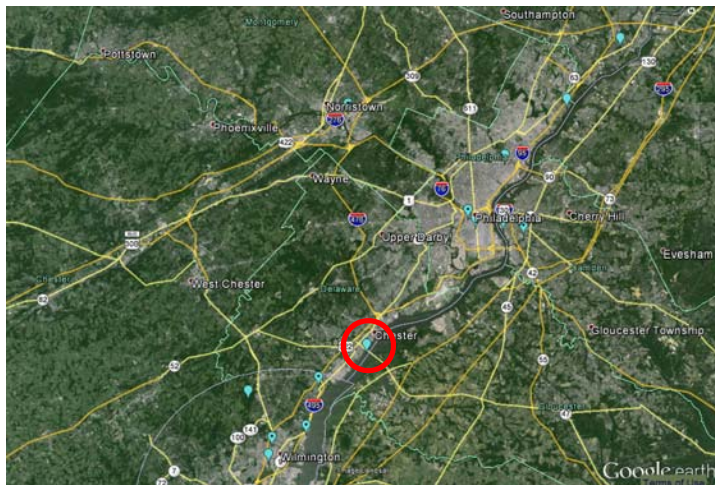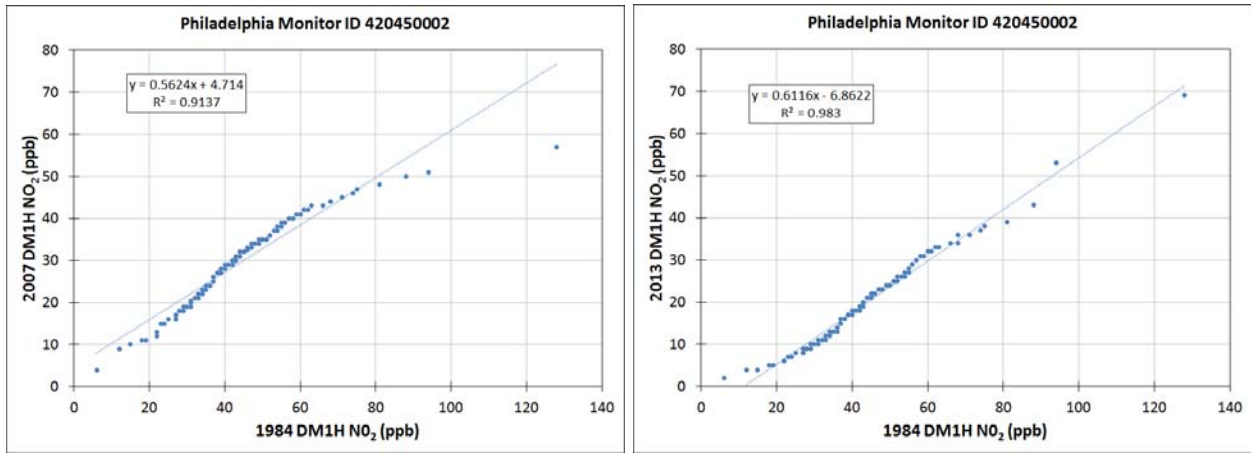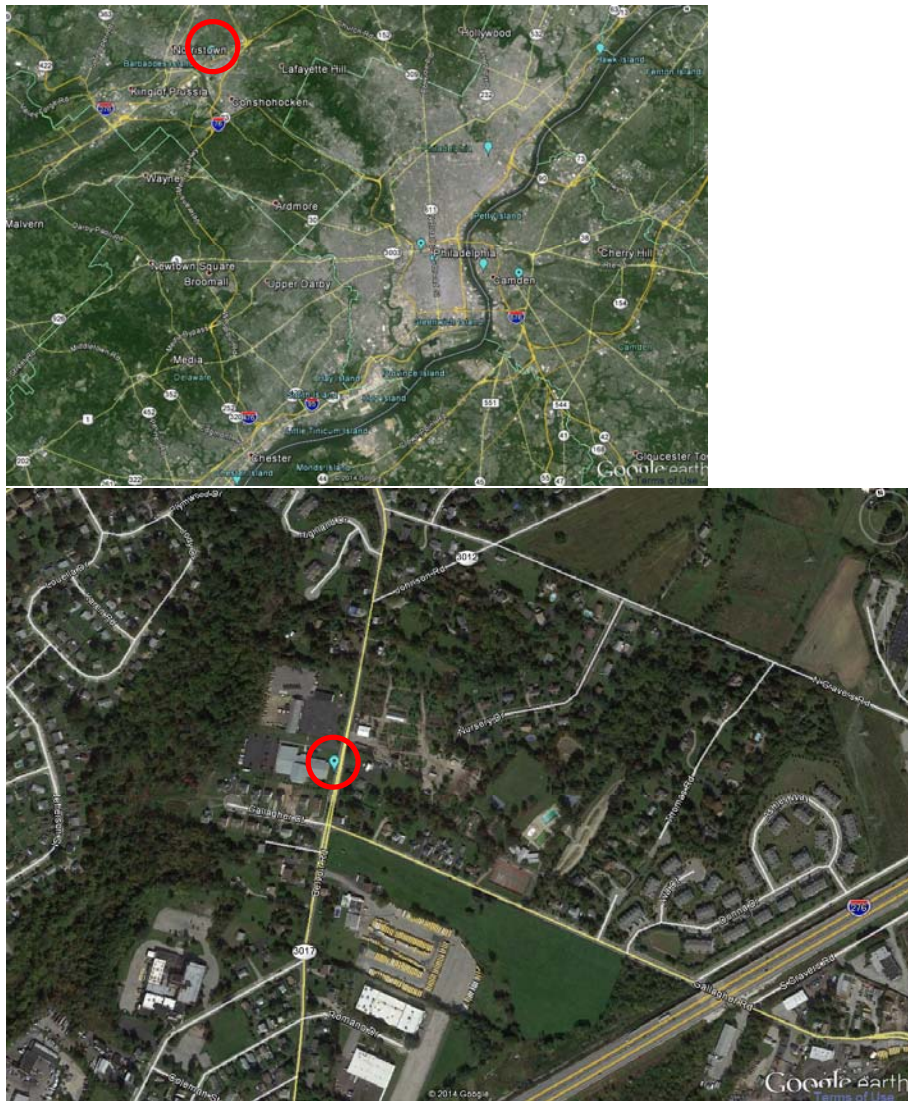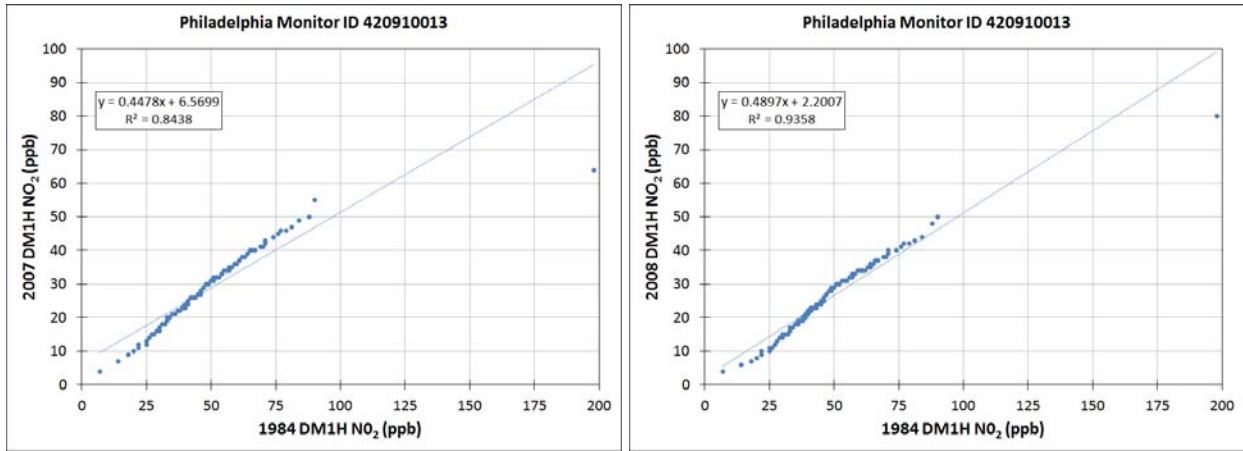
Figure B-2. Comparison of low- (2007, y-axis top left panel; 2013, y-axis top right panel) and high- (1984, x-axes top panels) concentration years for Philadelphia CBSA monitor ID 420170012. Map indicating the monitor location within CBSA (middle panel) and expanded view (bottom panel) to show the local built-environment and natural features proximal to the monitor.
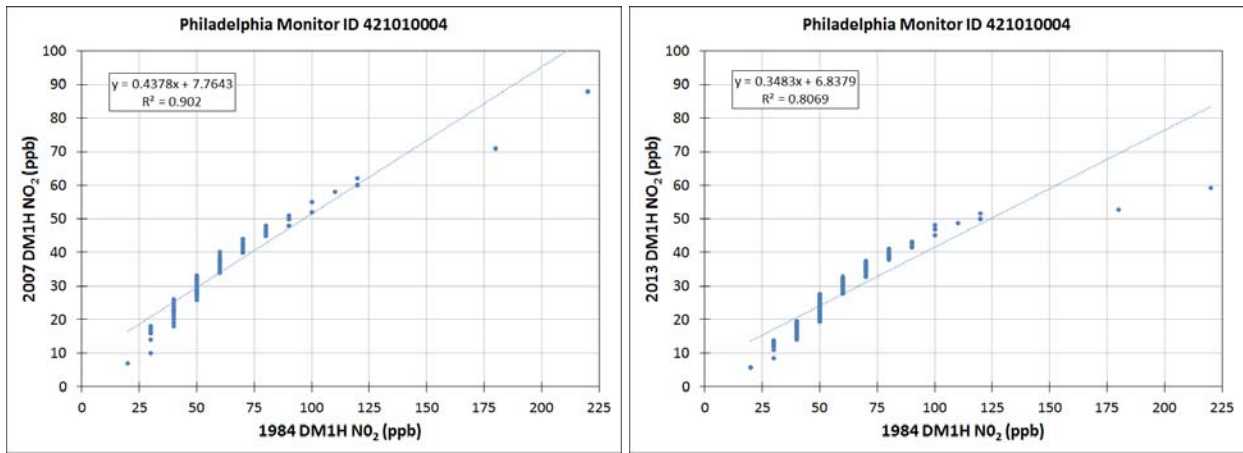
B-2

Figure B-3. Comparison of low- (2007, y-axis top left panel; 2013, y-axis top right panel) and high- (1984, x-axes top panels) concentration years for Philadelphia CBSA monitor ID 420450002. Map indicating the monitor location within CBSA (middle panel) and expanded view (bottom panel) to show the local built-environment and natural features proximal to the monitor.

Figure B-4. Comparison of low- (2007, y-axis top left panel; 2008, y-axis top right panel) and high- (1984, x-axes top panels) concentration years for Philadelphia CBSA monitor ID 420910013. Map indicating the monitor location within CBSA (middle panel) and expanded view (bottom panel) to show the local built-environment and natural features proximal to the monitor.
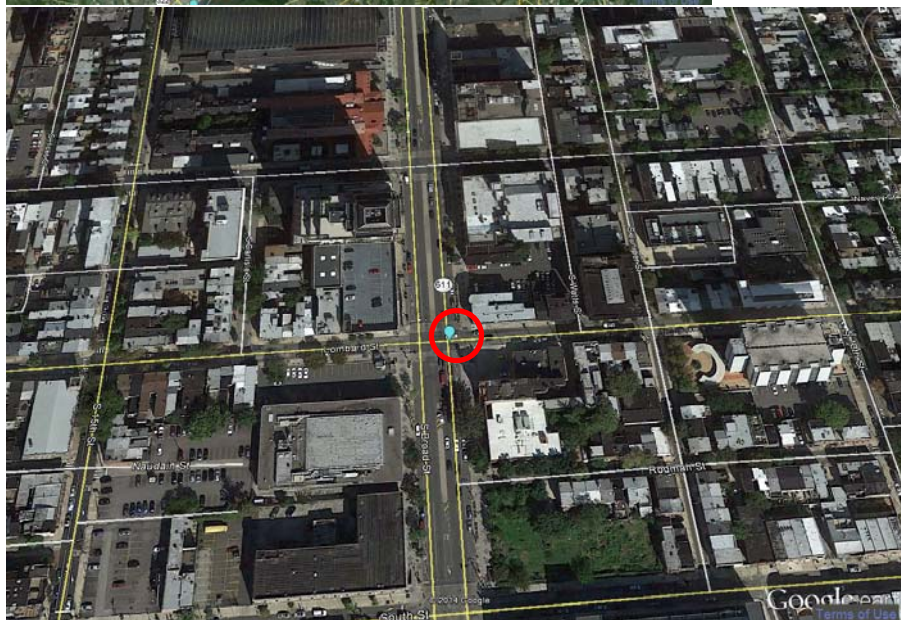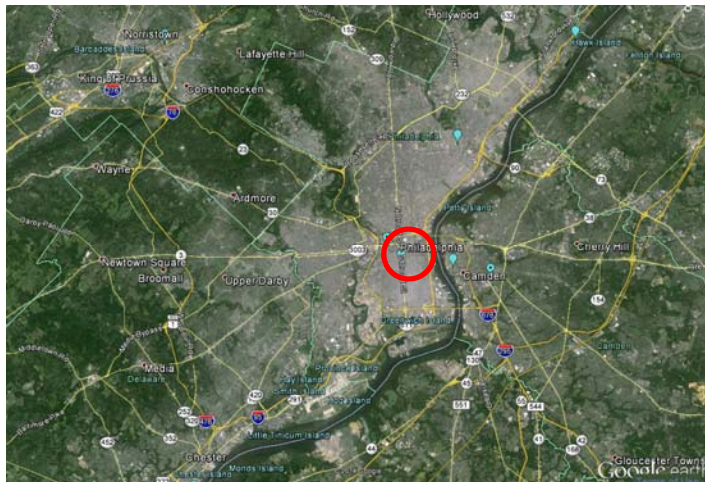
Figure B-5. Comparison of low- (2007, y-axis top left panel; 2013, y-axis top right panel) and high- (1984, x-axes top panels) concentration years for Philadelphia CBSA monitor ID 421010004. Map indicating the monitor location within CBSA (middle panel) and expanded view (bottom panel) to show the local built-environment and natural features proximal to the monitor.
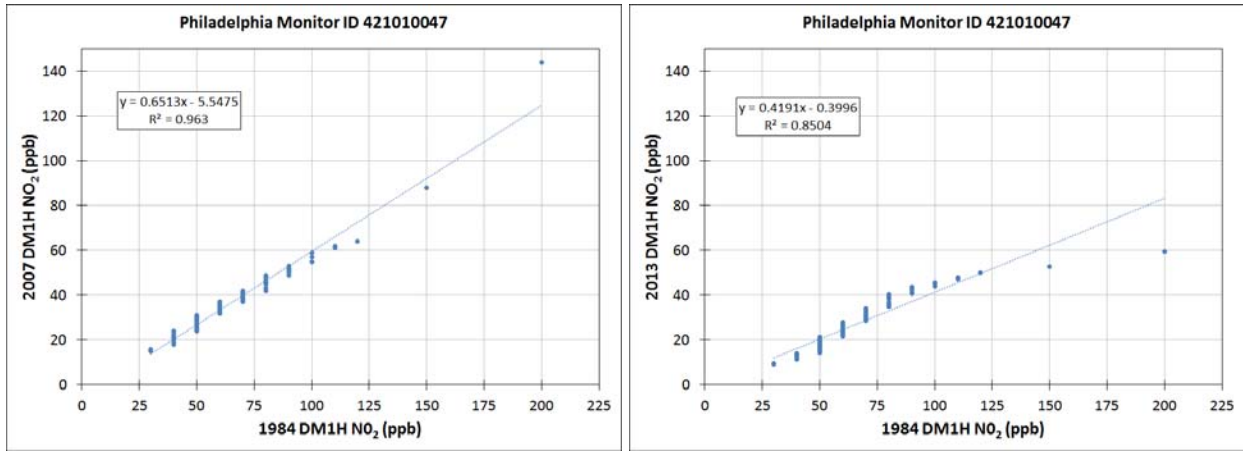
Figure B-6. Comparison of low- (2007, y-axis top left panel; 2013, y-axis top right panel) and high- (1984, x-axes top panels) concentration years for Philadelphia CBSA monitor ID 421010047. Map indicating the monitor location within CBSA (middle panel) and expanded view (bottom panel) to show the local built-environment and natural features proximal to the monitor.

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