

Comments on the Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards, Draft Version 2

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Disclaimer: Dr. Lefohn's comments contained within this document are his own; he represents only himself; no person or organization has seen these comments prior to submission to the Government Docket; and he has not been reimbursed for the time necessary to produce these comments. His comments are directed at clarifying the state-of-science evidence presented in the second draft Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards. His comments are submitted to the Ozone Docket (No. EPA-HQ-OAR-2018-0279) for the purpose of providing scientific clarification for the ozone (O₃) rulemaking activity. Previously, Dr. Lefohn provided comments to the Ozone Docket on the first draft of the Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards (Lefohn, 2022), the 2019 draft Integrated Science Assessment document (Lefohn, 2019a) and the 2019 draft Policy Assessment document (Lefohn et al., 2019b) that the EPA finalized in 2020 (EPA, 2020a, 2020b), and the 2020 draft Review of the Ozone National Ambient Air Quality Standards (Federal Register, 2020a) proposal (Lefohn, 2020).

About the Author

Dr. Allen S. Lefohn is President and Founder of A.S.L. & Associates, LLC (<http://www.asl-associates.com/>), which is located near Helena, Montana. He received his Bachelor of Science degree from UCLA in 1966 and a Ph.D. in physical chemistry from the University of California Berkeley in 1969. His Ph.D. advisor was Professor George C. Pimentel, who served as deputy director of the National Science Foundation from 1977 to 1980. Dr. Lefohn, for the period 1989 – 1999, served as an Executive Editor of the internationally recognized journal *Atmospheric Environment* and is an Emeritus Editor of the Journal. Dr. Lefohn has published approximately 125 peer-reviewed publications, edited four books, presented numerous oral papers, and participated in panel presentations. He is the editor and author of the book *Surface-level Ozone Exposures and Their Effects on Vegetation* published by Lewis Publishers, Inc., Chelsea, MI. For many years, he served as an Adjunct Professor of Environmental Engineering at Montana Tech in Butte, Montana. Dr. Lefohn has been involved in all the American Lung Association's annual State of the Air reports (2000-2023). These reports provide a county-by-county summary of ozone (smog) and PM_{2.5} (soot) concentrations experienced across the United States. The report is important in better understanding the variability in O₃ exposure (both spatially and temporally) across the US as emission reductions and climate change factors interact with one another in a changing world. During a career spanning over 50 years, his research has focused on (1)

developing exposure-response relationships and indices that describe the effects of O₃ on vegetation and human health, (2) investigating biological mechanisms that define the weighting of the higher concentrations more than the mid- and low-level values for responses to O₃ for both human health and vegetation, (3) understanding the importance of background O₃ and specific natural processes in relation to ambient concentrations, and (4) integrating results from the EPA's air quality database for (a) characterizing co-occurrence patterns (i.e., both simultaneous and sequential) of criteria air pollutants under ambient conditions, (b) characterizing O₃ trend patterns, and (c) designing research experiments that utilize realistic ambient exposures for assessing human health and vegetation effects. He designed the hour-by-hour concentration regimes for the Adams (2003, 2006a) experiments used in Dr. Adams' controlled human exposure studies which continue to play an important role in the O₃ standard-setting science assessments. Dr. Lefohn designed, with the assistance of Dr. Milan Hazucha of the University of North Carolina, Chapel Hill, the hour-by-hour exposure regimes for the controlled human exposure study performed by Schelegle et al. (2009), which the EPA focused on in its 2015 and 2019 O₃ rulemaking for the 70 ppb O₃ primary standard. He served as Chairman of the Science Advisory Committee of the Center for Ecological Health Research, University of California, Davis and served as a member of the Committee until January 2002. His research results associated with identifying exposure metrics for assessing effects have been used by the EPA. Between 2007 and 2015, EPA staff and CASAC considered the application of an exposure metric, the W126 exposure index, as the federal secondary standard to protect vegetation. Dr. Lefohn introduced the metric into the peer-review literature in 1987 and 1988. Currently the EPA uses the 8-h O₃ standard to limit W126 exposures to protect vegetation; the EPA continues to utilize the W126 metric as an indicator of the potential risk of ambient O₃ exposures to vegetation. He was the lead consultant scientist for the EPA in authoring the air quality characterization chapter and the vegetation exposure-response section for the Ozone Criteria Document in 1996 and contributed to the Ozone Criteria Documents in 1985 and 2006. His research results were cited in the EPA 2019 Integrated Science Assessment and 2019 Policy Assessment documents, as well as the December 2020 decision by the EPA Administrator on the O₃ NAAQS. Some of his research results associated with O₃ trends, changing patterns of exposure over time, vegetation effects, and realistic ambient exposure regimes are cited in the second draft of the Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards. Dr. Lefohn presented testimony in March 2015 to the House Committee on Science, Space, and Technology about background O₃. In 2015, Dr. Lefohn was a co-guest editor for the *Atmospheric Environment* special issue: Observations and source attribution of O₃ in rural regions of the Western United States. Dr. Lefohn was a member of the first Steering Committee (2014-2019) of the international research effort, Tropospheric Ozone Assessment Report (TOAR). Dr. Lefohn was the lead author (with 23 additional co-authors) of the well-cited TOAR paper, **Global Ozone Metrics for Climate Change, Human Health and Crop/Ecosystem Research**, which was published in April 2018 and is available at <https://www.elementscience.org/article/10.1525/elementa.279/>. Dr. Lefohn continues to be an active contributor in his research areas of interest. He has lived in Montana for almost 47 years and continues to perform his worldwide research from this location.

Executive Summary

1. Two key fundamental principles influence the O₃ rulemaking activity. The first fundamental principle is **Higher Hourly Average O₃ Concentrations Should be Weighted More than Middle and Lower Values when Assessing Human Health and Environmental Effects**. For human health, this principle is based on an important series of human health clinical studies published by Hazucha et al. (1992) and Adams (2003, 2006a, b). These controlled human health clinical studies showed that greater responses were observed shortly after peak O₃ exposures in stepwise and triangular (i.e., profiles with increases and decreases in concentration) profiles rather than in constant concentration exposure protocols. (Section 2).
2. For vegetation, the first fundamental principle is supported by key research results reported in the 1980s and 1990s. In the most current ISA (EPA, 2020a), the Agency continues to conclude, based on experimental studies, that (1) O₃ effects in plants are cumulative; (2) higher O₃ concentrations appear to be more important than lower concentrations in eliciting a response; (3) plant sensitivity to O₃ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the O₃ hourly concentrations and preferentially weight the higher concentrations, improves the predictive power of exposure/response models for growth and yield, in comparison with using exposure indices based on mean and other exposure metrics. Results from a “natural experiment” site in the San Bernardino National Forest, where substantial reductions over the years in the higher hourly average O₃ concentrations in the Los Angeles area occurred, provide independent confirmation of the experimental studies for the greater importance of the higher hourly average O₃ concentrations in influencing vegetation effects. (Section 2).
3. One result of the first fundamental principle is that Haber’s Rule ($C \times t = k$, where C is the concentration of the gas (mass per unit volume), t is the amount of time necessary to produce a given toxic effect, and k is a constant), is **not** applicable for O₃. (Section 2).
4. In implementing its Air Quality Index (AQI) reported across the U.S., EPA recognizes the importance of the higher O₃ concentration. Ozone pollutant specific sensitive groups are separated by 8-h daily maximum O₃ concentrations as shown below in Fig. ES-1. The higher the O₃ concentration exposures, the greater the potential effect on human health. (Section 2).

8-hour Ozone Concentration	Air Quality Index Levels
0 - 54 ppb	■ Good (Green)
55 - 70 ppb	■ Moderate (Yellow)
71 - 85 ppb	■ Unhealthy for Sensitive Groups (Orange)
86 - 105 ppb	■ Unhealthy (Red)
106 - 200 ppb	■ Very Unhealthy (Purple)
>200 ppb	■ Hazardous (Maroon)

Figure ES-1. Air quality index levels (AQI) related to 8-h concentrations.

5. The Administrator noted (Federal Register, 2020b – page 87266) that in the review completed in 2015 that an 8-h averaging time remained appropriate for addressing health effects associated with short-term exposures to ambient air O₃ and that it could effectively limit health effects attributable to both short- and long-term O₃ exposures (80 FR 65348, October 26, 2015). Simply stated, by reducing *the higher part of the distribution of hourly average concentrations* (not just the peak hourly values), the EPA Administrator believed in 2015 that the risk to human health and vegetation would be reduced by reducing the hourly average O₃ concentrations at the upper part of the distribution curve. In addition, the EPA in its 2015 decision (Federal Register, 2015 – pages 65358 – 65359) anticipated that a revised standard with a level of 70 ppb would also reduce the occurrence of exposures to O₃ concentrations at least somewhat below 60 ppb based on its modeling results in the 2014 Health Risk and Exposure Assessment document (EPA, 2014b, Figs. 4-9 and 4-10). Thus, even if some members of at-risk populations might experience effects following exposures to O₃ concentrations somewhat below 60 ppb, the Administrator believed in 2015 that a revised level of 70 ppb would be anticipated to reduce the occurrence of such exposures. Thus, the EPA believed it had considered O₃ exposures that could be relevant for at-risk populations, such as children and people with asthma, and did not agree in 2015 that controlled human exposure studies reporting respiratory effects in healthy adults following exposures to 60 ppb O₃ necessitate a standard level below 70 ppb. In 2015, it was the opinion of the EPA (Federal Register, 2015 – page 65358) that both acute and chronic effects would be reduced in implementing the revised O₃ standards. (Section 5).

6. In December 2020, following a review of the scientific evidence associated with the current O₃ standards, the EPA issued its final decision to retain the existing standards without revision. The Agency noted that O₃ is the most appropriate indicator for a standard meant to provide protection against ambient photochemical oxidants (Federal Register, 2020b – page 87301). The EPA noted

As an initial matter, the Administrator recognizes the continued support in the current evidence for O₃ as the indicator for photochemical oxidants (as recognized in section II.D.1 of the proposal). As recognized in the proposal, no newly available evidence has been identified in this review

regarding the importance of photochemical oxidants other than O₃ with regard to abundance in ambient air, and potential for health effects, and “the primary literature evaluating the health and ecological effects of photochemical oxidants includes ozone almost exclusively as an indicator of photochemical oxidants” (ISA, p. IS-3). Accordingly, the information relating health effects to photochemical oxidants in ambient air is also focused on O₃. Thus, the Administrator concludes it is appropriate for O₃ to continue to be the indicator for the primary standard for photochemical oxidants.

The second draft PA (EPA, 2023, pages 3-102 – 3-103 and page 4-117) reaffirms EPA’s continued support for O₃ as the indicator for photochemical oxidants. (Section 1).

7. The second key fundamental principle is that **Daily Maximum Hourly Averaged O₃ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide**. As O₃ precursor emissions are reduced at specific locations across the U.S. to attain the current O₃ NAAQS, not all hourly average O₃ concentrations at these locations shift downward. In fact, at many locations, the lower hourly average O₃ concentrations shift upward as described in the next two items below. The second fundamental principle is supported using empirical data, as well as atmospheric chemistry/meteorological modeling results published in the peer-reviewed literature. (Section 3).
8. As emissions are reduced to meet the O₃ NAAQS, a compression of the higher and lower hourly average O₃ concentrations occurs at many sites in the U.S. The higher individual 8-h daily maximum (MDA8) values are reduced downward toward the mid-level concentrations, while the lowest MDA8 values *increase* toward the mid-level values. As an example, for an urban-influenced site in 1985 in Jefferson County, Kentucky, there were frequent occurrences of high and low hourly average O₃ concentrations (Fig. ES-2). The site in 1985 appeared to be influenced by NO titration of O₃ by NO_x emissions because of the occurrence of frequent low hourly average concentrations. The distribution of hourly average data for the same Kentucky site in 2017 is shown in Fig. ES-3. Although the site is still influenced by anthropogenic sources, the highest hourly average O₃ concentration has been reduced from 112 ppb (experienced in 1985) to 77 ppb (experienced in 2017). In addition, a shift of the lower concentrations toward the mid-level values has occurred. The shift is associated with less titration of O₃ by NO as reduction in NO_x emissions occur (Lefohn et al., 1998; Lefohn et al., 2010b; EPA, 2014b, 2020b; Simon et al., 2015; Lefohn et al., 2017, 2018). The reduction of O₃ precursors results in both the high and the low concentrations shifting toward the mid-level values, resulting in a compression of the distribution of hourly average concentrations. (Section 3).

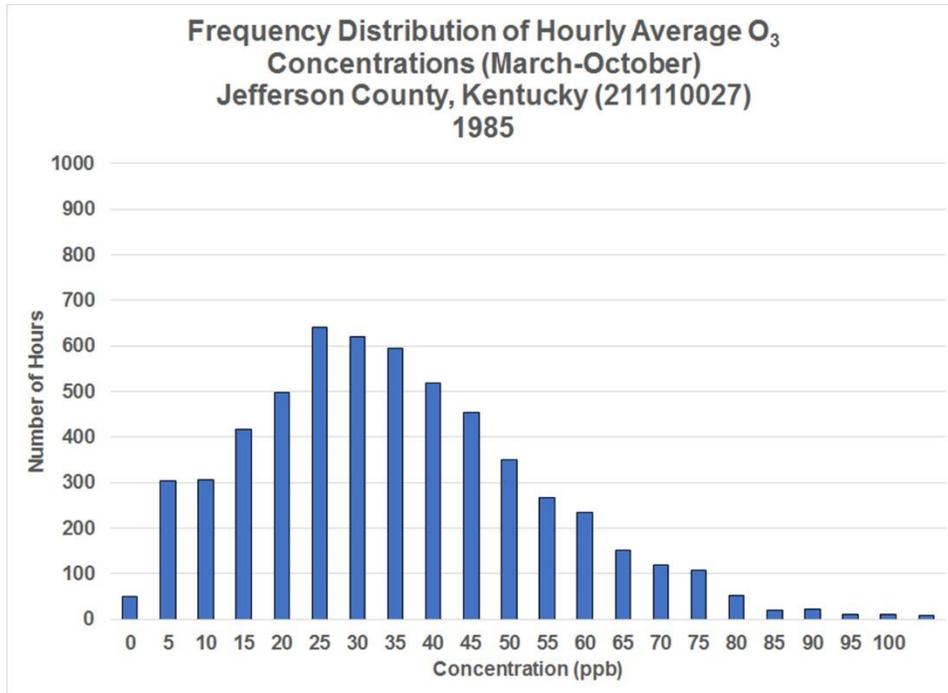


Figure ES-2. Frequency distribution of the hourly average O₃ concentrations in 1985 for an urban-influenced site in Jefferson County (KY) (AQS ID 211110027) monitoring station. Source of data is from the EPA’s AQS database.

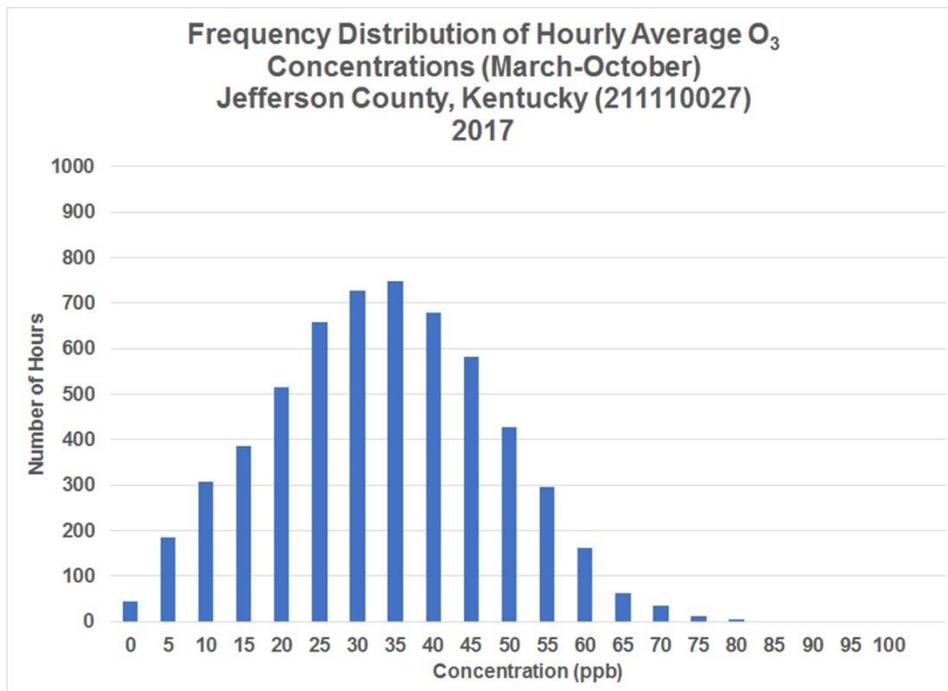


Figure ES-3. Frequency distribution of the hourly average O₃ concentrations in 2017 for an urban-influenced site in Jefferson County (KY) (AQS ID 211110027) monitoring station. Source of data is from the EPA’s AQS database.

9. Because of the unfortunate COVID-19 pandemic, many countries around the world during the spring (Northern Hemisphere) and fall (Southern Hemisphere) of 2020 instituted immediate lockdown orders. As a result of these orders, anthropogenic emissions were severely reduced. Recognizing the opportunity to investigate how severe emission reductions influenced air pollution concentrations, researchers documented the air quality changes associated with this “natural experiment.” One important result of the 2020 lockdown was documentation about the changes that occurred in the distribution of hourly average O₃ concentrations as emissions were reduced. During the 2020 COVID-19 lockdown, scientists characterized these changes by applying different O₃ exposure metrics (e.g., 24-h daily averages, median daily maximum 8-h concentrations, etc.). Some of the exposure metrics focused on the lower part of the distribution of hourly average concentrations (e.g., 24-h average concentrations), while other metrics focused on the upper part of the distribution (daily maximum 8-h average concentrations). Those investigators who used metrics focused on the higher concentrations reported *decreasing* O₃ concentrations during lockdown; those who applied metrics focused on the lower end of the distribution reported *increasing* O₃ concentrations. The lockdown that occurred during the spring in 2020 in the U.S. resulted in large emission reductions of O₃ precursors. The results described by Sommer et al. (2020) for U.S. sites in a National Public Radio analysis, as well as supplemented by the analysis of 52 U.S. O₃ monitoring sites described in these comments, illustrate the decrease in daily maximum 8-h average concentrations at many locations. One investigator, using exposure metrics not focused on the higher hourly average concentrations during the lockdown period, reported mixed patterns for 28 O₃ monitoring sites in the U.S. As noted, some of the researchers who characterized O₃ monitoring sites around the world during lockdown applied metrics focused on the lower end of the distribution. Some of these investigators reported that increasing O₃ concentrations appeared to be related to the reduction of NO titration of O₃. The result of this “natural experiment” provides additional evidence that emission reductions result in increases in the less biologically important lower hourly average concentrations. The results of the “natural experiment” during the COVID-19 lockdown support the second fundamental principle that **Daily Maximum Hourly Averaged O₃ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide.** (Section 3).
10. The compression of the distribution of hourly average O₃ concentrations results in *annual average or median concentration* values *increasing* at some sites. For example, in Fig. ES-4, the annual 4th highest daily maximum 8-h concentration for 2000-2021 illustrates the trend patterns in O₃ concentrations for five sites. As expected, the southern California and the New York sites show declines over time in the 8-h metric. The three rural National Park sites at Denali National Park (AK), Voyageurs National Park (MN), and Yellowstone National Park (WY) experience 8-h O₃ exposures lower than the two urban sites. When the annual average is plotted (Fig. ES-5) for the same period with the same data, the ordering of the sites from the highest to the lowest annual average concentrations shows a different pattern. While the Simi Valley site in southern California experiences the highest 8-h average O₃ exposures of the 5 sites, the annual average concentration for the southern California site is comparable to values for the three rural National Park sites. The New York site experiences the lowest annual average

exposure. Clearly, the ordering of the sites from highest to lowest exposures observed when using the 8-h metric is much different than the ordering when the annual average index is used. The increase in the annual average concentration values, even though emissions are being reduced, is associated with the low end of the distribution increasing due to less titration of O₃ by NO. The three National Park sites in the figures do not experience high 8-h average concentration values comparable to many of the urban sites in the U.S. The high-elevation Yellowstone National Park site experiences much higher annual average values than any of the remaining 4 sites. The hourly average O₃ concentrations experienced at Yellowstone National Park (WY) are influenced by frequent occurrences of stratospheric tropospheric transport to the surface (STT-S), which is a naturally occurring process that contributes to background O₃ levels (Lefohn et al., 2001, 2011, 2012, 2014). When the Mann-Kendall nonparametric statistical test is applied (similar to test described in the second draft PA (EPA, 2023, page 2-17)), no trend has been observed at Yellowstone National Park for the 2000-2022 period using the annual 4th highest daily maximum 8-h average concentration metric. (Section 3).

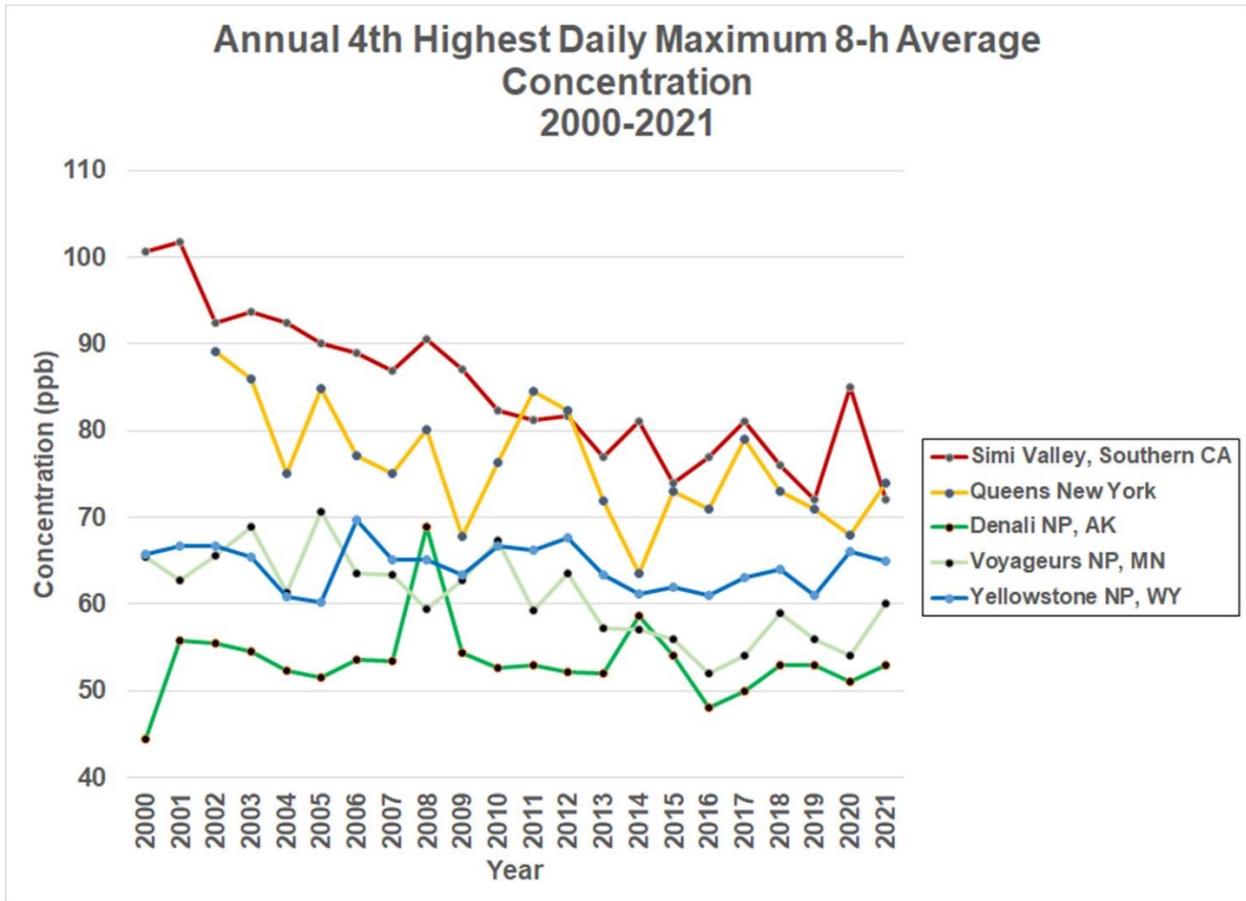


Figure ES-4. The annual 4th highest daily maximum 8-h average O₃ concentration for the period 2000-2021 for Simi Valley, CA (061112002), Queens New York (360810124, Denali National Park, AK (020680003), Voyageurs National Park, MN

(271370034), and Yellowstone National Park, WY (560391011). Source of data is from the EPA’s AQS database.

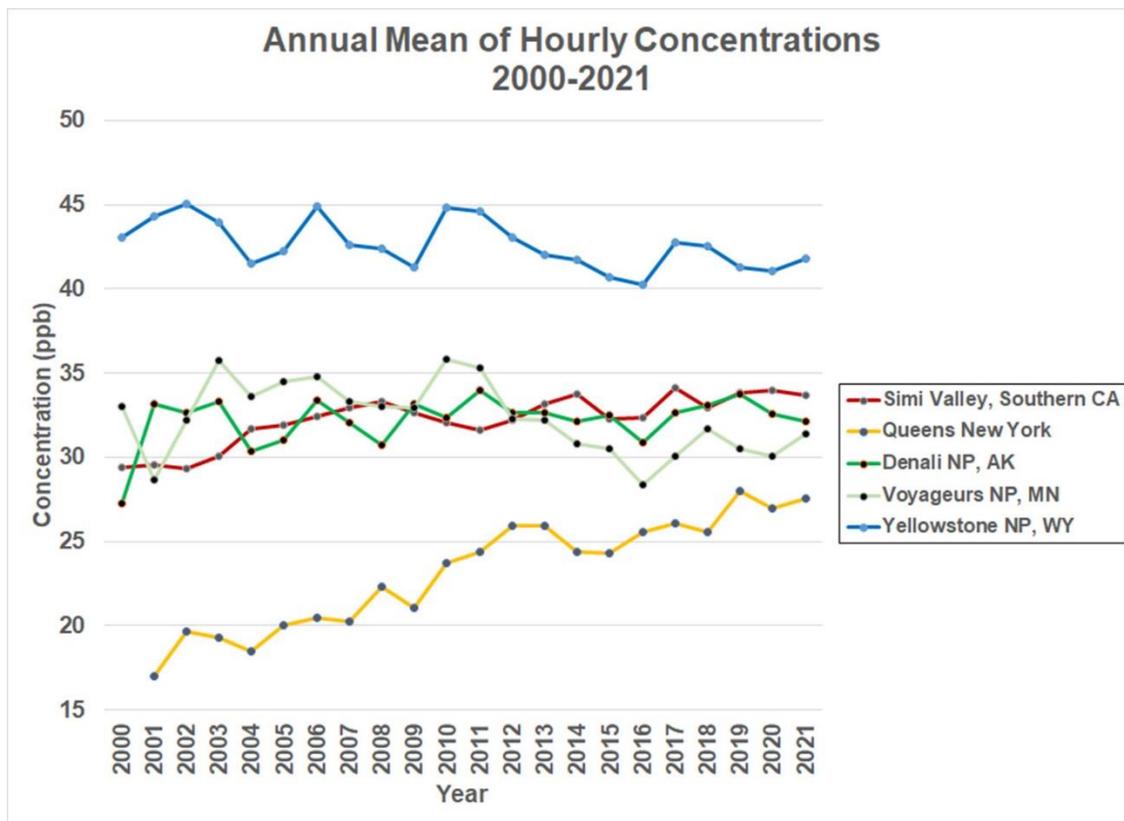


Figure ES-5. The annual mean of the hourly average O₃ concentrations for the period 2000-2021 for Simi Valley, CA (061112002), Queens New York, NY (360810124), Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011). Source of data is from the EPA’s AQS database.

11. One does not necessarily have to select an annual average or seasonal average concentration metric to estimate “chronic” (i.e., long-term) human health effects from O₃ exposures. Based on empirical data, annual average concentrations increase as emissions are reduced. The biologically important higher hourly average O₃ concentrations are reduced as emissions are reduced. In its review of the 2015 O₃ NAAQS rulemaking, the Administrator noted that an 8-hour averaging time remained appropriate for addressing health effects associated with short-term exposures to ambient air O₃ and that it could effectively limit health effects attributable to both short- and long-term O₃ exposures. (Section 5).

12. The use of the W126 metric as a secondary standard to protect vegetation has received strong support from CASAC in previous reviews (Henderson, 2006; Samet, 2010; Frey, 2014; Cox, 2020). Samet (2010), in summarizing CASAC’s comments on the

reconsideration of the O₃ NAAQS by the Obama Administration, noted that in recommending the W126 for the secondary welfare standard, the Agency acknowledged the distinction between the effects of acute exposures to O₃ on human health and the effects of chronic O₃ exposures on welfare, namely that vegetation effects were more dependent on the cumulative exposure to, and uptake of, O₃ over the course of the entire growing season. At the time, CASAC pointed out that the Agency was responding to the clear need for a secondary standard that was different from the primary standard in averaging time, level, and form. In 2010, the EPA proposed to set the level of a proposed W126 secondary standard within the range of 7-15 ppm-hours (ppm-hrs). In its 2015 O₃ NAAQS decision, the Administrator chose to use the current form of the human health standard and not the W126 index as the form and averaging time for the secondary standard. (Section 4).

13. After publication of the final rule in 2015, several industry groups, environmental and health organizations, and certain states filed petitions for judicial review in the D.C. Circuit. The industry and state petitioners argued that the revised standards were too stringent, while the environmental and health petitioners argued that the revised standards were not stringent enough to protect public health and welfare as the Act requires. In August 2019, the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) remanded the 2015 secondary standard to the EPA for further justification or reconsideration (US Court of Appeals, DC Circuit, August 23, 2019). As noted in the second draft PA (EPA, 2023, page 4-2),

In the August 2019 decision, the court held that EPA had not adequately explained its decision to focus on a 3-year average for consideration of the cumulative exposure, in terms of W126, identified as providing requisite public welfare protection, or its decision to not identify a specific level of air quality related to visible foliar injury. The EPA's decision not to use a seasonal W126 index as the form and averaging time of the secondary standard was also challenged, but the court did not reach a decision on that issue, concluding that it lacked a basis to assess the EPA's rationale because the EPA had not yet fully explained its focus on a 3-year average W126 in its consideration of the standard. Accordingly, the 2020 decision included discussion of these areas to address these aspects of the court's decision. (Section 4).

14. In the 2020 Review of the Ozone National Ambient Air Quality Standards document (Federal Register, 2020b), EPA noted that CASAC's use of the phrase "unusually damaging years" (Frey, 2014) focused on the use of the W126 index in the 2015 review when considering the form and averaging time for a revised secondary standard. CASAC was specifically concerned about the use of the 3-year average W126 versus the annual W126 index for the protection of vegetation from "unusually damaging years" (Frey, 2014). In its 2020 decision (Federal Register, 2020b), EPA expanded the term "unusually damaging years" to include concern for the control of high-concentration years. The EPA concluded in its final rulemaking of the O₃ NAAQS in 2020 that focusing solely on the W126 index (either in terms of a single year or 3-year average) in considering the public

welfare protection provided by the current standard *would not* be considering all the relevant scientific information. Thus, in its 2020 decision, the EPA expanded the focus of its prevention of “unusually damaging years,” as originally defined by CASAC (Frey, 2014), in terms of the level of the W126 index (either in terms of a single year or 3-year average) to also include concern for the control of the frequency of hourly or daily O₃ average concentrations ≥ 100 ppb. For this additional aspect, the second draft PA (EPA, 2023, page 4-9) notes that EPA considers air quality analyses of peak hourly concentrations in the context of considering protection against “unusually damaging years.” In Section 4.5 of the enclosed comments, a discussion is provided about the comparison between the elevated O₃ concentrations (i.e., hourly average values ≥ 100 ppb) observed across the U.S. today and the O₃ exposure regimes applied in the tree seedling experiments for the most sensitive species. The discussion provides additional insight into the important aspects associated with the concept introduced by CASAC in 2014 and expanded upon by the EPA in its 2020 decision on the O₃ NAAQS for the prevention of “unusually damaging years.” (Section 4).

15. As discussed in the second draft PA (EPA, 2023), Lefohn et al. (1997) documented the frequency of occurrence of elevated O₃ concentrations (i.e., number of hourly O₃ average concentrations ≥ 100 ppb - referred to as N100) in the tree seedling experiments for some of the experimental treatments. Since the reporting of this phenomenon in the peer-review literature, there has been discussion about the use of the W126 exposure index, coupled with the N100 metric, to provide a tighter relationship between (1) the exposure-response models developed in the crop and tree seedling experiments and (2) current ambient O₃ exposures for predicting vegetation effects. Lefohn et al. (1997) identified for the most sensitive species, at the 10% loss level, a 24-h W126 value of 5.9 ppm-hrs coupled with an N100 of 6 occurrences for the Southern Appalachian region. For its Air Quality Related Value (AQRV) for vegetation, the Forest County Potawatomi Community established the O₃ thresholds for the 3-year average of the 3-month (June, July, and August) 24-h cumulative W126 value at 7.0 ppm-hrs coupled with a 3-year average of the 3-month (June, July, and August) number of hours ≥ 100 ppb (N100) of 4 occurrences. The second draft PA (EPA, 2023, page 4A-34) summarizes the RBLs for 16 tree species analyzed by Lee et al. (2022). The seedling O₃ exposure studies used by Lee et al. (2022) for western and eastern tree species were conducted from 1988 to 1995 at the U.S. Environmental Protection Agency research laboratory in Corvallis, Oregon, Michigan Technological University’s Ford Forestry Center in Alberta, Michigan and by researchers from Appalachian State University at Great Smoky Mountains National Park near Gatlinburg, Tennessee. The 16 species are widespread across the U.S., are ecologically important, and include a variety of deciduous and coniferous, and faster and slower growing trees. The 3-month 12-h W126 estimated to result in a 5% biomass loss was 2.5-9.2 ppm-hrs for sensitive species, 20.8-25.2 ppm-hrs for intermediate species, and > 28.7 ppm-hrs for insensitive species. For the most sensitive tree species, the N100 values ranged from 0 to 7 for 12-h 92-day W126 exposures of 2.5-9.2 ppm-hrs. The authors noted that these species were ecologically important and widespread across the U.S. Lee et al. (2022) noted that while episodic peaks have been declining across the U.S. in the past few decades, there are still areas with N100 peaks where sensitive species occur. The authors believed that the exposure-response relationships reported in their

study remain relevant to current O₃ exposures, especially for the sensitive species. There exists evidence that 12-h W126 exposures at and below 7 ppm-hrs with N100 values that occur infrequently can result in biomass loss for sensitive tree species.

16. Tables 4A-7 and 4A-8 in the second draft PA (EPA, 2023, pages 4A-21 – 4A-26 and 4A-28 – 4A-30) show that W126 exposures associated with levels approximating 7 ppm-hrs experience less infrequent occurrences of hourly average O₃ concentrations \geq 100 ppb. For some of the tree seedling experiments, the N100 values were either 0 or close to 0 when the W126 value approximated 7 ppm-hrs. For the most sensitive tree species, Lee et al. (2022) in their study noted that the N100 values ranged from 0 to 7 for 12-h 92-day W126 exposures of 2.5-9.2 ppm-hrs exposures. While many of the tree seedling experiments experienced high numbers of N100 values in the highest treatments, such was not the case for those treatments that experienced the lower W126 values of interest (e.g., a W126 value of 7 ppm-hrs).

17. Fig. ES-6 from the second draft PA (EPA, 2023, page 4-69) illustrates the relationship between the W126 metric and the design values for the current 8-h O₃ NAAQS over the period 2018-2020. The W126 index is analyzed for both the 3-year design value period (left), as well as the annual period (right).

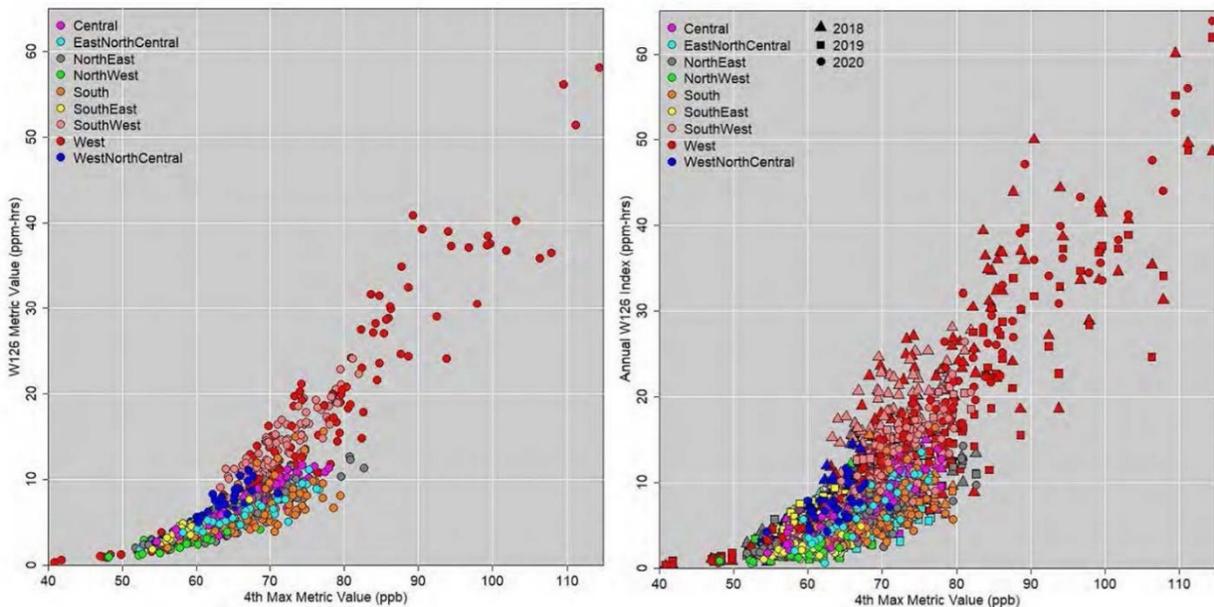


Figure 4-12. Relationship between the W126 index and design values for the current standard (2018-2020). The W126 index is analyzed in terms of averages across the 3-year design value period (left) and annual values (right).

Figure ES-6. Relationship between the W126 index and design values for the current standard (2018-2020). The W126 index is analyzed in terms of averages across the 3-year design value period (left) and annual values (right) (EPA, 2023, page 4-69).

Lee et al. (2022) noted that the 3-month 12-h W126 estimated to result in a 5% biomass loss was 2.5-9.2 ppm-hrs for sensitive species. At the 7 ppm-hrs annual level, Fig. ES-6 above (Fig. 4-12 in EPA (2023 page 4-69)), the right side of the figure shows that the range of values for the current 8-h form of the O₃ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the O₃ NAAQS of 0.070 ppm (i.e., 70 ppb), numerous occurrences greater than 7 ppm-hrs are evident and that these occurrences are not geographically isolated to the western part of the US (Fig. ES-6 above, right side). Therefore, at a W126 level of 7 ppm-hrs there does not appear to be a strong relationship between the annual W126 and the current form (3-year average) of 8-h O₃ NAAQS. The W126 metric and the 8-h O₃ NAAQS behave differently at levels of 7 ppm-hrs and below. A strategy of using the current O₃ NAAQS to limit annual W126 values to 7 ppm-hrs or below is questionable. (Section 4).

18. Based on its analyses described in Section 4 in the second draft of the PA (EPA, 2023), the Agency appears to reach the following conclusions for the secondary O₃ NAAQS that appear to be accurate based on the current state of science for the effects of O₃ exposures on vegetation:

- The EPA believes that the evidence supports a standard that protects against (1) cumulative growth impacts, such as those characterized by the W126 index, as well as against (2) high, peak hourly concentrations that are associated with visible foliar injury, as well as reduced tree growth. (See EPA (2023), page 4-131).
- The EPA believes that there are aspects of O₃ air quality not controlled by a cumulative index, such as the W126 index, that contribute to the risk of vegetation effects. These air quality aspects include occurrences of elevated hourly O₃ concentrations, which have demonstrated a role in effects such as reduced growth and visible foliar injury. (See EPA (2023), page 4-132).

19. Based on its analyses described in Section 4 in the second draft of the PA (EPA, 2023), the EPA appears to reach the following conclusions for the secondary O₃ NAAQS that it is suggested requires further modifications on wording to clarify the current state of science more accurately for the effects O₃ exposures on vegetation:

- The EPA believes that the W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. In its analysis, the EPA notes that some of the experimental exposures on which the RBL E-R functions are based upon W126 index values which are experienced in areas of the U.S. that meet the current standard. However, some of the experimental exposures include tens of hours at/above 100 ppb, which are not exhibited in current areas meeting the current 8-h standard. (See EPA (2023), page 4-133).

It is suggested that the current conclusion be modified as follows:

The W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. In its analysis, the EPA notes that some of the experimental exposures on which the RBL E-R functions are based upon W126 index values which are experienced in areas of the U.S. that meet the current standard. However, some of the experimental exposures include tens of hours at/above 100 ppb, which are not exhibited in current areas meeting the current 8-h standard. However, the experimental treatment W126 levels for the most sensitive tree species and thus, the most vulnerable vegetation, exhibit an infrequent number of hourly average O₃ concentrations ≥ 100 ppb at the 5% PRBL level. Lee et al. (2022) reported for the most sensitive species in their analysis, a biomass loss of 5% at W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures.

- The EPA believes that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. To achieve such control, in addition to control of cumulative exposures to protect against vegetation effects, the Agency concludes that if a secondary standard in the form of the W126 index were to be considered by the Administrator, the secondary standard should also be accompanied by an additional standard more focused on control of high hourly concentrations. (See EPA (2023), pages 4-132 – 4-133).

It is suggested that the current conclusion be modified as follows:

The EPA believes that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. To achieve such control, in addition to the control of cumulative exposures to protect against vegetation effects, the Agency concludes that if a secondary standard in the form of the W126 index were to be considered by the Administrator, the secondary standard should include the level of the cumulative W126 exposure index, coupled with the level of the number of elevated hourly average O₃ concentrations, the Administrator best believes would be protective of the most sensitive vegetation.

- The EPA believes that the body of evidence does not call into question the adequacy of the protection for vegetation provided by the current secondary standard. (See EPA (2023), page 4-134).

It is suggested that the current conclusion be modified as follows:

The EPA believes that the body of evidence calls into question the adequacy of the protection for vegetation provided by the current secondary standard. The EPA Administrator should consider a cumulative W126 exposure value selected at a level protective of the most sensitive vegetation, coupled with a

measure of the number of hourly average O₃ concentrations equal to or greater than 100 ppb (N100) and those levels be established at an annual W126 value of 7 ppm-hrs accompanied with an N100 value equal to or greater than 1.

20. A W126 value of 17 ppm-hrs would not adequately protect the most sensitive tree species, which was the W126 level the EPA had focused on in previous reviews. As noted in Figure ES-7 below, the most sensitive tree species (highlighted in yellow) identified by Lee et al. (2022) reported a biomass loss of 5% at W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures. (Section 4).

Table 4A-10. RBL for sixteen tree species (Lee et al., 2022), and the median, for W126 index from 30 to 7 ppm-hrs.

W126	Douglas Fir	Virginia Pine	Red maple	Sugar maple	Red Alder	Ponderosa Pine	Quaking Aspen	Tulip Poplar	Eastern White Pine	Black Cherry	American Sycamore	Winged Sumac	Sweetgum	Chestnut Oak	Table Mountain Yellow Buckeye	Median (16 species)	# of Species ≤ 2%	# of Species ≤ 5%	# of Species ≤ 10%	# of Species ≤ 15%	
30	2.9%	4.6%	7.1%	5.2%	15.4%	17.3%	25.9%	35.3%	4.2%	45.8%	19.7%	53.6%	25.7%	3.6%	2.5%	0.0%	11.2%	1	6	7	7
29	2.9%	4.4%	6.9%	5.1%	14.9%	16.7%	24.8%	34.3%	4.0%	44.7%	19.1%	51.1%	19.2%	3.5%	2.4%	0.0%	10.9%	1	6	7	8
28	2.8%	4.3%	6.7%	4.9%	14.4%	16.2%	23.7%	33.4%	3.9%	43.5%	18.5%	48.6%	14.1%	3.4%	2.3%	0.0%	10.4%	1	7	7	9
27	2.7%	4.1%	6.4%	4.7%	13.9%	15.7%	22.6%	32.4%	3.8%	42.4%	17.9%	46.0%	10.1%	3.3%	2.3%	0.0%	8.3%	1	7	7	9
26	2.6%	4.0%	6.2%	4.5%	13.4%	15.1%	21.5%	31.4%	3.6%	41.2%	17.3%	43.4%	7.1%	3.2%	2.2%	0.0%	6.7%	1	7	8	9
25	2.5%	3.8%	6.0%	4.4%	13.0%	14.6%	20.4%	30.4%	3.5%	40.0%	16.7%	40.8%	4.9%	3.0%	2.1%	0.0%	5.5%	1	8	8	10
24	2.4%	3.7%	5.7%	4.2%	12.5%	14.1%	19.3%	29.4%	3.3%	38.7%	16.1%	38.2%	3.4%	2.9%	2.0%	0.0%	5.0%	1	8	8	10
23	2.3%	3.5%	5.5%	4.0%	12.0%	13.5%	18.2%	28.3%	3.2%	37.5%	15.5%	35.6%	2.2%	2.8%	1.9%	0.0%	4.8%	2	8	8	10
22	2.2%	3.4%	5.3%	3.9%	11.5%	13.0%	17.2%	27.3%	3.1%	36.2%	14.8%	33.0%	1.5%	2.7%	1.8%	0.0%	4.6%	3	8	8	11
21	2.1%	3.2%	5.0%	3.7%	11.0%	12.4%	16.1%	26.2%	2.9%	34.9%	14.2%	30.5%	0.9%	2.6%	1.8%	0.0%	4.4%	3	8	8	11
20	2.0%	3.1%	4.8%	3.5%	10.5%	11.9%	15.0%	25.2%	2.8%	33.5%	13.6%	28.0%	0.6%	2.4%	1.7%	0.0%	4.2%	4	8	8	11
19	1.9%	2.9%	4.6%	3.3%	10.0%	11.3%	14.0%	24.1%	2.7%	32.1%	12.9%	25.5%	0.4%	2.3%	1.6%	0.0%	4.0%	4	8	8	12
18	1.8%	2.8%	4.3%	3.2%	9.5%	10.8%	13.0%	23.0%	2.5%	30.7%	12.3%	23.1%	0.2%	2.2%	1.5%	0.0%	3.8%	4	8	9	12
17	1.7%	2.6%	4.1%	3.0%	9.0%	10.2%	12.0%	21.8%	2.4%	29.3%	11.7%	20.8%	0.1%	2.1%	1.4%	0.0%	3.5%	4	8	9	12
16	1.6%	2.5%	3.9%	2.8%	8.5%	9.6%	11.0%	20.7%	2.2%	27.9%	11.0%	18.6%	0.1%	2.0%	1.3%	0.0%	3.3%	5	8	10	12
15	1.5%	2.3%	3.6%	2.6%	8.0%	9.0%	10.1%	19.5%	2.1%	26.4%	10.4%	16.4%	0.0%	1.8%	1.3%	0.0%	3.1%	5	8	10	12
14	1.4%	2.2%	3.4%	2.5%	7.5%	8.5%	9.1%	18.4%	2.0%	24.9%	9.7%	14.4%	0.0%	1.7%	1.2%	0.0%	2.9%	6	8	12	13
13	1.3%	2.0%	3.2%	2.3%	7.0%	7.9%	8.2%	17.2%	1.8%	23.3%	9.1%	12.4%	0.0%	1.6%	1.1%	0.0%	2.7%	7	8	12	13
12	1.2%	1.8%	2.9%	2.1%	6.4%	7.3%	7.3%	16.0%	1.7%	21.7%	8.4%	10.6%	0.0%	1.5%	1.0%	0.0%	2.5%	7	8	12	13
11	1.1%	1.7%	2.7%	1.9%	5.9%	6.7%	6.4%	14.7%	1.5%	20.1%	7.7%	8.9%	0.0%	1.3%	0.9%	0.0%	2.3%	8	8	13	14
10	1.0%	1.5%	2.4%	1.8%	5.4%	6.1%	5.6%	13.5%	1.4%	18.5%	7.0%	7.4%	0.0%	1.2%	0.8%	0.0%	2.1%	8	8	13	14
9	0.9%	1.4%	2.2%	1.6%	4.9%	5.5%	4.8%	12.2%	1.3%	16.8%	6.4%	6.0%	0.0%	1.1%	0.8%	0.0%	1.9%	8	10	13	14
8	0.8%	1.2%	2.0%	1.4%	4.3%	4.9%	4.0%	10.9%	1.1%	15.1%	5.7%	4.7%	0.0%	1.0%	0.7%	0.0%	1.7%	9	12	13	14
7	0.7%	1.1%	1.7%	1.2%	3.8%	4.3%	3.3%	9.6%	1.0%	13.3%	5.0%	3.6%	0.0%	0.9%	0.6%	0.0%	1.5%	9	13	14	15

Figure ES-7. Relative biomass loss (RBL) estimates for sixteen individual tree seedlings and median at various W126 index values using data from Lee et al. (2022) (EPA, 2023, page 4A-34).

21. The Court of Appeals for the District of Columbia Circuit (DC Circuit) in its August 19, 2019, decision addressed arguments regarding considerations of background O₃ concentrations, and socioeconomic and energy impacts. Regarding background O₃, the Court rejected the argument that the EPA was required to take background O₃ concentrations into account when setting the NAAQS. The Court found that the text of the Clean Air Act section 109(b) precluded this interpretation because it would mean that

if background O₃ levels in any part of the country exceeded the level of O₃ that is requisite to protect public health, the EPA would be obliged to set the standard at the higher nonprotective level. Thus, the Court concluded that the EPA did not act unlawfully or arbitrarily or capriciously in setting the 2015 NAAQS without regard for background O₃. While it is believed that background O₃ currently is not a consideration in the setting of the level of the O₃ standard, background O₃ plays an important role in influencing human health effects risk assessments. The human health risk and exposure assessments play an important role in the margin of safety determinations. Background O₃ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total ambient O₃ levels and potentially can influence those human health risk assessments associated with the margin of safety determinations for the setting of the primary O₃ NAAQS. (Section 3).

22. As emissions are reduced, besides the compression of the high and low concentrations toward the mid-level values, models predict that the highest concentrations, which normally have occurred in the past during the summer months, shift at some sites in the U.S. from the summer months to the March-June months. Besides modeling results, the data in the EPA's AQS database indicate sites where maximum concentrations shift from summer to spring months. There are also sites across the U.S. where the highest O₃ exposures occur in the spring independent of emission reductions. These observations have important ramifications for assessing the validity of background O₃ modeling estimates. (Section 3).
23. At sites influenced by natural processes involving the stratosphere (i.e., stratospheric-tropospheric transport to the surface (STT-S)), there is a tendency for the highest O₃ exposures to occur during the spring months (i.e., March, April, May, and mid-June). However, some sites (e.g., high-elevation sites) may experience STT-S contributions throughout the year. This observation has important ramifications for assessing the validity of background O₃ modeling estimates. (Section 3).
24. In the ISA (EPA, 2020a, page 1-53), EPA states that background O₃ seasonal and monthly means of hourly data are also included because longer averaging times are relevant for assessments of human health and ecological effects. This statement is not accurate. In many cases, assessment of human health and ecological effects are not based on longer averaging times. For the vegetation related W126 exposure index, which is a cumulative metric rather than an average exposure index, hourly average concentrations are weighted using a sigmoidal function and then accumulated over a specific period for assessing risk. Hourly average background O₃ concentrations contribute to the observed concentrations and therefore, contribute to the cumulative risk. For some human health risk assessments, at times daily 8-h average concentrations are used in a time series. Daily maximum 8-h average concentrations contain background O₃ concentrations, which contribute to the estimated human health risk assessment. (Section 3).
25. The authors note that the term **US background (USB)** is used to assess background O₃. Background O₃ is defined using the zero-out approach rather than other modeling methodologies. Simplicity of interpretation and consistency with previous analyses

appear to be the reasons that USB rather than apportionment based USB_{AB} was used in the modeling described in the second draft PA (EPA, 2023). A key point made in the ISA (page 1-56) is that the difference between USB and USB_{AB} is small in remote areas most strongly affected by USB sources but *can be substantial in urban areas strongly affected by anthropogenic sources that influence both production and destruction of O₃* (Dolwick et al., 2015). The selection of USB rather than apportionment-based U.S. background (USB_{AB}), as well as not performing bias adjustments to the USB estimates, appear to be important concerns about the adequacy of the background O₃ modeling results presented in the second draft PA (EPA, 2023). The USB estimate is a prediction of what concentrations could be without U.S. anthropogenic emissions. USB cannot be used to estimate the fraction of background O₃ in an observed O₃ concentration. (Section 3).

26. The 2013 Ozone ISA (EPA, 2013) reported higher seasonal mean USB and NAB concentration estimates in spring than in summer for most regions of the U.S. EPA notes in the current ISA (EPA, 2020a, page 1-65) that while some new results are consistent with this pattern, other results suggest that summer USB O₃ concentrations can be comparable to or greater than spring concentrations. The ISA did not resolve the conflicting conclusions about when seasonal mean background O₃ is greatest. Data in the EPA's AQS database indicate sites where maximum concentrations have shifted from summer to spring months. There are also sites across the U.S. where the highest O₃ exposures occur in the spring independent of emission reductions. At some National Park Service (NPS) sites, the highest O₃ exposures across the U.S. occur during the springtime (March to mid-June). The EPA (2014c, page 7A-12) provided the highest 3-month W126 values and the timeframe corresponding to those W126 exposures for the Parks for the period 2006-2010. Several of the O₃ monitors in the Parks experienced their highest W126 exposures during the spring months (defined as March, April, May or April, May, June) period. In the 2015 NAAQS rulemaking (Federal Register, 2015 – page 65416), the EPA determined that the lengthening of the O₃ monitoring seasons in 32 states and the District of Columbia was appropriate. The Agency indicated that ambient O₃ concentrations in these areas could approach or exceed the level of the NAAQS, more frequently and during more months of the year compared with the length of the O₃ seasons prior to 2015. The EPA described the results of its analysis (Rice, 2014) and extended the seasons for specific states and the District of Columbia. In Section 3, additional material is provided that identifies the March-June occurrences of the highest O₃ exposures. The pattern when the highest exposures occur is important for validating model performance estimating background O₃ concentrations. There continues to be strong evidence, as supported in the literature, that background O₃ across the U.S. is highest at many sites during the springtime (including into the month of June) and background O₃ is an important contributor at many high-elevation sites throughout the year. (Section 3).
27. The USB modeling results described in the second draft PA (EPA, 2023) indicate the following seasonal patterns: The natural contribution has a single maximum in late summer in the West, whereas, in the East there is evidence of two peaks—the largest in late Spring and a second peak in early Fall (page 2-48). The current analysis as described

in the second draft PA (EPA, 2023, page 2-64) indicates that natural and U.S. anthropogenic O₃ contributions peak during the traditional O₃ season (May through September), while long-range intercontinental transport of international O₃ (i.e., contributions from China, India, etc.) peaks in the spring (February through May). Reviewing Fig. ES-8, the total of the natural and international components shows for the West that March-August appears to be the period of highest total background O₃; for the East, the period March-June appears to be when the total highest background levels occur. The previous conclusion in the 2014 PA (EPA, 2014a) and the 2013 ISA (EPA, 2013, in section 3.4) was that background O₃ was greatest over the U.S. during the spring and early summer (i.e., March-June), which agrees with the Jaffe et al. (2018) conclusion. The different patterns in the West noted in the second draft PA (EPA, 2023) do not agree with the patterns described by Dolwick et al. (2015) and Lefohn et al. (2014). The difference may be attributable to the lack of bias adjustment in the EPA model described in the second draft PA. The second draft PA (EPA, 2023) noted that bias adjustment was not performed in the modeling described in the second draft PA. Lefohn et al. (2014) and Dolwick et al. (2015) performed bias adjustments. In their analyses, Lefohn et al. (2014) noted that model performance at low-elevation sites tended toward larger under prediction biases in the cool months (i.e., November-April) and larger over prediction biases in warm months (June-October), particularly for sites in the southern and eastern U.S. As is recognized today, hourly average concentrations associated with background O₃ can, at limited times and locations, be significantly higher because of stratospheric-tropospheric transport to the surface (Lefohn et al., 2011, 2012, 2014; Emery et al., 2012; Lin et al., 2012; Federal Register, 2015; EPA, 2020b). At many sites, stratospheric-tropospheric transport to the surface is important during the springtime. As noted in the second draft PA (EPA, 2023, page 2-66), the background O₃ modeling analysis did not attempt to quantify the contributions from individual Natural sources (e.g., lightning, soil, fires, stratosphere) or to address exceptional events beyond basic screening to remove very large fire plumes. The inability to adequately quantify the contribution of stratospheric-tropospheric transport to the surface for background O₃ may explain some of the discrepancy between the most current EPA background O₃ modeling results described in the second draft PA (EPA, 2023) and those reported in the 2014 PA (EPA, 2014a).

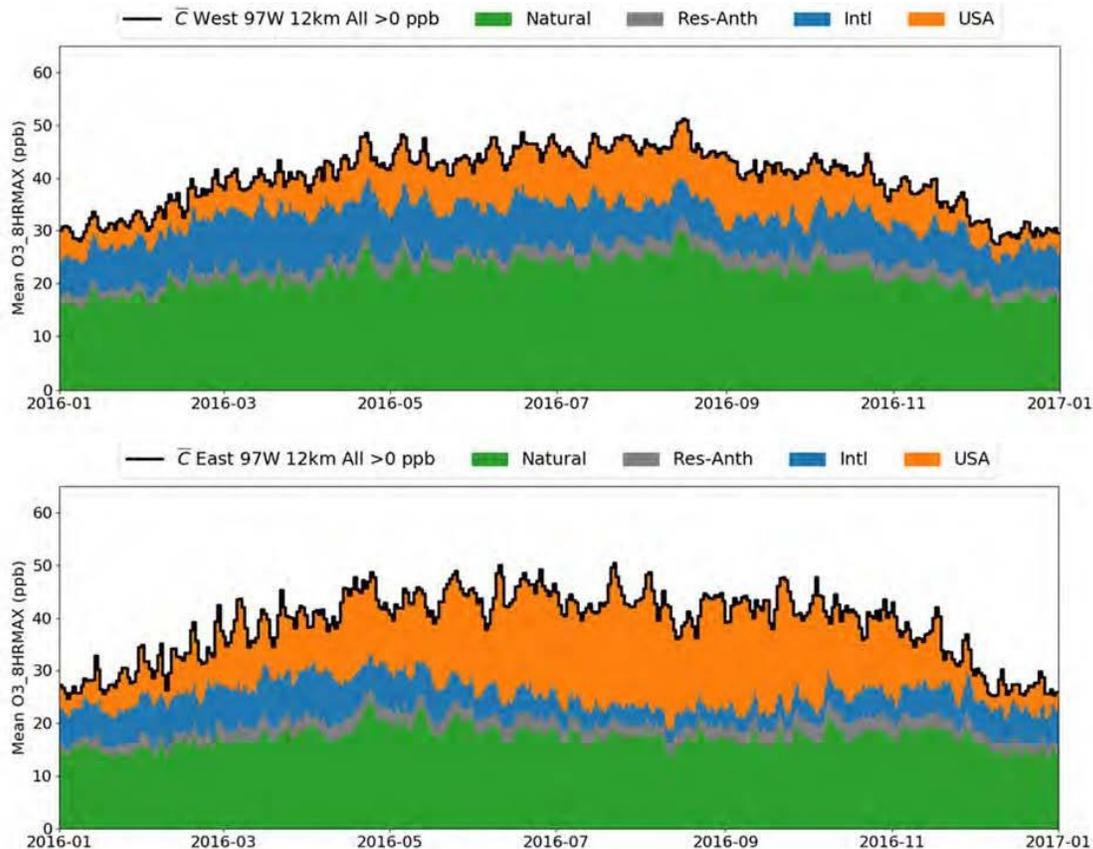


Figure ES-8. Annual time series of regional average predicted MDA8 total O₃ concentration and contributions of each source (see legend) for the West (top), and the East (bottom). Natural is global natural sources, Intl is international anthropogenic sources, USA is U.S. anthropogenic sources, and Res-Anth is the residual anthropogenic. (Source: EPA, 2023 Fig. 2-23, page 2-49).

28. Empirical data indicate that as emission reductions occur across the U.S., the higher MDA8 concentrations shift at many O₃ monitoring sites from the summer, toward the March-June months. In addition, as emission reductions occur, the distribution of hourly average concentrations shift from the higher values toward the middle values and the lower values shift upward toward the middle values. There is a compression of the distribution of hourly average O₃ concentrations. In addition, as emission reductions occur, background O₃ concentrations increase their percentage in the observed total O₃ concentration with the result that the compressed distribution of hourly average concentrations based on empirical data begins to resemble at some locations the distribution of background O₃. The patterns derived from empirical data showing the (1) compression of the distribution of hourly average O₃ concentrations and (2) seasonal shift from the summer months to the March-June period that result from emission reductions provide an opportunity to assess the adequacy of models that estimate background O₃ levels. Background O₃ is an important component of the margin of safety determinations. (Section 3).

29. The ISA (EPA, 2020a) attributes increasing trend patterns observed at high-elevation western U.S. sites to long-range transport from Asia. Long-range transport from Asia has not influenced trend patterns at all western U.S. high-elevation O₃ monitoring sites. Not all high-elevation western U.S. sites have exhibited statistically significant trends during the springtime, when transport is expected to be highest from Asia. An evaluation of trend patterns of high-elevation western U.S. sites during the springtime, using the 4th highest daily maximum 8-h concentration exposure metric, shows that some sites have not experienced increasing trends over the period 2000-2014. (Section 3).
30. Depending upon the specific monitoring sites, background O₃ contributes varying amounts to the higher hourly average O₃ concentrations. For example, the high-elevation Yellowstone National Park site in Wyoming is dominated by background O₃ throughout the year with minor anthropogenic contributions (Lefohn et al., 2014). In Fig. ES-9 below, the relative comparison of background O₃ levels (noted by blue) to anthropogenic (noted by red) within each concentration level shows that background contributes greater than 80% at this site across all ranges of concentrations. In comparison, Fig. ES-10 illustrates that for the Los Angeles area, a site heavily influenced by anthropogenic emissions, background O₃ contributes less than 40% at the higher hourly average concentrations. (Section 3).

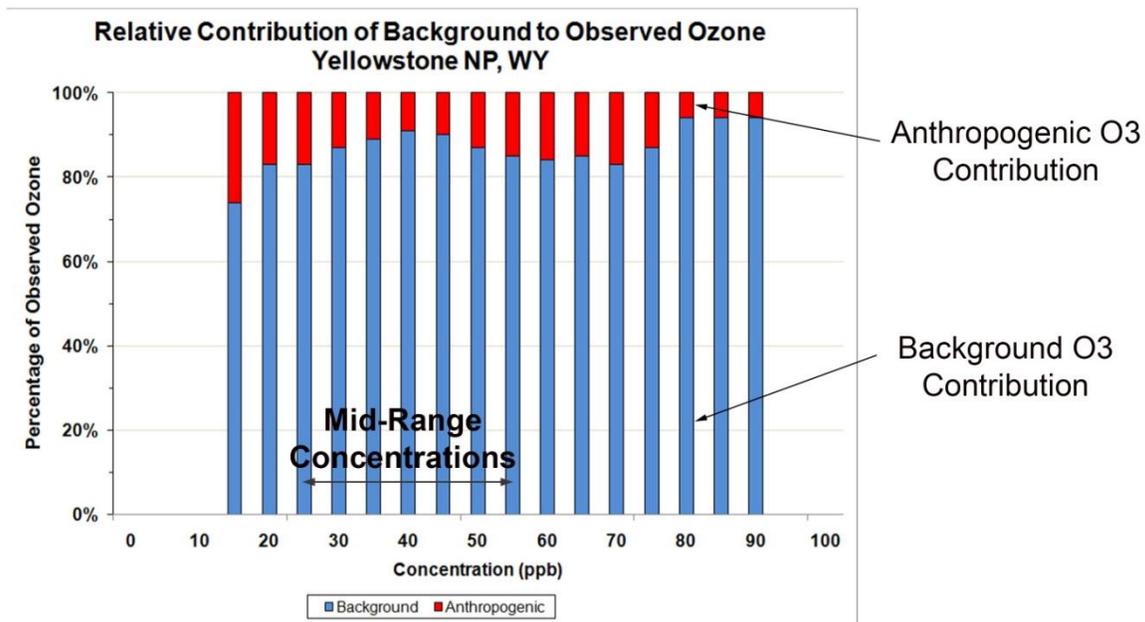


Figure ES-9. Average relative contributions of current hourly background (blue) and anthropogenic O₃ (red) for Yellowstone NP (WY) (AQS ID 560391011) in 2006. (Source: Lefohn et al., 2014).

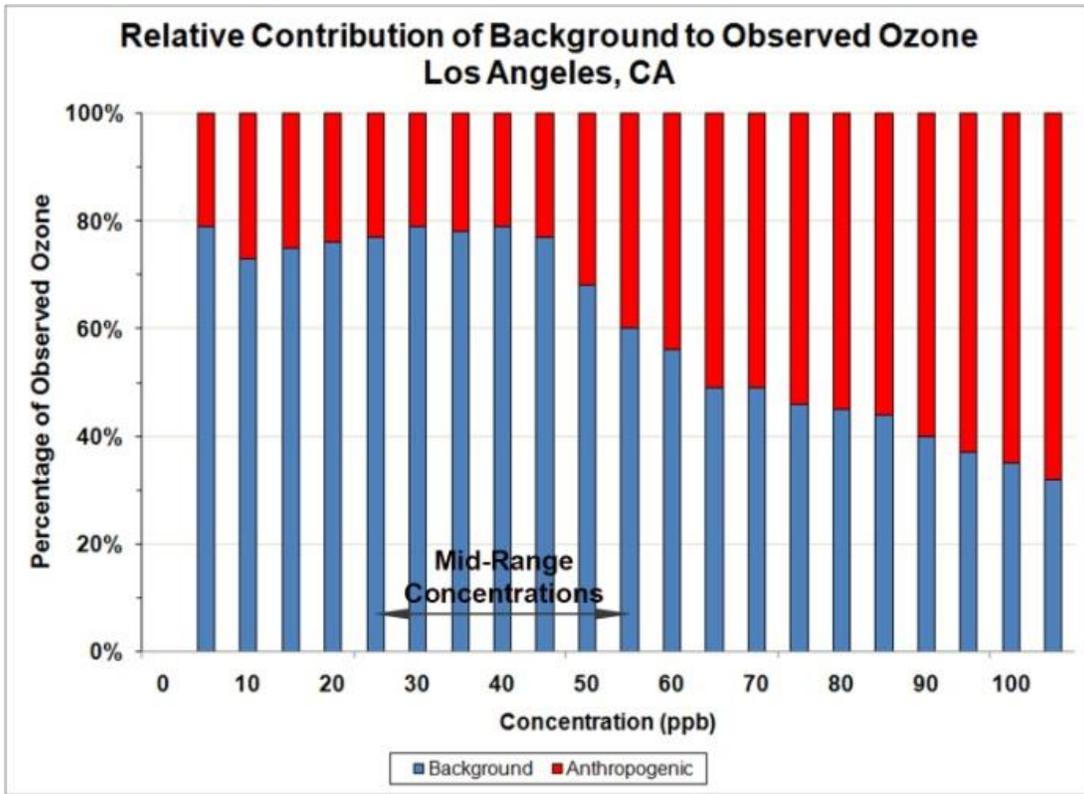


Figure ES-10. Average relative contributions of current hourly background (blue) and anthropogenic O₃ (red) for the Los Angeles (CA) (AQS ID 060719004) area in 2006. (Source: Lefohn et al., 2014).

31. In the second draft PA (EPA, 2023), analyses are presented that estimate exposure and risk for simulated populations in eight study areas (Atlanta, Boston, Dallas, Detroit, Philadelphia, Phoenix, Sacramento, and St. Louis). The eight study areas represent a variety of circumstances about population exposure to short-term concentrations of O₃ in ambient air. The eight study areas range in total population size from approximately two to eight million and are distributed across the U.S. in seven different NOAA climate regions: The Northeast, Southeast, Central, East North Central, South, Southwest, and West. The second draft PA did not provide examples of the time series for 2016 for the observed and USB concentrations for the eight sites used in the second draft PA (EPA, 2023) risk assessment. In Figs. ES-11 through ES-17, total observed O₃ concentrations, USB_{AB} estimates (data provided by the EPA), and STT-S counts are presented for 2007 for seven of the eight sites (i.e., Atlanta, Boston, Dallas, Detroit, Philadelphia, Sacramento, and St. Louis) used by the EPA in its risk analyses presented in the second draft PA (EPA, 2023). In the figures, gaps (i.e., the difference between the observed total O₃ (noted by the black line) and USB_{AB} concentrations (noted by the green line)) occur, indicating the apparent influence of anthropogenic sources. (Section 3).

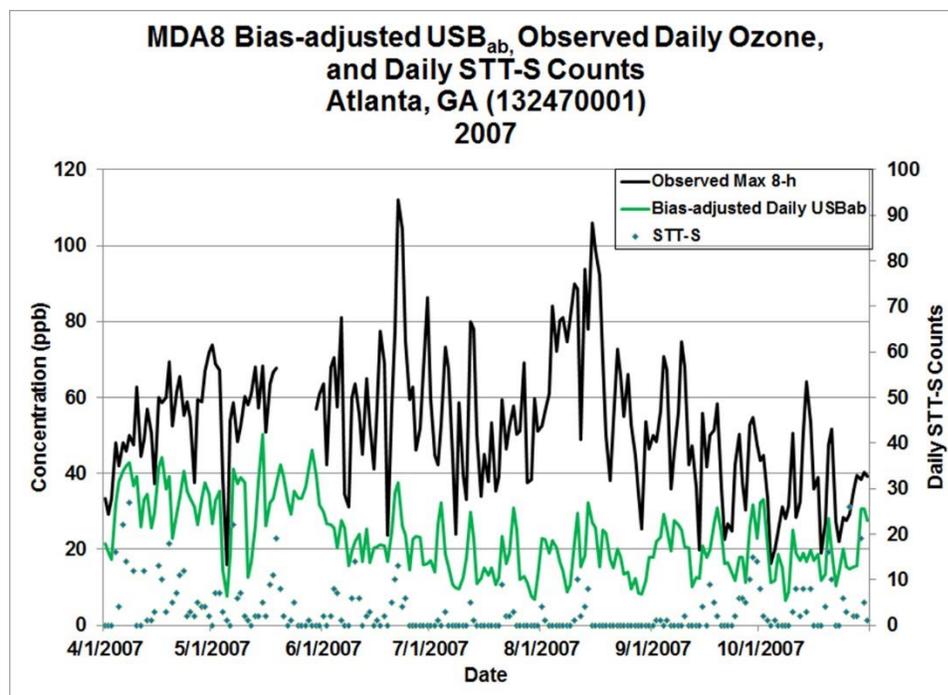


Figure ES-11. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Atlanta, Georgia (AQS ID 132470001) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O_3 values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

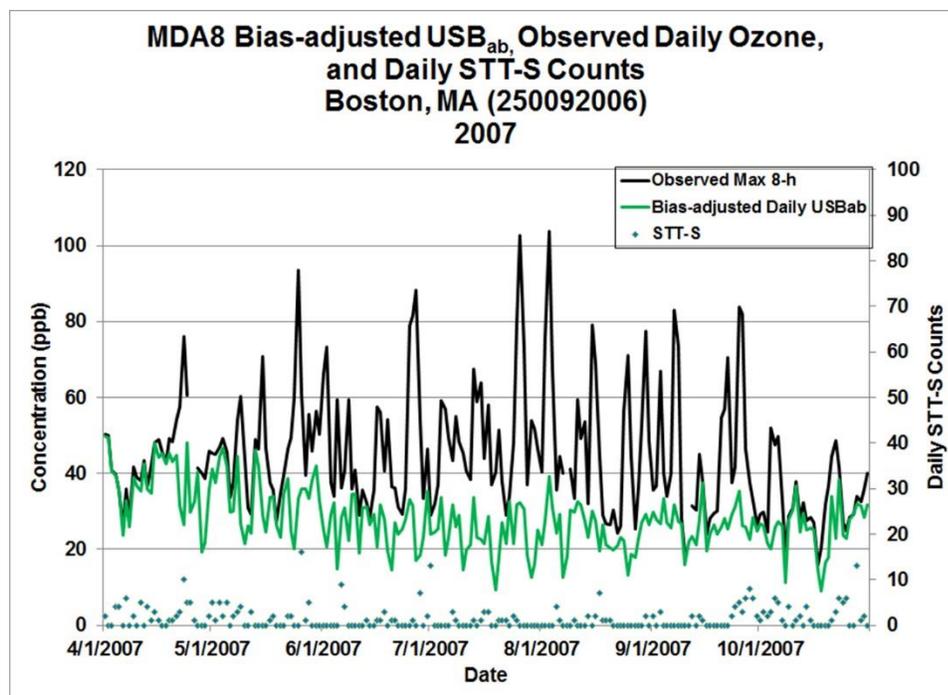


Figure ES-12. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Boston, Massachusetts (AQS ID 250092006) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O_3 values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

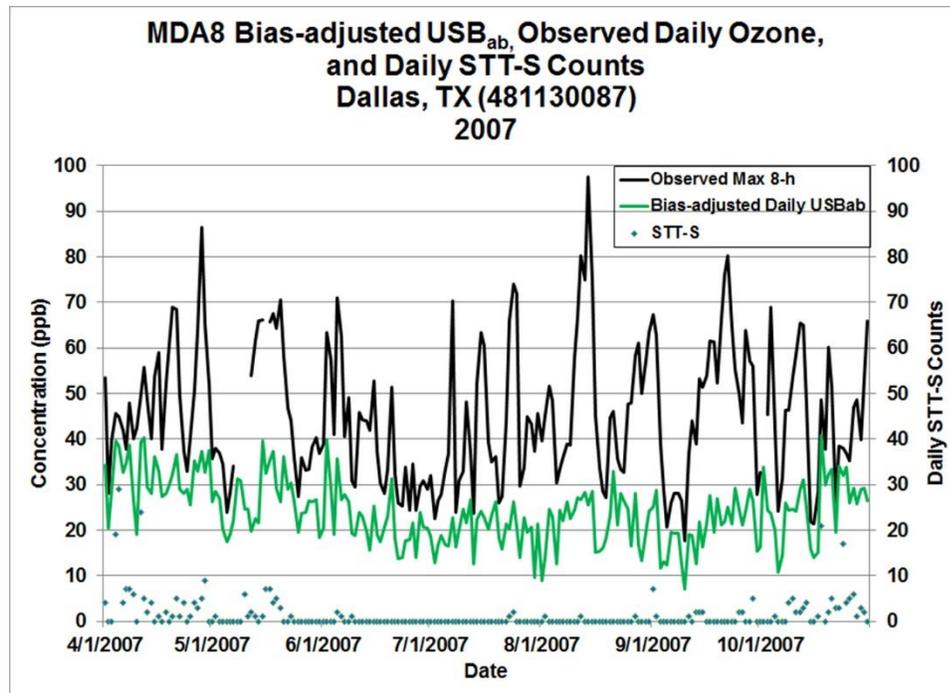


Figure ES-13. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Dallas, Texas (AQS ID 481130087) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O_3 values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

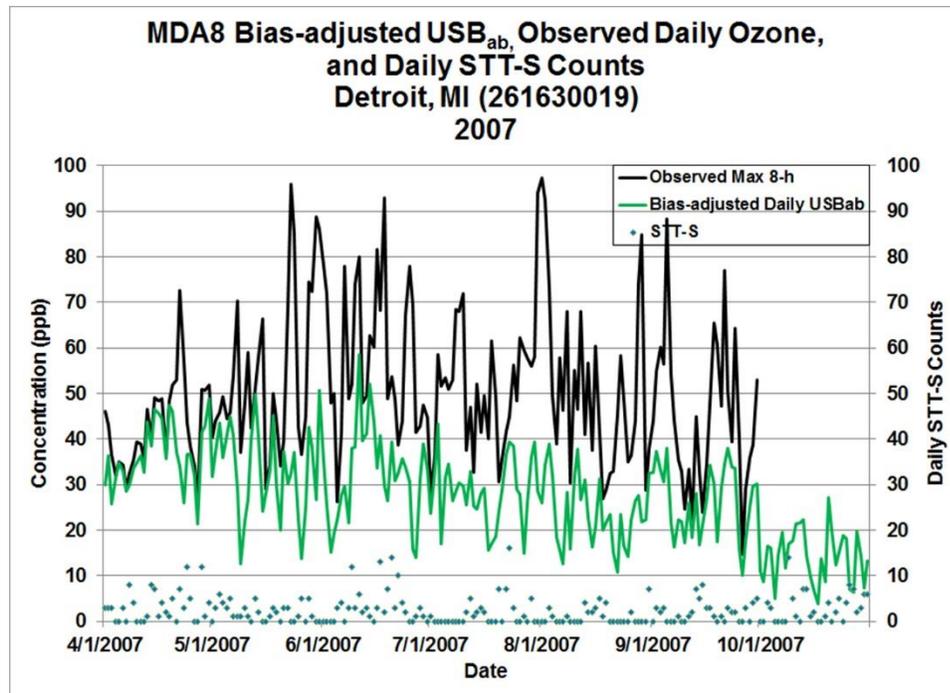


Figure ES-14. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Detroit, Michigan (AQS ID 261630019) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O_3 values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

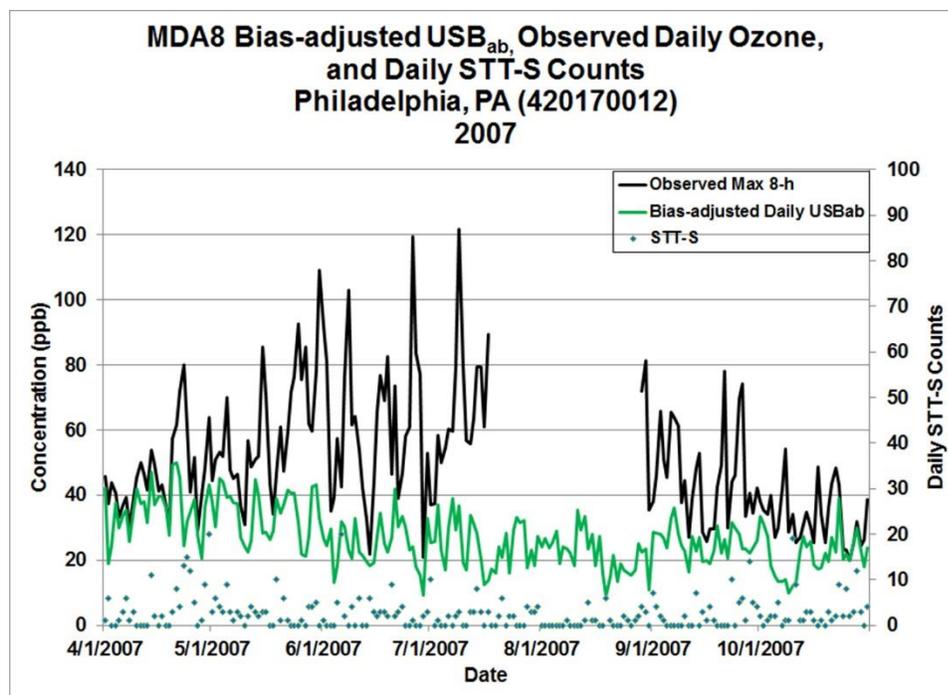


Figure ES-15. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Philadelphia, Pennsylvania (AQS ID 420170012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

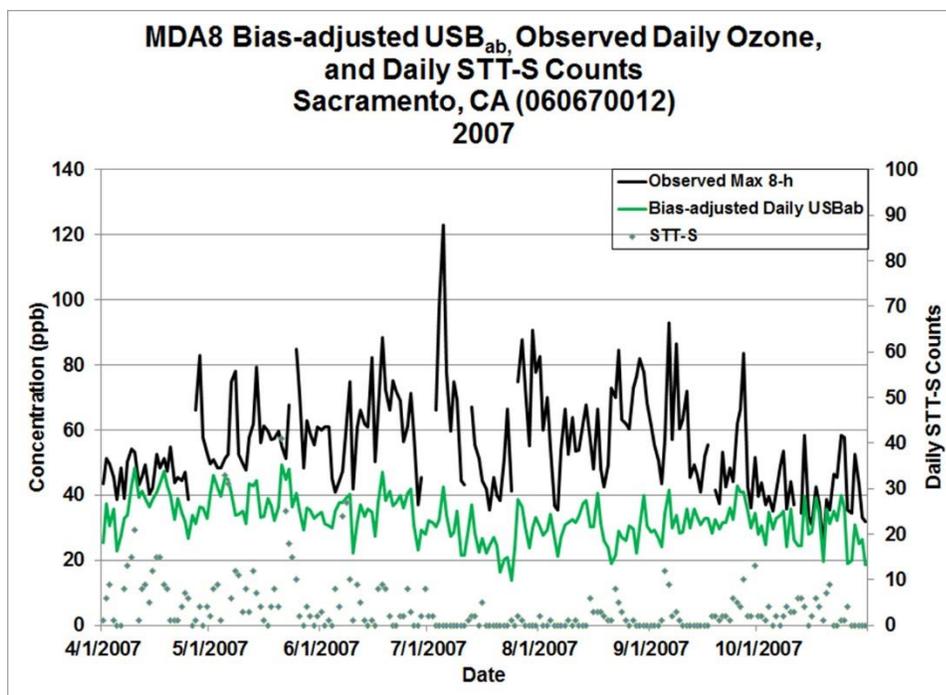


Figure ES-16. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Sacramento, California (AQS ID 060670012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

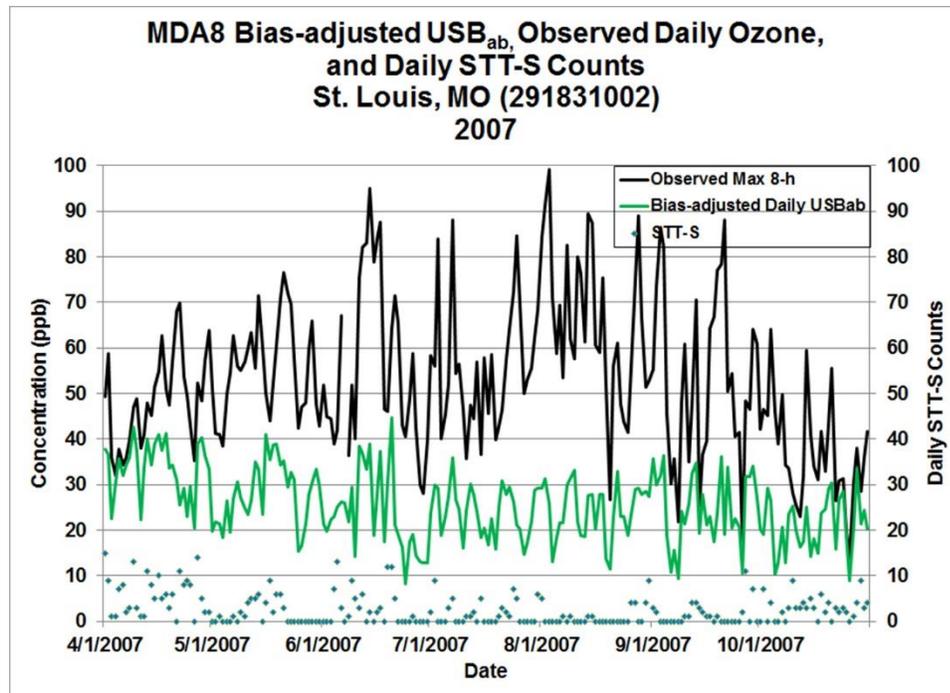


Figure ES-17. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in St. Louis, Missouri (AQS ID 291831002) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O_3 values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

1. Introductory Comments

The second draft Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards (PA) (EPA, 2023) document released in March 2023 was prepared as a part of the current reconsideration of the 2020 final decision on the national ambient air quality standards (NAAQS) for ozone (O₃). When final, the PA is intended to incorporate the scientific and technical information assessed in the 2020 Integrated Science Assessment for Ozone and Related Photochemical Oxidants (2020 ISA), as well as any air quality, exposure, and risk analyses available in the reconsideration, and the judgments required of the Administrator.

The review of the O₃ NAAQS, completed in 2015, established the current primary and secondary standards (80 FR 65291, October 26, 2015). As a part of its 2015 final decision (Federal Register, 2015 – page 65347, the EPA noted

In the 2008 review, the EPA focused on O₃ as the most appropriate indicator for a standard meant to provide protection against ambient photochemical oxidants. In this review, while the complex atmospheric chemistry in which O₃ plays a key role has been highlighted, no alternatives to O₃ have been advanced as being a more appropriate indicator for ambient photochemical oxidants. More specifically, the ISA noted that O₃ is the only photochemical oxidant (other than NO₂) that is routinely monitored and for which a comprehensive database exists (U.S. EPA, 2013, section 3.6). Data for other photochemical oxidants (e.g., peroxyacetyl nitrate, hydrogen peroxide, etc.) typically have been obtained only as part of special field studies. Consequently, no data on nationwide patterns of occurrence are available for these other oxidants; nor are extensive data available on the relationships of concentrations and patterns of these oxidants to those of O₃ (U.S. EPA, 2013, section 3.6). In its review of the second draft PA, CASAC stated “The indicator of ozone is appropriate based on its causal or likely causal associations with multiple adverse health outcomes and its representation of a class of pollutants known as photochemical oxidants” (Frey, 2014c, p. ii).

In addition, the PA notes that meeting an O₃ standard can be expected to provide some degree of protection against potential health effects that may be independently associated with other photochemical oxidants, even though such effects are not discernible from currently available studies indexed by O₃ alone (U.S. EPA, 2014c, section 4.1). That is, since the precursor emissions that lead to the formation of O₃ generally also lead to the formation of other photochemical oxidants, measures leading to reductions in population exposures to O₃ can generally be expected to lead to reductions in population exposures to other photochemical oxidants. In considering this information, and CASAC’s advice, the Administrator reached the proposed conclusion that O₃ remains the most appropriate indicator for a standard meant to provide protection against photochemical oxidants.¹¹⁵

Footnote 115 notes the following:

¹¹⁵The DC Circuit upheld the use of O₃ as the indicator for photochemical oxidants based on these same considerations. *American Petroleum Inst. v. Costle*, 665 F. 2d 1176, 1186 (D.C. Cir. 1981).

In December 2020, following a review of the scientific evidence associated with the current O₃ standards, the EPA issued its final decision to retain the existing standards without revision. The Agency noted that O₃ is the most appropriate indicator for a standard meant to provide protection against ambient photochemical oxidants (Federal Register, 2020b – page 87301). The EPA noted

As an initial matter, the Administrator recognizes the continued support in the current evidence for O₃ as the indicator for photochemical oxidants (as recognized in section II.D.1 of the proposal). As recognized in the proposal, no newly available evidence has been identified in this review regarding the importance of photochemical oxidants other than O₃ with regard to abundance in ambient air, and potential for health effects, and the “the primary literature evaluating the health and ecological effects of photochemical oxidants includes ozone almost exclusively as an indicator of photochemical oxidants” (ISA, p. IS-3). Accordingly, the information relating health effects to photochemical oxidants in ambient air is also focused on O₃. Thus, the Administrator concludes it is appropriate for O₃ to continue to be the indicator for the primary standard for photochemical oxidants.

The second draft PA (EPA, 2023, pages 3-102 – 3-103 and page 4-117) reaffirms EPA’s continued support for O₃ as the indicator for photochemical oxidants.

The primary and secondary NAAQS were first established for photochemical oxidants in 1971 based on the air quality criteria developed in 1970. The second draft PA (EPA, 2023, pages 1-6 – 1-16) summarizes the history associated with the O₃ NAAQS. In the second draft of the PA (EPA, 2023, page 1-10), the document summarizes the history associated with the current primary and secondary NAAQS as follows:

The EPA’s final decision in this review was published in October 2015, establishing the now-current standards (80 FR 65292, October 26, 2015). In this decision, based on consideration of the health effects evidence on respiratory effects of O₃ in at-risk populations, the EPA revised the primary standard from a level of 0.075 ppm to a level of 0.070 ppm, while retaining all the other elements of the standard (80 FR 65292, October 26, 2015). The EPA’s decision on the level for the standard was based on the weight of the scientific evidence and quantitative exposure/risk information. The level of the secondary standard was also revised from 0.075 ppm to 0.070 ppm based on the scientific evidence of O₃ effects on welfare, particularly the evidence of O₃ impacts on vegetation, and quantitative analyses available in the review.¹³ The other elements of the standard were retained. This decision on the secondary standard also incorporated the EPA’s response to the D.C. Circuit’s remand of the 2008 secondary standard in *Mississippi v. EPA*, 744 F.3d 1344 (D.C. Cir. 2013). The 2015 revisions to the

NAAQS were accompanied by revisions to the data handling procedures, and the ambient air monitoring requirements¹⁴ (80 FR 65292, October 26, 2015).¹⁵

After publication of the final rule, a number of industry groups, environmental and health organizations, and certain states filed petitions for judicial review in the D.C. Circuit. The industry and state petitioners argued that the revised standards were too stringent, while the environmental and health petitioners argued that the revised standards were not stringent enough to protect public health and welfare as the Act requires. On August 23, 2019, the court issued an opinion that denied all the petitions for review with respect to the 2015 primary standard while also concluding that the EPA had not provided a sufficient rationale for aspects of its decision on the 2015 secondary standard and remanding that standard to the EPA (*Murray Energy Corp. v. EPA*, 936 F.3d 597 [D.C. Cir. 2019]).

As noted above, in the decision on the litigation on the 2015 decisions, the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) remanded the 2015 secondary standard to the EPA for further justification or reconsideration (US Court of Appeals, DC Circuit, August 23, 2019). As noted in the second draft PA (EPA, 2023, page 4-2),

In the August 2019 decision, the court held that EPA had not adequately explained its decision to focus on a 3-year average for consideration of the cumulative exposure, in terms of W126, identified as providing requisite public welfare protection, or its decision to not identify a specific level of air quality related to visible foliar injury. The EPA's decision not to use a seasonal W126 index as the form and averaging time of the secondary standard was also challenged, but the court did not reach a decision on that issue, concluding that it lacked a basis to assess the EPA's rationale because the EPA had not yet fully explained its focus on a 3-year average W126 in its consideration of the standard. Accordingly, the 2020 decision included discussion of these areas to address these aspects of the court's decision.

In Section 4 of this review, detailed comments are provided on the adequacy of the 3-year average of the 4th highest daily 8-h maximum concentration exposure metric to control for the level of the W126 exposure index, as well as the frequency of elevated hourly average O₃ concentrations, over either an annual or a 3-year period to protect vegetation. In addition, comments are provided on the use of an annual W126 index by the EPA versus the application of a 3-year average of the W126 metric to protect vegetation.

In the second draft PA (EPA, 2023), the EPA appears to express concern about the frequency and level of elevated hourly average O₃ concentrations used in the vegetation experiments. The Agency noted that these levels were much higher than experienced in today's ambient exposures. The second draft PA (EPA, 2023) notes that across the full set of treatments, the number of hourly average O₃ concentrations \geq 100 ppb extended into the hundreds and up above 500 over 121 days. As will be discussed in Section 4, some of the experimental exposure protocols applied in both the National Crop Loss Assessment Network (NCLAN) and tree

seedling experiments were designed to explore the vegetation growth effects associated with episodic versus daily peak exposures; thus, providing scientific evidence for the importance of the higher hourly O₃ concentrations versus more moderate levels. While some of the simulated treatment exposures applied in these experiments were designed to mimic ambient exposures, other simulated profiles in these experiments were not designed to mimic ambient exposures. As indicated in the second draft PA (EPA, 2023), estimated exposures for the most sensitive tree species at a biomass loss of 5% appear to be associated with hourly average O₃ concentrations \geq 100 ppb from 0 to 7. Of the 1105 O₃ monitoring sites analyzed in the second draft PA (EPA, 2023), almost 25% experienced hourly average O₃ concentrations \geq 100 ppb for the 2018-2020 period. In Section 4, further discussion will focus on EPA's concern as noted above, as well as the scientific evidence for the EPA Administrator to establish a secondary O₃ NAAQS that uses the W126 exposure index, coupled with an additional metric that measures the number of hourly average O₃ concentrations \geq 100 ppb.

1.1 Purpose of the Policy Assessment (PA) Document

As indicated in the second draft PA (EPA, 2023, page 1-15),

This PA is being developed for consideration by the EPA Administrator in reaching his decision on the reconsideration of the December 2020 decision to retain the existing O₃ NAAQS. In assessing the policy implications of the available scientific information, this PA for the reconsideration, as for the 2020 PA, is intended to help “bridge the gap” between the Agency’s scientific assessment, presented in the 2020 ISA, and quantitative technical analyses, and the judgments required of the Administrator in determining whether it is appropriate to retain or revise the O₃ NAAQS. Accordingly, the PA for reconsideration will again address policy-relevant questions based on those identified in the 2018 IRP. With regard to considerations related to the primary standard, the PA for the reconsideration focuses on the evidence described in the 2020 ISA,²⁵ and the exposure/risk analyses presented in the 2020 PA, which will be included in full in this PA. With regard to considerations related to the secondary standard, the PA for reconsideration focuses on the evidence documented in the 2020 ISA, along with quantitative analyses presented in the 2020 PA and in subsequent technical memos, which have been updated to reflect recent air quality data. As described below, an additional publication related to quantitative relationships of O₃ exposure and tree seedling biomass is also considered.

In the second draft PA (EPA, 2023, page 1-16), the EPA has described the timetable associated with the decision-making processes linked with the reconsideration of the O₃ NAAQS. The Agency notes the following:

The EPA was initially targeting completing decision-making in this reconsideration by the end of 2023. Based on the steps that have occurred so far, the EPA has determined that it will need additional time to complete the

reconsideration process. The EPA now anticipates issuing a proposed decision in this reconsideration in Spring 2024, and it intends to work expeditiously to complete its decision-making in the reconsideration, using notice and comment procedures. While the timing of a final decision will depend on several factors that cannot be fully determined at this point, such as the volume and nature of public comments received on the proposed decision, the EPA anticipates that the reconsideration cannot be completed any more expeditiously than the end of 2024. This estimate reflects consideration of a number of circumstances, including the time that was necessary for the CASAC to complete its thorough and independent evaluation of the scientific issues in the 2020 ISA to inform its future review of the PA, and the time that the EPA needed to update the draft PA to reflect its consideration of the comments the CASAC had offered on the PA thus far. This estimate also reflects the time that the EPA currently anticipates will be needed for it and the CASAC to complete the remaining steps in this reconsideration, including a rigorous review of the second version of the draft PA by the CASAC; development of a final PA based on the EPA's consideration of the CASAC advice and public comments on the draft PA; development of a proposed decision and providing public notice and an opportunity for public comment on that proposal; and development of a final decision, after considering the public comments offered on the proposed decision.

Thus, according to the second draft PA (EPA, 2023), there are several additional steps, requiring a substantial amount of time, before the Administrator can decide on whether the current O₃ NAAQS are adequate or whether a revision is required to either the primary or the secondary or both O₃ NAAQS.

In the comments provided in this review, careful attention is focused on the technical aspects that the Agency has summarized in the second draft PA (EPA, 2023) from the (1) 2020 ISA (EPA, 2020a), (2) 2020 PA (EPA, 2020b), (3) 2020 O₃ NAAQS findings (Federal Register, 2020b), and (4) additional peer-reviewed publications that Agency believes are important for the updated second draft of the PA (EPA, 2023). It is important that the most relevant scientific results be accessible to the EPA Administrator so that he can reach his decision, based on the best available science, on the reconsideration of the O₃ NAAQS.

2. Important Concepts Guiding the Human Health and Welfare Ozone Standards

There are two key scientific fundamental principles that help guide the form and level of the health and welfare Federal O₃ standards in the United States. The first fundamental principle is

Higher Hourly Average O₃ Concentrations Should be Weighted More than Middle and Lower Values when Assessing Human Health and Environmental Effects.

The first fundamental principle will be discussed in this section. The second fundamental principle is

Daily Maximum Hourly Averaged O₃ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide.

The second principle will be discussed in Section 3.

Without adhering to the first fundamental principle, the selection of O₃ exposure indices for health and welfare assessment purposes would be based on the personal choice of the investigator rather than based on biologically relevant metrics which are founded on experimental or empirical observations. Lefohn et al. (2018), in the 24-coauthor international Tropospheric Ozone Assessment Report (TOAR) paper, discussed the scientific rationale, based on the peer-reviewed literature, for the selection of specific O₃ exposure metrics for assessing human health, vegetation, and climate change effects. The first fundamental principle provides guidance to policymakers and researchers on which part of the distribution of hourly average O₃ concentrations are biologically relevant for assessing human health and vegetation effects.

As noted in the Administrator's 2020 Review of the Ozone National Ambient Air Quality Standards document (Federal Register, 2020b – page 87267), the EPA Administrator noted that in 2015 the EPA Administrator in her decision set the human health standard at 70 ppb. The decision was based on the following rationale:

As for the decision on adequacy of protection provided by the combination of all elements of the existing standard, the 2015 decision to set the level of the revised standard at 70 ppb placed the greatest weight on the results of controlled human exposure studies and on quantitative analyses based on information from these studies, particularly analyses of O₃ exposures of concern, consistent with CASAC advice and interpretation of the scientific evidence (80 FR 65362, October 26, 2015; Frey, 2014b).³⁷ This weighting reflected the recognition that controlled human exposure studies provide the most certain evidence indicating the occurrence of health effects in humans following specific O₃ exposures, and, in particular, that the effects reported in the controlled human exposure studies are due solely to O₃ exposures, and are not complicated by the presence of co-occurring pollutants or pollutant mixtures (as is the case in epidemiologic studies) (80 FR 65362-65363, October 26, 2015). With regard to this evidence, the Administrator at that time recognized that: (1) the largest respiratory effects, and the broadest range of effects, have been studied and reported following exposures to 80 ppb O₃ or higher (i.e., decreased lung function, increased airway inflammation, increased respiratory symptoms, airway hyperresponsiveness, and decreased lung host defense); (2) exposures to O₃ concentrations somewhat above 70 ppb have been shown to both decrease lung function and to result in respiratory symptoms; and (3) exposures to O₃ concentrations as low as 60 ppb have been shown to decrease lung function and to increase airway inflammation (80 FR 65363, October 26, 2015). The Administrator also noted that 70 ppb was

well below the O₃ exposure concentration documented to result in the widest range of respiratory effects (i.e., 80 ppb), and below the lowest O₃ exposure concentration shown in 6.6 hour exposures with quasi-continuous exercise to result in the combination of lung function decrements and respiratory symptoms (80 FR 65363, October 26, 2015).

In controlled human health clinical studies (Hazucha et al., 1992; Adams 2003, 2006a, b), greater O₃ hour-by-hour peak responses were observed when applying stepwise and triangular (i.e., smooth increases and decreases in concentration) exposures rather than in the application of constant concentration exposure protocols. The focus by the EPA on the higher part of the distribution of the hourly average O₃ concentrations rather than the lower part provides clear guidance to those who are responsible for developing emissions reduction strategies to protect the public.

For vegetation, EPA reached the conclusion in 2015 that the higher concentrations should be weighted greater than mid and lower values (Federal Register, 2015 – page 65373). The Administrator noted the following:

The main conclusions from the 1996 and 2006 O₃ AQCDs [Air Quality Criteria Documents] regarding indices based on ambient exposure remain valid. These key conclusions can be restated as follows: ozone effects in plants are cumulative; higher O₃ concentrations appear to be more important than lower concentrations in eliciting a response; plant sensitivity to O₃ varies with time of day and plant development stage; [and] quantifying exposure with indices that cumulate hourly O₃ concentrations and preferentially weight the higher concentrations improves the explanatory power of exposure/response models for growth and yield, over using indices based on mean and peak exposure values.

The importance of the higher O₃ concentrations, coupled with the cumulative nature of the effects of O₃ on vegetation, is the basis for the EPA recommending the W126 exposure index (Lefohn and Runeckles, 1987; Lefohn et al., 1988) for assessing vegetation risk. The second draft PA (EPA, 2023, page 4-8) notes:

⁸ Both the 2020 and 2013 ISAs reference the longstanding recognition of the risk posed to vegetation of peak hourly O₃ concentrations (e.g., “[h]igher concentrations appear to be more important than lower concentrations in eliciting a response” [ISA, p. 8-180]; “higher hourly concentrations have greater effects on vegetation than lower concentrations” [2013 ISA, p. 91-4] “studies published since the 2006 O₃ AQCD do not change earlier conclusions, including the importance of peak concentrations, ... in altering plant growth and yield” [2013 ISA, p. 9-117]). While the evidence does not indicate a particular threshold number of hours at or above 100 ppb (or another reference point for elevated concentrations), the evidence of greater impacts from higher concentrations (particularly with increased frequency) and the air quality analyses that document variability in such concentrations for the same W126 index value led the Administrator to judge such a multipronged approach to be needed to ensure

appropriate consideration of exposures of concern and the associated protection from them afforded by the secondary standard (85 FR 87340, December 31, 2020).

As noted in the second draft PA (EPA, 2023, page 4-8), the Administrator in 2020 found it appropriate to continue to consider the seasonal W126 index averaged over a 3-year period to estimate median relative biomass loss (RBL). In addition, he judged it appropriate to also consider other exposure metrics, including peak hourly concentrations (Federal Register, 2020b). Further comment on the Administrator's use of the number of hourly average O₃ concentrations \geq 100 ppb (N100), as well as the number of days when hourly average O₃ concentrations are \geq 100 ppb (D100), will be discussed in detail in Section 4.

In this section, further information is provided that elaborates on the scientific evidence associated with the importance of the higher hourly average O₃ concentrations for assessing human health and vegetation effects.

2.1 Human Health

As noted, the second draft PA (2023) is developed for consideration by the EPA Administrator to reach a decision on the reconsideration of the December 2020 decision to retain the existing O₃ NAAQS. For the 2015 O₃ rulemaking, the EPA noted that higher O₃ concentrations have a proportionately greater impact than lower concentrations and therefore, are an important consideration in determining ozone's health impact. Important published results by Hazucha et al. (1992) and Adams (2003, 2006a, 2006b) formed the scientific foundation for the focus on higher O₃ concentrations. The work utilized ambient-type elevated concentrations and compared them to constant concentration exposures with the same concentration \times time product to assess respiratory effects. Commenting on these earlier results, Lefohn, Hazucha, Shadwick, and Adams (Lefohn et al., 2010a) concluded that higher O₃ concentrations are important in FEV₁ responses and that *the effect is cumulative* based on exposure. EPA notes in the ISA (EPA, 2020a, page 3-11) that greater peak responses have been observed in stepwise and triangular (smooth increases and decreases in concentration) exposures versus constant concentration exposure protocols. In addition, Hazucha et al. (1987), based on their meta-analysis, concluded that O₃ concentration exerts a greater influence on spirometric lung function decline than the intensity of exercise (i.e., minute ventilation). Conclusions from important papers discussing the importance of the higher hourly average O₃ concentrations and their influence on the form of the human health O₃ standard are as follows:

Hazucha et al. (1992)

Several recent studies have suggested that in estimating exposure dose (O₃ concentration [C] \times exposure time [T] \times ventilation [V]), O₃ concentration needs to be weighted more heavily than either ventilation or duration of exposure in the estimates. Our observations demonstrate that the product of \bar{C} (mean) \times T \times V is not a sufficient index of exposure.

Adams (2003, 2006a)

These results support previous evidence that O₃ concentration has a greater singular effect in the total inhaled O₃ dose than V_E and exposure duration.

Hazucha and Lefohn (2007)

As indicated in Hazucha et al. (1992), several recent studies have suggested that in estimating exposure dose (O₃ concentration [C] × exposure time [T] × ventilation [V]), O₃ concentration needs to be weighted more heavily than either ventilation or duration of exposure in the estimates. Our observations demonstrate that the product of \bar{C} (mean) × T × V is not a sufficient index of exposure. This formula, often referred to as an effective dose, has been used in several mathematical models to estimate O₃-induced lung function changes. Each term in the effective dose expression has an equal weight and the associations between the terms were assumed to be linear. Thus, a two-fold increase in any one parameter will double the effective dose. Early laboratory human exposure studies indicate that the linear relationship between the effective dose parameters could be extended to pulmonary function endpoints as well, specifically changes in FEV₁. The models based on this assumption, however, were suitable for qualitative estimation. only and did not provide consistent quantitative relationships (Silverman et al., 1976; Adams et al., 1981; Kulle et al., 1985). However, some of these studies already questioned the assumption of linearity of concentration–response relationship (e.g., Silverman et al., 1976). Subsequent studies employing various protocol designs have not only confirmed Silverman’s observations (Folinsbee et al., 1978) but also extended them further by demonstrating that concentration has a greater weight in eliciting spirometric response than minute ventilation (Adams et al., 1981). In a study designed to investigate the relative importance of O₃ concentration, ventilation, and exposure duration on lung function response, Drechsler-Parks et al. (1990) found that in acute exposure protocols, O₃ concentration was a dominant factor, followed by V_E and exposure duration in eliciting spirometric response.

Results from controlled laboratory exposures of human volunteers indicate that higher ozone (O₃) hourly average concentrations elicit a greater effect on hour-by-hour physiologic response (i.e., forced expiratory volume in 1 s [FEV₁]) than lower hourly average values, which implies a nonlinear dose–response relationship. To date, most of the empirical models derived from laboratory human experiments of concentration dose–response have been based on a constant exposure regime. The relationship between O₃ and spirometric lung function decrements is not linear. In attempting to derive the O₃ exposure–response relationship, we urge caution in curve-fitting exercises that focus on identifying the “best-performing” mathematical functions. Some of the functions identified may not be physiologically relevant. We recommend that investigators focus on identifying models that have biological plausibility and apply these models with data that are derived from variable exposure regimes. The use of a

logistic (i.e., sigmoid) model appears to be biologically justified. It is continuous, does not require the identification of a population threshold concentration, and deals with plateau considerations at the high end of the distribution of exposures.

The current 8-h average human-health O₃ standard does not appear to be consistent for predicting the hour-by-hour pattern of FEV₁ responses obtained following exposures to realistic concentration profiles. Because of nonlinearity, the elevated hourly average concentrations influence the observed responses in FEV₁ decrement shortly after peak O₃ exposures. Alternative forms of the O₃ standard might employ cumulative exposure indices that weigh the higher hourly average concentrations more than the mid- and low-level values. Logistic models based on data from variable exposures appear to perform fairly well and appear to be physiologically the most relevant for predicting human health dose–response effects.

Lefohn, Hazucha, Shadwick, and Adams (2010a)

Controlled human laboratory studies have shown that there is a disproportionately greater pulmonary function response from higher hourly average ozone (O₃) concentrations than from lower hourly average values and thus, a nonlinear relationship exists between O₃ dose and pulmonary function (FEV₁) response. We have reanalyzed data from five controlled human response to O₃ health laboratory experiments as reported by Hazucha et al. (1992), Adams (2003, 2006a, 2006b), and Schelegle et al. (2009). Our findings indicate a common response pattern across most of the studies. Schelegle et al. (2007) introduced a concept of a phased ventilatory response associated with O₃ exposures based on the frequency of breathing (fB) endpoint. In a subsequent paper, Schelegle et al. (2009) applied this concept to include an FEV₁ endpoint. Based on VAR/STW (i.e., variable/stepwise) FEV₁ response pattern, we have used a similar approach and identified three FEV₁ phases associated with exposure to VAR/STW O₃ concentrations: (i) a 2 to 3-h initial “induction phase” in response, (ii) followed by the onset of a statistically significant FEV₁ nonlinear “response phase,” and (iii) a final “reversal phase” (i.e., change in direction of the slope of the FEV₁ decrement towards baseline as the hourly average O₃ concentration is decreased). The first phase noted by Schelegle et al. (2009) more or less coincides with our Phase 1. Their second and third phases correspond to our second phase. We have added a new third phase, the “reversal phase,” which was absent in Schelegle et al.’s (2009) study.

During this post-induction phase, prolonged ambient type (i.e., VAR and STW) exposure studies find disproportionately greater responses from the higher hourly average O₃ concentrations and a nonlinear relationship than are found in SQW studies of equivalent O₃ dose (Hazucha & Lefohn 2007). Plausibly, once the antioxidant capacity is depleted, allowing for oxidative epithelial damage and related physiological response, other defense mechanisms are activated (e.g., reflex inhibition of inspiration) as reflected in more pronounced FEV₁ changes (Hazucha et al., 1992). During this phase, the FEV₁ response for the VAR and

STW profiles for the five studies illustrate the cumulative nature of the FEV₁ response to O₃. The term “cumulative response” refers to a monotonic decrement in FEV₁ as the O₃ exposure progresses. Other SQW studies by Folinsbee et al. (1988), Horstman et al. (1990), McDonnell et al. (1991), Hazucha et al. (1992), Adams and Ollison (1997), and Adams (2003, 2006a, 2006b) provide similar evidence for the cumulative nature of the FEV₁ response.

Results from these controlled human laboratory studies applying ambient pattern exposures (Adams 2003, 2006a, 2006b; Hazucha et al., 1992) illustrate the importance of the higher hourly average O₃ exposures compared to the lower hourly average values and a nonlinear relationship between O₃ dose and FEV₁ pulmonary function (Hazucha & Lefohn 2007).

Lefohn et al. (2018)

Controlled human exposure studies that explore induced decrements in lung function indicate that the higher ozone concentrations should carry greater weight than the moderate and lower concentrations (Hazucha and Lefohn, 2007; Lefohn et al., 2010a). Such studies vary the (1) intensity, duration and frequency of exercise from light to very heavy load on a treadmill or a bicycle ergometer to increased minute ventilation, (2) duration of exposures over 6.6-h and 8-h periods, and (3) application of varying hour-by-hour concentrations versus constant concentrations. In the 1980s and early 1990s, EPA investigators published the initial studies on the effects of 6.6-h exposures on healthy humans (Folinsbee et al., 1988; Horstman et al., 1990). In 1992, the first 8-h exposure study of ozone on lung function comparing the results using a constant concentration and variable concentration profile that mimicked typical diurnal patterns existing under ambient conditions was published (Hazucha et al., 1992). Both the constant and the variable concentration regimes used the same effective dose although the variable regime included exposure to high hourly average ozone concentrations. Compared to the square-wave exposure profile (i.e., constant concentration), the hourly lung function decrements in pulmonary function of subjects exposed to the variable concentration regime were substantially greater one hour after the peak exposure, with the conclusion that the higher concentrations should be weighted more than the mid- and low-level values. Several later studies (Adams 2003, 2006a, 2006b) employing either variable (continually changing) or stepwise (increasing or decreasing from one hour to the next) exposure profiles confirmed the results reported by Hazucha et al. (1992). These studies showed that equivalent doses (varying versus constant exposures) produced different hourly responses which depended on the applied hourly ozone concentration pattern.

The second draft PA (EPA, 2023, pages 3-40 and 3-41) notes that the exposure conditions given primary focus in the past several reviews are of the 6.6-hour study design that involve exposure maintained at a constant (unchanging) concentration for the full duration, although a subset of studies have concentrations that vary in a stepwise manner across the exposure period and are selected to achieve a specific target concentration as the exposure

average. The second draft PA (EPA, 2023, page 3-41 provides an example of the hour-by-hour O₃ concentrations applied in a stepwise controlled human exposure chamber study in footnote 55

⁵⁵ In these studies, the exposure concentration changes for each of the six hours in which there is exercise and the concentration during the 35-minute lunch is the same as the prior (third) hour of exercise. For example, in Adams (2006b), the protocol for a target exposure concentration average of 0.06 ppm for the 6.6-hour period is as follows: 60 minutes at 0.04 ppm, 60 minutes at 0.07 ppm, 95 minutes at 0.09 ppm, 60 minutes at 0.07 ppm, 60 minutes at 0.05 ppm and 60 minutes at 0.04 ppm.

The ISA (EPA, 2020a, page 3-11) notes the following:

Although greater peak responses have been observed in step-wise and triangular (smooth increases and decreases in concentration) exposures versus constant concentration exposure protocols, similar FEV₁ responses have been reported at 6.6 hours regardless of the exposure protocol (i.e., constant versus step-wise) for average ozone exposures to 60, 80, and 120 ppb (Adams, 2006, 2003a; Adams and Ollison, 1997).

Although this statement is accurate based on documenting the FEV₁ response after 6.6 hours, compared to the square-wave exposure profile, the hourly lung function decrements in pulmonary function of subjects exposed to the variable concentration regime were substantially greater one hour after the peak exposure. Results reported by Hazucha et al. (1992) and Adams (2003, 2006a, 2006b) approximated ambient conditions by using variable (VAR) or stepwise (STW) O₃ concentration patterns that peaked during the middle of their 6.6- and 8-h exposures. Hazucha et al. (1992) reported that exposure to a constant profile showed the largest decrement in FEV₁ near or at the end of the exposure. However, the triangular exposure profile induced the maximum effects about 2 hours following the peak concentration, well before the end of the exposure. The mean peak FEV₁ decrement was significantly different and almost twice as large as the one measured after the constant concentration profile exposure. The findings of the study confirmed that the usual metric of exposure, the dose (i.e., concentration multiplied by time and minute ventilation [V_E]) was not sufficient, and that O₃ concentration needed to be weighted more heavily in the predictive models of response than previously used indices.

The results of Hazucha et al. (1992), Adams (2003, 2006a, 2006b) produced different responses depending on the applied hourly O₃ concentration pattern and that the higher hourly average O₃ concentrations were more important than the lower values. Adams (2006a) described the results from a study that included six profiles (FA (0 ppb exposure)), two square wave (SQW (60 and 80 ppb)), and three stepwise (STW (40, 60, 80 ppb)) 6.6-h O₃ exposures. Table 2-1 illustrates the hourly concentration profiles used in Adams (2006a). At measurement time 4 (MT4), the STW 80 ppb treatment, when the hourly chamber O₃ average was 150 ppb, became significant in comparison to the filtered air treatment. The 6.6-hour average FEV₁ responses in the variable (or stepwise) studies were similar to the 6.6-hour average FEV₁ constant concentration studies. However, greater FEV₁ decrements occurred in the variable (or stepwise) studies shortly after peak O₃ exposures, as well as "recovery" occurred in contrast to the

continued response associated with the square-wave exposure studies. Fig. 2-1 illustrates the hourly FEV₁ decrement response in the Adams (2006a) study over 6.6 hours.

The hourly averaged concentrations were designed to increase in an “almost” symmetric manner. For the 80 ppb stepwise exposure, the peak hourly average concentration occurred at 4th hour and then began to decrease. The result was that the three FEV₁ phases described in Lefohn et al. (2010a) (Lefohn, Hazucha, Shadwick, and Adams), associated with exposure to the variable/stepwise O₃ concentrations occurred. The first phase was the 2 to 3-h initial “induction phase” in response. The second phase was the onset of a statistically significant FEV₁ nonlinear “response phase,” and the final phase was the “reversal phase” (i.e., change in direction of the slope of the FEV₁ decrement towards baseline as the hourly average O₃ concentration was decreased). Note that although the “reversal phase” occurred, the FEV₁ response remained statistically significantly different than the control at the end of the 6.6-hour experiment for several of the exposure regimes applied. The reversal phase was noted in the ISA (EPA, 2020a) and was presented as evidence supporting the statement on page 3-11 in the ISA that “...similar FEV₁ responses have been reported at 6.6 hours regardless of the exposure protocol (i.e., constant versus stepwise).” However, the hour-by-hour FEV₁ response showed substantial differences between the profile that mimicked ambient exposures and the constant concentration profile, indicating the importance of the higher hourly average O₃ concentrations.

Following the peak FEV₁ response, usually 1 to 2 h after the peak hourly O₃ concentration, lung function improved despite continuing lower hourly O₃ exposures. Results from the 80 ppb variable profiles applied in Adams (2003, 2006a) show a recovery as O₃ concentrations decline to 50 ppb. Similarly, results from Hazucha et al. (1992) and Adams (2006b) show reversal of FEV₁ response as the concentration was reduced from 60 to 0 ppb. The relatively rapid recovery observed during the variable exposures at 20 EVR possibly reflects replenishment or enhancements of airway antioxidant levels as well as decreased stimulation of lung receptors as O₃ hourly average concentrations decrease towards 50 ppb and lower.

Table 2-1. 6.6-h O₃ concentration (ppb) sequence used in Adams (2006a)

<i>Profile^a</i>	<i>EC†</i>	<i>Concentration</i>	<i>MT1*</i>	<i>MT2*</i>	<i>MT*3</i>	<i>MT4*</i>	<i>MT5*</i>	<i>MT6*</i>
1	Filtered Air	0 ppb	0	0	0	0	0	0
6	Stepwise	40 ppb	30	40	50	50	40	30
5	Stepwise	60 ppb	40	70	90	70	50	40
4	Square-wave	60 ppb	60	60	60	60	60	60
2	Square-wave	80 ppb	80	80	80	80	80	80
3	Stepwise	80 ppb	30	70	100	150	80	50

† Exposure condition.

^a Ozone regime number designated by the study author.

* Measurement time.

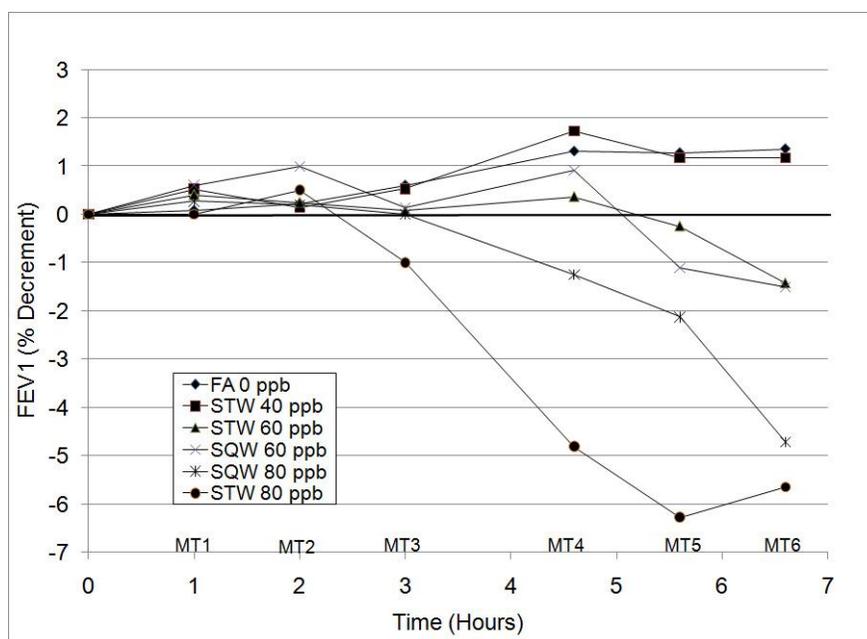


Figure 2-1. Ozone-induced FEV₁ changes as a function of O₃ exposure profiles. MT = measurement time. FA=Filtered Air, SQW=Square wave, and STW= Stepwise. Source: Lefohn et al. (2010a). Data were originally published in Adams (2006a).

The EPA recognizes in the 2020 ISA (EPA, 2020a, page 3-11) that greater peak FEV₁ responses occurred shortly after stepwise and variable exposures versus constant concentration exposure treatments. However, because similar FEV₁ responses were reported at the end of 6.6 hours, the Agency did not appear to highlight the importance of the higher hourly average O₃ concentrations that resulted in large FEV₁ decrements reported shortly after peak O₃ exposures during the various studies. The use of the FEV₁ response at the end of the 6.6 h study tends to obscure the variability of the hour-by-hour FEV₁ response following exposures to realistic concentration profiles.

The observation that greater FEV₁ decrements occurred shortly after peak O₃ exposures in the variable exposure regimes means that the 8-h standard may not be as protective as intended. As noted previously, Hazucha and Lefohn (2007) indicated that the current 8-h average human-health O₃ standard does not appear to be sensitive to the hour-by-hour pattern of FEV₁ responses observed following exposures to realistic concentration profiles. Compared to the constant exposure profile, the hourly lung function decrements in pulmonary function of subjects exposed to the variable (or stepwise) concentration regime were substantially greater one hour after the peak exposure. This appears to indicate that the 8-h average form of the O₃ standard may not be as protective as believed and may need to be revisited sometime in the future.

Because of nonlinearity, the elevated hourly average concentrations influence the observed instantaneous human health response (i.e., FEV₁ decrement). Hazucha and Lefohn (2007) suggested that alternative forms of the O₃ standard might employ cumulative exposure indices that weigh the higher hourly average concentrations more than the mid- and low-level

values. Logistic models, based on data from variable exposures, appear to perform well and appear to be physiologically the most relevant for predicting human health dose–response effects.

Conclusions from the Hazucha et al. (1992) and Adams (2003, 2006a) results illustrate that ***“controlled human laboratory studies have shown that there is a disproportionately greater pulmonary function response from higher hourly average ozone (O₃) concentrations than from lower hourly average values and thus, a nonlinear relationship exists between O₃ dose and pulmonary function (FEV₁) response”*** (Lefohn, Hazucha, Shadwick, and Adams, 2010). The implication of this is that a simple cumulative calculation of C x T (concentration multiplied by time) is not a valid determination of cumulative O₃ exposure. For understanding cumulative O₃ effects on FEV₁, in future controlled human health laboratory experiments, there should be varying hour-by-hour concentration regimes over the time of exposure, as well as more extensive research on the “induction” and “reversal” phases noted by previous investigators.

During the 2015 rulemaking O₃ activity, as well as the 2019 rulemaking activity (Federal Register, 2020b), one controlled human exposure experiment published was the key study that the EPA focused on for the selection of the primary O₃ standard. While other controlled human exposure and epidemiological studies were also available, the EPA focused on the Schelegle et al. (2009) study for setting the primary O₃ standard. Professor Milan Hazucha and I designed the hour-by-hour exposure regimes for the Schelegle et al. (2009) study. However, before discussing why this single controlled human exposure experiment outweighed the other studies, it is important to describe the Agency’s rationale in the 2015 O₃ rulemaking that provided the context for why, among all the studies in the published literature, this single experiment formed the basis for the current level of the O₃ standard to protect human health.

When evaluating the epidemiological and controlled human exposure study results, the higher O₃ concentrations were an important consideration (Federal Register (2015 – page 65354). The EPA (Federal Register, 2015 – page 65343) agreed with the conclusions of the 2014 Agency’s Policy Assessment Report (EPA, 2014a) that controlled human exposure studies provided the most certain evidence indicating the occurrence of health effects in humans following exposures to specific O₃ concentrations. Specifically, the Agency recognized that the effects reported in controlled human exposure studies were due solely to O₃ exposures. In contrast, epidemiological studies incorporate confounding factors that may have obscured the cause-effect relationship. The ISA (EPA, 2020a, page IS-1) indicated that the strongest evidence comes from controlled human exposure studies demonstrating O₃-induced decreases in lung function and inflammation in healthy, exercising adults. The second draft PA (EPA, 2023, page 3-105) notes that epidemiologic studies are limited in their ability to provide insights regarding exposure concentrations associated with health outcomes that might be expected under air quality conditions that meet the current standard. As noted in the second draft PA (EPA, 2023, page 3-105)

In considering what may be indicated by the epidemiologic evidence with regard to exposure concentrations eliciting effects, we recognize that of the numerous single-city epidemiologic studies of respiratory outcome associations with O₃ in

ambient air, none were conducted in U.S. locations during time periods when the current standard would have been met. In fact, the vast majority of single- and multi-city studies were conducted in locations and during time periods that would not have met the current standard, thus making them less informative for considering the potential for O₃ concentrations allowed by the current standard to contribute to health effects. While there were a handful of multi-city studies in which the O₃ concentrations in a subset of the study locations and for a portion of the study period may appear to have met the current standard, data were not available in some cities for the earlier years of the study period when design values for other cities were well above 70 ppb (as discussed in section 3.3.3). We recognize that the study analyses and associations reported were based on the combined dataset across the full time period (and, for multicity studies, from all cities), and the extent to which risk associated with exposures derived from the concentrations in the subset of years (and locations) that would have met the current standard compared to that from the years (and locations) that would not have met the standard influenced the study findings is not clear. There were no studies conducted in U.S. locations with ambient air O₃ concentrations that would meet the current standard for the entire duration of the study (i.e., with design values¹⁰⁷ at or below 70 ppb). Thus, the epidemiologic studies are extremely limited in their ability to provide insights regarding exposure concentrations associated with health outcomes that might be expected under air quality conditions that meet the current standard (section 3.3.3 above). Accordingly, the studies of 6.6-hour exposures with quasi-continuous exercise, and particularly those for concentrations ranging from 60 to 80 ppb continue to provide an appropriate focus in this reconsideration.

Thus, the EPA concluded in the second draft PA (EPA, 2023), as in previous PA documents published over the years, that the studies of 6.6-hour exposures with quasi-continuous exercise, and particularly those for concentrations ranging from 60 to 80 ppb continue to provide an appropriate focus in the current reconsideration of the primary O₃ NAAQS.

The EPA noted that controlled human exposure studies report the combination of lung function decrements and respiratory symptoms in healthy adults engaged in intermittent, moderate exertion following 6.6 hour exposures to concentrations as low as 72 ppb (based on Schelegle et al., 2009), and lung function decrements and pulmonary inflammation following exposures to O₃ concentrations as low as 60 ppb (based on Kim et al., 2011) (Federal Register, 2015 – page 65343). However, the EPA also noted that a level of 60 ppb is below the lowest concentration where the *combined* occurrence of respiratory symptoms and lung function decrements were observed (i.e., 72 ppb), a combination judged adverse by the American Thoracic Society according to the EPA (Federal Register, 2015 – page 65357). The EPA, noting the Kim et al. (2011) findings, had less confidence that health effects would occur below an O₃ concentration of 72 ppb. Thus, only one controlled human exposure study (i.e., Schelegle et al., 2009) informed the EPA's decision as to the level of the standard.

For the epidemiological studies, the EPA noted that the interpretation of studies' results was complicated by the presence of co-occurring pollutants or pollutant mixtures. In addition, in its 2015 review, the EPA placed less weight on epidemiologic-based risk estimates because of key uncertainties about (1) which co-pollutants was responsible for any health effect observed, (2) the heterogeneity in effect estimates between locations, (3) the potential for exposure measurement errors, and (4) uncertainty in the interpretation of the shape of concentration-response functions for O₃ concentrations in the lower portions of ambient distributions (EPA, 2023, page 3-62). As noted in the Health Risk and Exposure Assessment (HREA) (EPA, 2014b) with respect to the epidemiological studies, not differentially weighting the concentration-response functions at higher O₃ concentrations greater than the mid- and low-level values resulted in small differences in the estimates of mortality and morbidity risks as a theoretical modeling effort was made to impose more stringent standards. This occurred because as shown by EPA's modeling, as well as our and other researcher's trend results, as emissions are reduced to meet lower standards, the high end of the concentrations shifts downward (i.e., reducing mortality estimates) but the low end of the distribution of concentrations shifts upward (i.e., increasing mortality estimates), **resulting in a small net benefit**. Because of the limitations in the epidemiology studies, the EPA did not use these studies to set the proposed range for the level of the standard except for the margin of safety consideration.

As described above, the EPA (Federal Register, 2015 – page 65357) had less confidence that adverse effects would occur following exposures to O₃ concentrations below 72 ppb. With the findings of the epidemiological studies being discounted because of numerous uncertainties affecting the interpretation of the results, the EPA relied on a single study to focus on the setting of the 2015 O₃ standard. The EPA went on to conclude that a standard level as high as 70 ppb, which CASAC concluded could be supported by the scientific evidence, could reasonably be judged to be requisite to protect public health with an adequate margin of safety (Federal Register, 2015 – page 65363).

EPA's recognition of the importance of the higher O₃ concentrations is reflected in its Air Quality Index (AQI) reporting across the U.S. (EPA, 2018). Local air quality agencies are required to report air quality using the Air Quality Index (AQI) as required in 40 CFR Part 58.50 and according to 40 CFR Appendix G to Part 58. Metropolitan Statistical Areas (MSAs) with a population of more than 350,000 are required to report the AQI daily to the public. MSAs must report the AQI daily, which is defined as at least five days each week. There are six AQI categories and their names and colors are as follows:

AQI Range	Descriptor	Color
0 to 50	Good	Green
51 to 100	Moderate	Yellow
101 to 150	Unhealthy for Sensitive Groups	Orange
151 to 200	Unhealthy	Red
201 to 300	Very Unhealthy	Purple
301 to 500	Hazardous	Maroon

The pollutant specific sensitive groups are separated by 8-h daily maximum O₃ concentrations as indicated in Fig. 2-2 below.

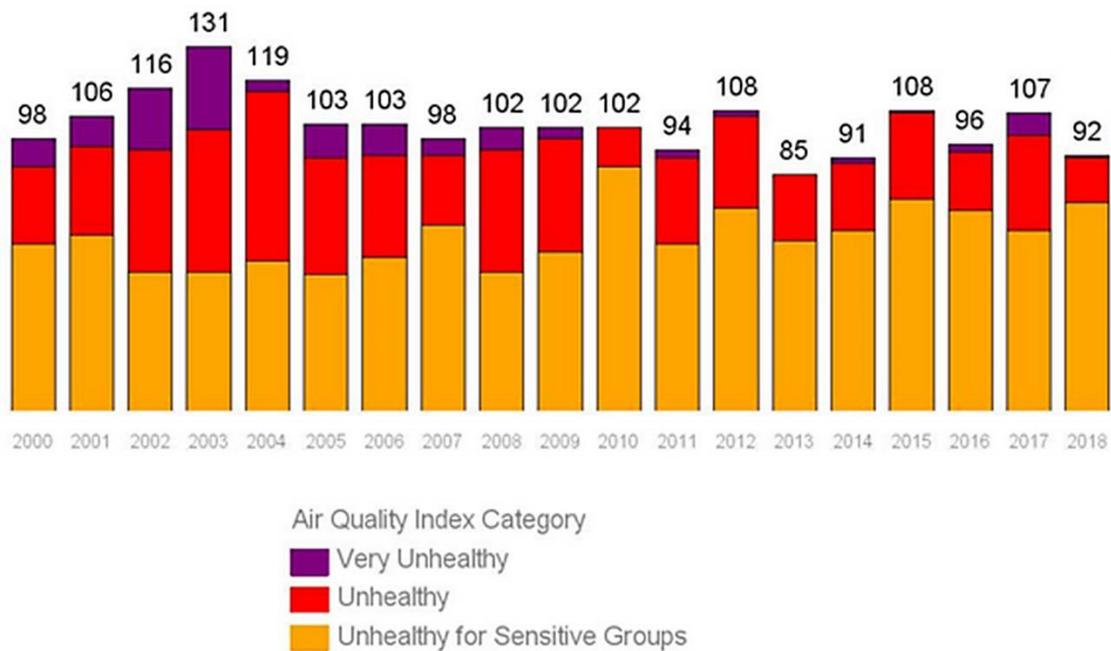
8-hour Ozone Concentration	Air Quality Index Levels
0 - 54 ppb	■ Good (Green)
55 - 70 ppb	■ Moderate (Yellow)
71 - 85 ppb	■ Unhealthy for Sensitive Groups (Orange)
86 - 105 ppb	■ Unhealthy (Red)
106 - 200 ppb	■ Very Unhealthy (Purple)
>200 ppb	■ Hazardous (Maroon)

Figure 2-2. Air quality index levels (AQI) related to 8-h concentrations.

An important aspect of the AQI index is that the higher the 8-h daily maximum O₃ concentration the higher the index. While this would appear to be an obvious statement, the ramification is that simply counting the number of exceedances of 8-h daily maximum concentrations greater than 70 ppb will not provide an accurate indication of the health risks associated with O₃ exposures during a specific period (e.g., March-October). ***Rather, it is the cumulative sum of the number of days above 70 ppb weighted by a factor that relates to each day's index range that is most important.*** For example, in Fig. 2-3 below for Los Angeles-Long Beach-Anaheim, CA (EPA, 2019c), when one compares the exposure for 2015 with the exposure for 2017, one might conclude based on the number of exceedance days (108 versus 107 above 70 ppb) that the annual O₃ exposures were similar. However, more days in 2017 occurred with orange and purple exceedances than in 2015. Based on the number of days that experienced orange and purple exceedances, the O₃ exposures (i.e., health risk) experienced by the public were higher in 2017 than 2015. As indicated, simply comparing the number of exceedance days among O₃ monitors is not an adequate way to quantify the exposure differences among monitors. In an effort to inform the public of potential health risks, the American Lung Association's annual State of the Air Report (ALA, 2022) (<http://www.lung.org/our-initiatives/healthy-air/sota/key-findings/>) utilizes weighting factors applied to each range of daily 8-h daily maximum O₃ concentrations associated with the Air Quality Index levels described above. The number of orange days (unhealthy for sensitive groups) experienced by each county receives a factor of 1; red days (unhealthy), a factor of 1.5; purple days (very unhealthy), a factor of 2; and maroon days (hazardous), a factor of 2.5. This weighting scheme provides a better way to inform the public of potential health risks than the simple determination used by various groups to count the number of exceedances above 70 ppb.

Los Angeles-Long Beach-Anaheim, CA

Number of Days Reaching Unhealthy for Sensitive Groups or Above on the Air Quality Index (for Ozone Only)



Data Source: Preliminary air quality data as reported to EPA's Air Quality System and AirNow.gov

Figure 2-3. Number of days reaching unhealthy levels for sensitive groups or above on the Air Quality Index for Los Angeles-Long Beach -Anaheim, CA. Source: EPA, 2019c: A Look Back: Ozone in 2018.

<https://epa.maps.arcgis.com/apps/Cascade/index.html?appid=9bec4031ba6f4887a9f332a8f058b198>

2.2 Vegetation

For vegetation, the EPA ISA (2013, 2020a) concluded that (1) O₃ effects in plants are cumulative; (2) higher O₃ concentrations appear to be more important than lower concentrations in eliciting a response; (3) plant sensitivity to O₃ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the O₃ hourly concentrations and preferentially weight the higher concentrations, improves the predictive power of exposure/response models for growth and yield, in comparison with using indices based on mean and other exposure indices. Results from a “natural experiment” site in the San Bernardino National Forest, where substantial reductions over the years in the higher hourly average O₃ concentrations in the Los Angeles area occurred, provide independent confirmation of the experimental studies described below for the greater importance of the higher hourly average O₃ concentrations in influencing vegetation effects. The 2013 ISA (EPA, 2013, page 9-106) noted that at the San Bernardino site, located near Los Angeles, reductions in ambient O₃ exposures between 1980 and 2000 were related to improvements in tree conditions. The frequency of midrange hourly average O₃ concentrations was little changed over this period. EPA suggested

that it was the reduction in the higher hourly average O₃ concentrations that was responsible for the improvement in tree health. EPA notes these results in the second draft PA (EPA, 2023, page 4-97). The San Bernardino study provides important empirical evidence of the importance of weighting the higher hourly average concentrations more than the mid-range and lower hourly average O₃ concentrations in eliciting an adverse response on vegetation. The pattern of occurrence of mid-range hourly concentrations (i.e., the number of hourly average concentrations between 50 and 89 ppb) suggested a lesser role for these concentration ranges compared to the higher values in either of the ground-level tree injury observations or the model simulation of growth over the 40-year period (EPA, 2013, page 9-106). The number of days experiencing hourly averaged O₃ concentrations greater than or equal to 95 ppb declined significantly from 163 days in 1978 to 103 days in 1997. Clearly the occurrence of the large number of hourly average concentrations greater than or equal to 95 ppb dominated the distribution of the O₃ concentrations associated with the total exposure. EPA (2013) noted that the San Bernardino study provided important information about the greater role for peak concentrations affecting plant growth inferred from air quality analyses for the southern California area. It should not be inferred that the San Bernardino study illustrates that only hourly average O₃ concentrations greater than or equal to a specific threshold are solely responsible for vegetation effects. Rather, the study provides empirical evidence of the importance of the higher hourly average O₃ concentrations in relation to the mid-range and lower concentrations. For more moderate O₃ exposure conditions (i.e., much below those occurring the San Bernardino area in the 1980s, as well as in the 2020s) occurring elsewhere, vegetation effects occur, but not as extreme as observed in the San Bernardino National Forest in the 1980s.

The key vegetation experimental studies that support the importance of the higher O₃ concentrations were performed 30 to 40 years ago (i.e., mostly in the 1980s and 1990s). Musselman et al. (1983) and Hogsett et al. (1985) performed research experiments that showed that the higher hourly average concentrations were more important than the mid- and low-level values in eliciting adverse vegetation effects. Following their results, a series of controlled experiments was undertaken worldwide for assessing the importance of the higher O₃ concentrations in eliciting vegetation response. These controlled fumigation experimental results (e.g., EPA, 1986, 1992, 1996a, 1996b, 2013; Musselman et al., 1986, 1994; Nussbaum et al., 1995; Yun and Lawrence, 1999; Lee and Hogsett, 1999; Oksanen and Holopainen, 2001; Köllner and Krause, 2003; Wang et al., 2008) provided additional evidence supporting the findings by Musselman et al. (1983) and Hogsett et al. (1985) that higher concentrations should receive greater weighting in comparison to the mid- and low-level values.

In the second draft PA (EPA, 2023, page 4-97), the authors discuss that in ambient regimes with similar values of cumulative, concentration-weighted metrics, differences in the magnitude and occurrence of peak concentrations appear to be influential regarding injury in tree seedlings. While this statement is accurate, the example provided is not necessarily the best one to highlight this point. The second draft PA (EPA, 2023) states

In an example highlighted in the 2006 AQCD and 2013 ISA, a study by Yun and Lawrence (sic) (1999) used exposure regimes constructed from 10 U.S. cities to demonstrate that in regimes with similar values of cumulative, concentration-weighted metrics, differences in the magnitude and occurrence of peak

concentrations were influential with regard to injury in tree seedlings (2006 AQCD, p. AX9-176; 2013 ISA, section 9.5.3.1; Yun and Lawrence (sic), 1999).

Most of the treatments, except for the nonexposed control, were simulated and did not necessarily mimic any of the hour-by-hour ambient O₃ exposures exhibited by the 10 sites mentioned in Yun and Lawrence (1999).

The authors conducted their O₃ exposures in four walk-in controlled-environment chambers for 5 days per week for 4 weeks. The treatments were as follows:

Treatment 1. Changing daily maximum O₃ concentration with 3 weeks between the two highest peaks.

Treatment 2. Changing daily maximum O₃ concentration with 1 week between the two highest peaks. To prevent unrealistically high O₃ injury, the highest peak was 180 ppb.

Treatment 3. Daily maximum O₃ concentration of 116 ppb.

Treatment 4. Nonexposed control.

Daily O₃ fumigations were from 08.00 to 15.00 hours with step changes at 30-minute intervals via a manually controlled output valve. The three O₃ treatments were designed to have the same SUM06 at the end of fumigation. Ozone data collected in the USA in 1988 and 1989 (183,960 hourly observations from Cumberland, North Carolina; Edmonson, Kentucky; Howland, Maine; Sheboygan, Wisconsin; Shelby, Alabama; Caddo Parish, Louisiana; Ithaca, New York), and in 1990 and 1991 from three sites in California (Crestline, Lassen, and Yosemite National Park) were examined by the authors. The authors noted that in general, once the O₃ concentration exceeded 60 ppb, elevated O₃ levels usually persisted for 3-4, but sometimes >10 days. The authors described the three most frequent O₃ structures as the following: (1) two different peaks above 60 ppb for 10-14 days with concentrations <60 ppb for 2-3 days between the two peaks, sometimes it is the first peak which is high and long, sometimes the second; (2) >60 ppb for 10-14 days with stable fluctuations <10 ppb; (3) >60 ppb for 10-14 days with fluctuations of 20-30 ppb and sometimes >40 ppb. Yun and Lawrence (1999) reported that their experimental treatments were based on these patterns. The investigators applied their three regimes to investigate: (a) the importance of a long (i.e., 3-week, Treatment 1) versus a short (i.e., 1 week, Treatment 2) duration of regimes with high peaks; (b) the effect of treatments with variable peak occurrence (Treatments 1 and 2) versus uniform peak occurrence (i.e., Treatment 3) during the exposure period. As mentioned above, all the treatments, except Treatment 4 (nonexposed control), were simulated and did not necessarily mimic any of the hour-by-hour ambient O₃ exposures exhibited by the 10 sites mentioned in their paper. The authors' focus was on maintaining the SUM60 cumulative exposures the same among the three treatments, while varying the different profiles containing the frequency and elevated O₃ hourly concentrations.

The citation of Yun and Lawrence (1999) is incorrect in the second draft of the PA (EPA, 2023, page 4-141). The correct citation is as follows:

Yun, S-C, Laurence, J.A. 1999. The response of sensitive and tolerant clones of *Populus tremuloides* to dynamic ozone exposure under controlled environmental conditions. *New Phytologist* 143, 305–313.

As indicated previously in this section, controlled human health clinical study results regarding the importance of the higher hourly average O₃ concentrations were reported in the early 1990s (Hazucha et al., 1992). Vegetation researchers reported in the 1960s that higher O₃ concentrations were an important factor for assessing vegetation O₃ effects. High O₃ concentrations were found in the 1960s to affect plant injury (e.g., visible plant injury) (Heck et al., 1966). Little research on the importance of higher O₃ concentrations in relation to the mid and lower levels affecting plant damage (e.g., growth) had been performed prior to the 1980s. In December 1981, in an informal discussion with the EPA at its research laboratory in Corvallis, Oregon, I discussed a possible hypothesis relating to the relative importance of the higher O₃ concentrations versus mid- and lower-level hourly average values for assessing plant damage resulting in economic impact. During the discussion, EPA inquired if I could design exposure regimes for the EPA that would test the hypothesis that the higher hourly average O₃ concentrations should be weighted more than the mid- and low-level values.

Soon after the December 1981 Corvallis discussion with the EPA researchers, Lefohn and Benedict (1982) published their paper that hypothesized that the higher hourly average concentrations should be provided greater weight than the mid- and low-level values when assessing crop growth reduction. Independent of the Lefohn and Benedict (1982) paper hypothesizing the importance of the higher hourly average O₃ concentrations, Musselman et al. (1983) published experimental evidence supporting the hypothesis. Hogsett et al. (1985), applying exposure regimes designed by me, provided additional experimental support of the Musselman et al. (1983) findings about the importance of the higher hourly average O₃ concentrations receiving greater weight than the mid and lower values in affecting vegetation.

Following the initial vegetation experiments by Musselman et al. (1983) and Hogsett et al. (1985), a series of controlled experiments was undertaken worldwide for assessing the importance of the higher O₃ concentrations in eliciting a vegetation response. These controlled fumigation experimental results as noted above provided additional evidence for emphasizing the importance of the higher concentrations in comparison to the mid- and low-level values (e.g., EPA, 1986, 1992, 1996a, 2013; Musselman et al., 1983, 1986, 1994; Hogsett et al., 1985; Nussbaum et al., 1995; Yun and Laurence, 1999; Lee and Hogsett, 1999; Oksanen and Holopainen, 2001; Köllner and Krause, 2003; Wang et al., 2008).

Based on the experimental evidence, these experiments helped form the basis for the focus on the higher hourly average O₃ concentrations. In other words, by reducing the highest hourly average O₃ concentrations in the distribution (not just the extreme peak values), the risk to vegetation would be reduced. Based on a thorough review of the literature, EPA (2013, 2020a) concluded that (1) O₃ effects in plants are cumulative; (2) higher O₃ concentrations appear to be more important than lower concentrations in eliciting a response; (3) plant sensitivity to O₃ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the O₃ hourly concentrations and preferentially weight the higher concentrations improves the predictive power of exposure/response models for growth and yield, over using

indices based on mean and other exposure indices. These conclusions have been reiterated in the current ISA (EPA, 2020a, page 8-180). The current ISA (EPA, 2020a, page 8-181) notes that no recent information available since the 2013 Ozone ISA (EPA, 2013) alters these basic conclusions.

As indicated in the ISA (EPA, 2020a, page 8-181), the authors discuss the flux uptake metric. The metric is based on determining the O₃ concentration from the atmosphere that enters the leaf and is discussed in the ISA as follows:

Another approach for improving risk assessment of vegetation response to ambient ozone is based on determining the ozone concentration from the atmosphere that enters the leaf (i.e., flux or deposition). Much work has been published in recent years, particularly in Europe, in using mathematically tractable flux models for ozone assessments at the regional, national, and European scale (Feng et al., 2017; Mills et al., 2011; Matyssek et al., 2008; Paoletti and Manning, 2007; Emberson et al., 2000b; Emberson et al., 2000a). While some efforts have been made in the U.S. to calculate ozone flux into leaves and canopies (Turnipseed et al., 2009; Uddling et al., 2009; Bergweiler et al., 2008; Hogg et al., 2007; Grulke et al., 2004; Grantz et al., 1997; Grantz et al., 1995), little information has been published relating these fluxes to effects on vegetation. Recently, Grantz et al. (2013) reported short-term ozone flux and related it to leaf injury in cotton in California. The authors reported that cotton leaves were most sensitive in the midafternoon, possibly due to changes in detoxification. They suggested with more research a sensitivity parameter may function well with the W126 metric. However, there remains much unknown about ozone stomatal uptake in vegetation at larger scales and how much uptake results in an injury or damage, which depends to some degree on the amount of internal detoxification occurring with each particular species. Those species having high amounts of detoxification potential may, in fact, show little relationship between ozone stomatal uptake and plant response (Musselman and Massman, 1999). The lack of data in the U.S. and the lack of understanding of detoxification processes have made this technique less viable for vulnerability and risk assessments in the U.S.

The interaction between O₃ and plant tissues is driven mainly by three distinct processes: changes in external O₃ concentration, O₃ uptake, and O₃ detoxification (Heath et al., 2009). As noted above in the ISA (EPA, 2020a), those species having high amounts of detoxification potential may, in fact, show little relationship between O₃ stomatal uptake and plant response. The diurnal pattern of detoxification does not necessarily match the diurnal patterns of external O₃ concentration and O₃ uptake (Heath et al., 2009; Wang et al., 2015; Dai et al., 2019; Wu et al., 2021). Lefohn et al. (2018) discussed the stomatal flux index as follows:

For assessing the potential for ozone to affect vegetation injury, growth and/or yield, exposure is defined as the integral of the instantaneous level over the period the vegetation is exposed to ozone (commonly expressed in unit of mol m⁻³ h or ppm-hrs) (Musselman et al., 2006). Examples of exposure indices are the W126

and AOT40 metrics (see Section 2.3.4). Although not necessarily considered exposure, seasonal average levels (e.g., 12-h daily average values averaged over a specified period) have also been referred to as exposure indices (EPA, 2013). In contrast, the ozone dose is determined by first calculating the stomatal flux, which is a temporally dynamic measure of the rate of entry of ozone into the leaf ($\text{nmol m}^{-2} \text{s}^{-1}$). Dose is the total amount of ozone that is absorbed into the leaf through the stomata, in units of nmol m^{-2} , over a period of time and is calculated by integrating over time the instantaneous stomatal flux (Fowler and Cape, 1982; Mills et al., 2011b). The flux is accumulated over a species-specific phenological time window and the vegetation-damaging ozone flux is expressed as the Phytotoxic Ozone Dose (POD_Y), where Y represents a detoxification threshold below which it is assumed that any ozone molecule absorbed by the leaf will be detoxified (Mills et al., 2011b).

Lefohn et al. (2018) noted that flux-based metrics involve accumulation above a fixed flux threshold which is included to represent the detoxification capacity of the plant that varies with vegetation type/species (Mills et al., 2011). While detoxification should ideally be represented as a dynamic variable rather than as a fixed threshold, modeling approaches are not yet able to take this dynamic variation into account for exposure-based (e.g., AOT40 or W126) or flux-based metrics. Results reported by Wang et al. (2015) for the diurnal changes of ascorbate, a major detoxification agent in the apoplast and leaf tissues of winter wheat, provide evidence for the dynamic nature of detoxification. Dai et al. (2019) observed apoplastic ascorbate (ASC_{apo}) as an important contributor to the detoxification of O_3 in plants. The diurnal variation of ASC_{apo} , with maximum values occurring in the late morning with lower values experienced in the afternoon, was observed. With the detoxification potential by ASC_{apo} being lower in the afternoon, the implication is that the period of greatest uptake (e.g., the late morning/early afternoon hours) of O_3 in vegetation may coincide with the period of greatest detoxification potential, while the period of less detoxification may occur in the late afternoon hours, when the highest hourly O_3 concentrations occur. Goumenaki et al. (2021) noted that ascorbic acid (AA) content and/or redox state was subject to day/night control. The investigators reported that plants exposed to equivalent O_3 fluxes administered during daytime versus nighttime exhibited a significant decline in biomass in both cases, and the losses were greater in plants subjected to equivalent O_3 flux at night. This observation was consistent with the nighttime depletion of cell wall-localized ascorbate. Wu et al. (2021) have suggested that O_3 detoxification should be a dynamic variable in flux-based O_3 metrics. The authors noted that acclimation to O_3 and O_3 detoxification of crops are likely greater during morning to noon hours and during flowering to grain-filling stages, when photosynthesis is maximal. Lloyd et al. (2020) indicate that changing detoxification by time of day is an important aspect of determining effective flux. Heath et al. (2009) hypothesized that the changing detoxification pattern during the time of day was a possible explanation for the higher O_3 concentrations (which occurred in the later part of the day) eliciting a greater effect than the mid- and low-level values.

The EPA made a conscious decision almost over 40 years ago, based on detailed analyses, to not use *average* O_3 concentration metrics as indicators of potential harm to vegetation. In 1986, the EPA addressed the issue of using seasonal average concentration metrics to protect vegetation. In the early 1980s, the EPA considered the seasonal 7-h daily average

concentration (referred to as the M7 (0900 h – 1559 h) metric) as a vegetation O₃ standard. In its Air Quality Criteria for Ozone and Other Photochemical Oxidants document (EPA, 1986), the Agency stated on pages 6-10 and 6-11:

A mean concentration (with various averaging times) is the most common statistic used. Because the mean is computed by summing the concentrations and dividing by time, it mathematically treats all concentrations as being equally effective in causing a plant response. The use of a mean concentration (with varying averaging times) to characterize long-term exposures minimizes the contributions of peak concentrations to the response by treating low-level, long-term exposures the same as high-concentration, short-term exposures. ***The use of a longer-term mean concentration ignores the importance of peak concentrations and is inconsistent with the literature*** (emphasis added). A number of studies have shown that concentration is more important than exposure duration in causing a response. For example, studies with beans and tobacco (Heck et al., 1966) showed that a dose over a short time period induced more injury than the same dose distributed over a longer time period. Studies with tobacco showed that the O₃ concentration was substantially more important than exposure duration in determining the extent of foliar injury (Tonneijck, 1984). In this study, tobacco was exposed to a range of O₃ concentrations (0.02 to 0.15 ppm) for 8 hr/day for 1 to 7 days. In beans, foliar injury developed when the internal O₃ flux exceeded 115 ~moles/m² within 1 hr (Bennett, 1979). However, a single 3-hr exposure at about half the O₃ concentration (0.27 compared to 0.49 ppm) required approximately 64 percent greater internal O₃ flux to induce the same amount of foliar injury as in the 1-hr exposure (Bennett, 1979). Amiro et al. (1984) showed that higher concentrations were more important than low concentrations in causing injury. Their study also suggested the existence of a biochemical injury threshold (i.e., the O₃ uptake rates that plants can experience without inducing visible foliar injury). The greater importance of concentrations compared to exposure duration has been reported by other authors also (e.g., Heck and Tingey, 1971; Henderson and Reinert, 1979; Reinert and Nelson, 1979). The total ozone dose (concentration multiplied by time) has been used to describe plant exposure; however, it suffers from the same problem as the mean. The total dose is simply the summation of the ppm-hr over the study period, which treats all concentrations as being equally effective. Several investigators have attempted to give greater importance to peak O₃ concentrations. Oshima et al. (1977a,b) and Lefohn and Benedict (1982), for example, have summed only the ppm-hr of exposure greater than some preselected value. Larsen et al. (1983) introduced the concept of "Impact" to describe the effects of O₃ and SO₂ on soybeans. The "Impact (I)" is calculated similarly to total dose, except that the concentration is raised to an exponent greater than one ($I = C^W \times T$); this method of calculation effectively gives greater weight to the higher concentrations. More recently, Larsen and Heck (1984) have suggested the term "effective mean" as an approach for describing the greater importance of higher concentrations. The "effective mean" is defined as the average hourly impact raised to an exponent and divided by the duration. Several lines of evidence suggest that higher concentrations have

a greater influence in determining the impact of O₃ on vegetation. Studies have shown that plants can tolerate some combinations of exposure duration and concentration without exhibiting foliar injury or effects on growth or yield, illustrating that not all concentrations are equally effective in causing a response. From the toxicological perspective, it is the peaks or concentrations above some level that are most likely to have an impact. Effects occur on vegetation when the amount of pollutant that the plant has absorbed exceeds the ability of the organism to repair or compensate for the impact.

On page 6-12, the EPA (1986) continues

Not only are concentration and time important but the dynamics of the O₃ exposure are also important; that is, whether the exposure is at a constant or variable concentration. Musselman et al. (1983) recently showed that fixed concentrations of O₃ cause the same kind of responses as variable concentrations at the equivalent dose. Fixed concentrations, however, had less effect on plant growth responses than variable concentrations at similar doses. Exposures of radishes to ambient O₃ in open-top exposure chambers showed that significant yield reductions occurred when the maximum O₃ concentration exceeded 0.06 ppm on at least 10 percent of the days when the crop was growing (Ashmore, 1984). Initial studies by Hogsett et al. (1985) have compared the response of alfalfa to daily peak and episodic O₃ exposure profiles which had the equivalent total O₃ dose over the growing season. Alfalfa yield was reduced to a greater extent in the episodic than the daily peak exposure. This study also illustrates the problem with the 7-hr seasonal mean concentration, which is that the peak concentrations are not properly considered. **The plants that displayed the greater growth reduction (in the episodic exposure) were exposed to a significantly lower 7-hr seasonal mean concentration** (emphasis added). Studies with SO₂ also showed that plants exposed to variable concentrations exhibited a greater plant response than those exposed to a constant concentration (McLaughlin et al., 1979; Male et al., 1983). These results suggest that the mechanisms causing the response are the same, but that exposures to fixed concentrations underestimate the magnitude of plant growth responses that can occur with episodic exposures.

Since 1986, the EPA has reiterated its commitment to focusing on weighting the higher hourly average concentrations more than the mid- and low-level hourly average concentrations to protect vegetation from both injury and damage (EPA, 1996; 2006; 2013; Federal Register, 2015; Federal Register, 2020b). As mentioned above, EPA (1986) noted the greater importance of concentrations compared to exposure duration. The total O₃ exposure (i.e., the sum of all hourly average concentrations over a period), referred to as SUM00, in the literature, has been used to describe plant exposure. However, this exposure metric suffers from the same problem as the mean. The total exposure (SUM00) is simply the summation of the concentration multiplied by time over the study period, which treats all concentrations as being equally effective. Even though the seasonal average (e.g., M7 and M12) exposure metrics are currently applied by some researchers in experimental studies for assessing vegetation risk, as EPA (1986, 1996a, 2006,

2013) noted in its literature reviews, both the seasonal average and SUM00 O₃ exposure metrics are inappropriate for assessing vegetation effects based on biological experiments using different exposure regimes, as well as empirical “uncontrolled” experiments (e.g., the San Bernardino National Forest study).

As noted earlier, it is not just the reduction of the "peak" concentrations that is important, but also the reduction of those hourly average concentrations in the upper part of the distribution that is important to reduce vegetation effects. To accomplish this goal, the Agency moved from its initial consideration of seasonal M7 (daily 7-h average concentration averaged over a growth season) or M12 (daily 12-h average concentration averaged over a growth season) exposure metrics to the use of cumulative exposure metrics (i.e., SUM06 and W126). The SUM06 exposure metric is the sum of all hourly average concentrations greater than or equal to 60 ppb. As mentioned above, the EPA has discussed the use of flux-based indices in its standard-setting process but believes that further research concerning dynamic detoxification and other considerations are required before flux indices might be considered as a practical use in the standard-setting process. The W126 O₃ exposure index, where the sigmoidal weighting is described in Lefohn and Runeckles (1987) and the W126 metric is described in Lefohn et al. (1988), is a weighted cumulative exposure index that provides greater weight to the higher hourly average O₃ concentrations rather than the mid- and lower-level values. Fig. 2-4 below illustrates the weighting scheme. In addition, the W126 index does not impose an artificial cutoff (i.e., threshold) and is not an "average" of several values collected over the course of a short- or long-term period. The differential weighting of the hourly average O₃ concentration as demonstrated in the W126 weighting approach, as noted by the EPA (2013), is supported by research results performed under *controlled* conditions, as well as under *natural* exposure conditions, such as observed in the San Bernardino National Forest in the Los Angeles area.

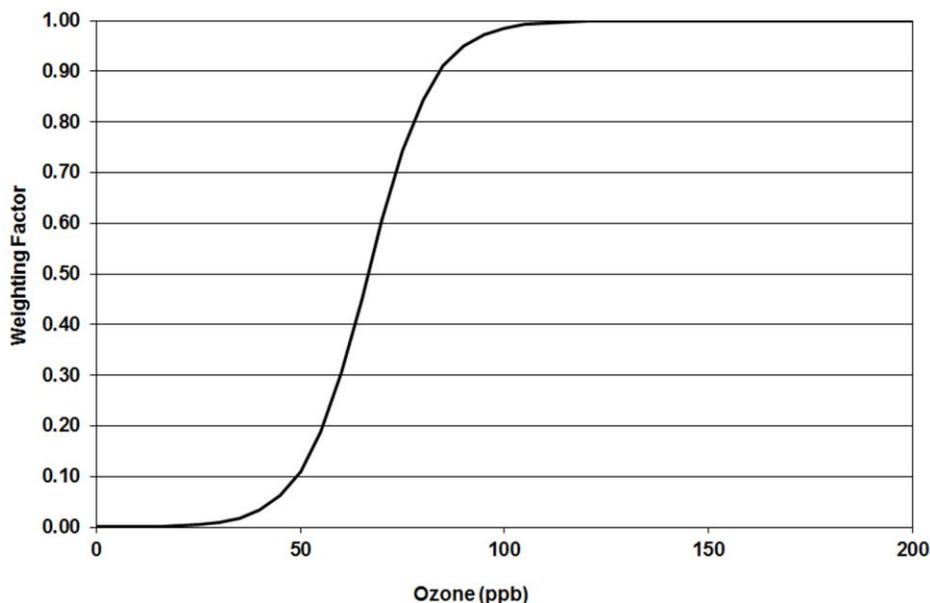


Figure 2-4. The weighting applied to hourly average O₃ values for the calculation of the W126 exposure index (see Lefohn and Runeckles, 1987 and Lefohn et al., 1988 for further details).

The 2013 ISA (EPA, 2013) noted that at the San Bernardino site, located near Los Angeles, reductions in ambient O₃ exposures between 1980 and 2000 were related to improvements in tree conditions. The frequency of midrange hourly average O₃ concentrations changed little over this period. EPA (2013) suggested it was the reduction in the higher hourly average O₃ concentrations that was responsible for the improvement in tree health.

As indicated above, based on a thorough review of the vegetation literature, (1) O₃ effects in plants are cumulative; (2) higher O₃ concentrations are more important than lower concentrations in eliciting a response; (3) plant sensitivity to O₃ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the O₃ hourly concentrations and preferentially weight the higher concentrations improves the predictive power of exposure/response models for growth and yield, over using indices based on mean and other exposure indices. Based on the peer-reviewed literature and its own research studies, the EPA (2013) identified the W126 cumulative exposure metric as the most appropriate to use to evaluate both the adequacy of the current secondary standard and the appropriateness of any potential revisions (Federal Register, 2015 – page 65373). In its O₃ current reconsideration of the O₃ NAAQS (EPA, 2023), the EPA again focuses on the W126 cumulative exposure index as the metric to protect vegetation. However, in addition to the W126 metric, the Agency has described the need for an additional index that focused on the frequency of hourly O₃ average concentrations ≥ 100 ppb. However, as discussed in Section 4, one does not require the occurrence of hourly average O₃ concentrations ≥ 100 ppb for vegetation visible foliar injury or damage to occur. The ISA (EPA, 2020a, page 8-181) noted that McGrath et al. (2015) reported for maize and soybean yield that the SUM06, AOTx, and W126 metrics performed well and the W126 was the preferred metric because it was potentially the most sensitive index. Mills et al. (2018) noted that their results illustrated that the largest range of ambient exposure values worldwide was associated with the W126 metric, where the stronger weighting for the highest hourly average O₃ concentrations resulted in areas with the highest W126 values standing out from other exposure metrics (i.e., AOT40 and M12), which exhibited lower values. A similar, but less pronounced worldwide exposure pattern was shown for the AOT40 metric, with the M12 metric showing proportionately less spatial variation across the worldwide sites, especially in the Northern Hemisphere.

2.3 The Importance of the Higher Concentrations for Assessing Human Health and Vegetation Effects and Haber's Rule

As noted in this section, both vegetation effects research and controlled laboratory studies of human volunteers indicate that higher O₃ hourly average concentrations elicit a greater effect on hour-by-hour physiologic response than lower hourly average values. The weighting of the higher values compared to the mid and lower hourly average O₃ concentrations results in a nonlinear response for both human health and vegetation (Hazucha and Lefohn, 2007; Lefohn, Hazucha, Shadwick, and Adams, 2010; Heath et al., 2009). The nonlinear response noted above for the human health clinical studies and the vegetation experiments has an important impact on the validity of Haber's rule (also referred to as Haber's law). Haber's rule states that, for a given poisonous gas, $C \times t = k$, where C is the concentration of the gas (mass per unit volume), t is the

amount of time necessary to produce a given toxic effect, and k is a constant, depending on both the gas and the effect. Haber's law or rule, as commonly understood in inhalation toxicology, states: $C \times T = \text{constant}$, meaning that identical products of concentration of an agent in air (C) and duration of exposure (T), the CT product, will yield an identical biological response. The formula was originally developed by the German physical chemist Fritz Haber (1868–1934) to characterize the acute toxicity of chemicals used in gas warfare. For example, the rule states that doubling concentration will halve the time. Haber's rule is an approximation and Haber himself acknowledged that it was not always applicable (https://en.wikipedia.org/wiki/Haber%27s_rule). The greater importance of concentration compared to exposure duration for O_3 results in the failure of Haber's rule. Specifically, when concentration is more important than the time required to elicit an adverse effect, Haber's rule will not be applicable when attempting to determine a cumulative exposure. Miller et al. (2000) discuss the fact that many toxicologists have used Haber's rule to analyze experimental data whether their chemicals, biological endpoints, and exposure scenarios were suitable candidates for the rule. As indicated by the literature, as well as the EPA reviews since 1986, Haber's rule does not appear to be applicable for O_3 .

2.4 The Importance of the Selection of Biologically Relevant Exposure Metrics for Assessing Human Health and Vegetation Effects

Both the vegetation and the clinical health studies show that because the higher hourly average O_3 concentrations have a greater effect than the mid- and lower-level values, **the use of long-term average concentrations, which combine all hourly values into one number, is an inappropriate index** to use because the quantification of the highest hourly average concentrations, which are more biologically important than the lower and mid values, is obscured. If the long-term average is used, then a comparison among different O_3 monitoring sites results in many of the sites having similar long-term average concentrations which differ in the magnitude and number of the biologically important elevated O_3 hourly average concentrations. The long-term average is not correlated with the number and magnitude of the biologically important elevated hourly average concentrations. The relationship of the long-term average to the occurrence of the higher concentrations will be discussed further in Section 3.

Following the setting of the 2015 O_3 standard (Federal Register, 2015) and the 2020 O_3 NAAQS review (Federal Register, 2020b), the metrics used in the United States to assess the risk of O_3 to human health and vegetation continue to be the 8-h daily maximum concentration (human health) and the W126 cumulative exposure index (vegetation). Both metrics as discussed above are biologically relevant. However, other exposure metrics are used for assessments by researchers (e.g., Lefohn et al., 2018; Wang et al., 2019; Lefohn et al., 2017). Using the same hourly data, it is possible to reach entirely different scientific conclusions for assessing trends and O_3 impacts utilizing different exposure metrics (Lefohn et al., 2017, 2018). **Although specific exposure metrics may appear to work in a selected effects model, what is most important is that the exposure metrics be biologically relevant and scientifically defensible.** In many cases, it may not be possible to identify a biologically relevant exposure metric based on statistical comparisons. Prior to selecting and running a model for assessing human health or vegetation effects/risks, it is important to identify an exposure metric to be used in the model that is justified based on biological principles (Hazucha and Lefohn, 2007; Lefohn et al., 1988).

The impacts of surface O₃ on human health and vegetation have prompted precursor emission reductions in the United States. As emissions change, the distribution of hourly O₃ concentrations also changes, as do the values of individual exposure metrics (Lefohn et al., 2017). The distribution changes can result in exposure metric trend patterns changing in a similar direction as trends in emissions (e.g., metrics increase as emissions increase) or, in some cases, in *opposite* directions. For example, Lefohn et al. (2017) reported, using the current form of the 8-h standard for the 196 U.S. sites studied in their analysis, 162 sites showed negative trends, 32 showed no trends, and 2 had insufficient data. In comparison, using the 6-month (April-September) 12-h daylight average concentration, 92 sites showed negative trends, 85 sites showed no trend, 19 sites showed increasing trends, and 0 sites showed insufficient data. Thus, while 162 sites exhibited a negative trend in O₃ using the 4th highest annual 8-h average exposure metric, using the 6-month 12-h daylight average exposure metric based on data from the same sites showed only 92 instances of negative trends and increasing trends for 19 sites compared to 0 sites for the 8-h form of the standard. The second draft PA (EPA, 2023, page 2-20) notes that trends analyses show that metrics impacted by averaging longer time periods of hourly O₃ measurements, such as the 6-month (April-September) average of daytime (8am-7pm) O₃ concentrations, show more variation than metrics that focus on the higher concentrations. Using the same data from U.S. sites, Lefohn et al. (2017) note that for reduction emissions the trend patterns of the (1) 4th highest annual 8-h average exposure metric and (2) 6-month 12-h daylight average exposure index were extremely different and provided an excellent example of why it is so important to select appropriate biologically relevant exposure metrics for assessing human health and/or vegetation effects.

In addition to the Lefohn et al. (2017) analysis, Lefohn et al. (2018) compared several exposure metrics that focused on the higher hourly average O₃ concentrations with two metrics that are based on all hourly average concentrations (i.e., mean and median). In Table 5 of their analyses, Lefohn et al. (2018) compared the trends using different metrics that were in the same direction (i.e., decreasing, increasing, or no significant change) compared to other metrics. Trends in the human health metric impacted by the high end of the distribution bear the least resemblance to trends in the mean and median values with generally less than 50% of sites having trends in the same direction. In many cases the trends were in the opposite direction (i.e., the metrics associated with the highest concentrations exhibited decreases over time, while the mean and median metrics exhibited increases). Table 2-2 below provides a summary of the relationship between (1) two of the exposure metrics focused on the highest concentrations (i.e., the annual 4th highest daily maximum 8-h average concentration (4th dma8epa) and the number of daily maximum 8-h averages greater than 70 ppb (nvgt070 summer) for the months of April-September) and the (2) median and mean values. As indicated above, overall, trends in the four mean/median metrics (i.e., median annual, mean annual, median summer, and mean summer) were not representative of the trends behavior of those metrics associated with the higher hourly average O₃ concentrations.

Table 2-2. Comparison of trends using different metrics that were in the same direction (i.e., decreasing, increasing, or no significant change) compared to other metrics. Source: Adopted from Lefohn et al. (2018).

	median annual	mean annual	median summer	mean summer
4th dma8epa	33%	39%	43%	50%
nvgt070 summer	37%	44%	43%	53%

Lefohn et al. (2017) cautioned that trends in mean or median concentrations did not appear to be well associated with those exposure metrics that are most optimum indicators of overall changes in anthropogenic emissions, biological effects, or climate-driven meteorology. Similar to the findings of other studies, Lu et al. (2018) reported that exposure indices, such as the median and the M12 average metrics (average of hourly O₃ concentrations for the 12-h period from 08:00 to 19:59 local time April–September), which focus on the midrange of the O₃ hourly average concentration distribution, did not appear to adequately describe the magnitude and frequency of high O₃ events. The authors reported that the median and seasonal M12 metrics, instead of experiencing much greater exposures in China than in Japan, South Korea, Europe, and the United States, were similar in values for these other countries, whereas much higher hourly averaged O₃ concentrations were experienced in China compared to the other four countries.

Both the human health clinical results and the vegetation experiments form the basis for the implementation of O₃ control strategies in the United States. Simply stated, by reducing the higher part of the distribution (not just the peak values), the risk to human health and vegetation will be reduced. As the number of elevated O₃ concentrations is reduced, the risk to human health and vegetation is reduced. In 2015, it was the opinion of the EPA (Federal Register, 2015 – page 65358) that both acute and chronic effects would be reduced in implementing the new O₃ standards by reducing the higher hourly average O₃ concentrations. As emission reductions occur, the higher part of the distribution of hourly average concentrations moves downwards toward the middle of the hourly average values. As a result of reducing NO_x emissions, a reduction in the NO titration of O₃ occurs with the result that lower hourly average O₃ concentrations shift upwards. As mentioned in the next section, as emissions are reduced, annual averages or seasonal averages tend to be related to the behavior of the upward shifts in the lower hourly average O₃ concentrations. Thus, for the purpose of assessing human health or vegetation chronic effects, one might wish to select exposure metrics based on the repeated occurrences of the higher hourly average concentrations over time. As stated previously, the EPA believed in 2015 that adequate protection for both acute and chronic exposures can be attained by focusing on the reduction of the repeated occurrences of exposures of concern (i.e., the higher hourly average concentrations). This is an especially important strategy to implement. For without the focus on the highest hourly average O₃ concentrations for reducing human health and vegetation effects, one might believe that the increasing lower hourly average concentrations that occur

during emission reductions will have a detrimental effect on human health and vegetation. However, as discussed in this section, based on the scientific literature about the importance of the higher hourly average concentrations, the shift from the lower concentrations toward the mid-level values may not necessarily be detrimental to human health and vegetation.

3. Emission Reductions, Changing Hourly Average Distribution Patterns, Background Ozone, and the Second Fundamental Principle

As indicated in the previous section (Section 2), the first fundamental principle indicates that for human health and vegetation effects, the higher hourly average O₃ concentrations should be provided greater attention than the mid- and lower values within the distribution of hourly average concentrations. The second fundamental principle addresses what happens within the distribution of hourly average concentrations when emission reductions are applied to reduce the higher hourly average O₃ concentrations to protect human health and welfare. Section 3 discusses the second fundamental principle which is described as

Daily Maximum Hourly Averaged O₃ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide.

Based on research studies and “natural experiments,” the first fundamental principle discussed in Section 2 indicates that the higher hourly average O₃ concentrations should be weighted more than middle and lower values when assessing human health and environmental effects. Emission control strategies in the U.S. focus on the reduction of the higher hourly average O₃ concentrations. In 2015, the EPA Administrator believed by reducing *the higher part of the distribution of hourly average concentrations* (not just the peak hourly values ≥ 100 ppb) that the risk to human health and vegetation would be reduced. The EPA in its 2015 decision (Federal Register, 2015 – pages 65358 – 65359) anticipated that a revised standard with a level of 70 ppb would also reduce the occurrence of exposures to O₃ concentrations at least somewhat below 60 ppb based on its modeling results in the 2014 Health Risk and Exposure Assessment document (EPA, 2014b, Figs. 4-9 and 4-10). The modeling results illustrated that as emissions were reduced, the O₃ concentrations would move downwards toward the middle hourly average O₃ values, while the lower concentrations would move upwards toward the middle values.

The observation of the lower hourly average O₃ concentrations shifting upward as emissions are reduced defines the basis for the second Fundamental Principle. If one were to assume that to control for chronic health effects that the annual or seasonal averages of the hourly O₃ concentrations must be reduced, then chemical models (EPA, 2014b, Figs. 4-9 and 4-10), as well as empirical air quality data, indicate that annual or seasonal average metrics will not perform as some researchers anticipate. As more and more anthropogenic emissions are reduced, there is a range of hourly average O₃ concentrations (at a specific monitoring site) influenced by these emissions that begins to appear resistant to further change. As will be discussed below, the process of identifying this range of O₃ concentrations begins with the movement of both the high and lower hourly average concentrations towards the mid-level values as emissions are reduced (i.e., the compression effect). As a result of the shifting patterns of the hourly average O₃

concentrations, the 8-h daily maximum (MDA8) concentrations at some sites are reduced downward toward the mid-level concentrations and the lowest MDA8 values *increase*. These patterns have been reported in the literature as well as EPA reviews (e.g., Lefohn et al., 1998; Lefohn et al., 2010b; EPA, 2014b, 2020b; Simon et al., 2015; Lefohn et al., 2017, 2018). The shift is associated with less titration of O₃ by NO of the lower hourly average concentrations as reduction in NO_x emissions occur.

To develop the discussion for the second Fundamental Principle, we first explore the changing patterns in the distribution of hourly average O₃ concentrations as emissions are reduced. We explore the changes in the frequency of both high and low levels of O₃ hourly average concentrations. Following the description of the changes in the distribution patterns, we then explore when during the year the highest hourly average O₃ concentrations occur at sites where emissions were reduced. Both modeling and empirical data are presented for comparison. Finally, we explore what the results from models, as well as empirical data, tell us about background O₃ levels. Although the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) in August 2019 ruled that background O₃ should not directly influence the setting of the level of the O₃ NAAQS, as described in Section 3.2, background O₃ levels influence risk assessments associated with margin of safety considerations. Integrating all the information described in this section provides us with the modeling and empirical support for explaining the science behind the second Fundamental Principle.

The importance of the second Fundamental Principle is that the continuation of emissions reductions will fail in achieving reductions of the lowest hourly average O₃ concentrations. Emission reductions will achieve the shifting of the higher hourly average O₃ concentrations toward the mid-level values. However, rather than emission reductions causing the mid-level hourly values to shift downward toward the very lowest values, the lower hourly average concentrations will shift upwards toward the mid-level values, with the result that a Gaussian-like (i.e., bell shaped) distribution of hourly average concentrations at some sites may occur depending upon the amount of emission reductions and the influence of the remaining anthropogenic contributions to ambient O₃ levels. This phenomenon is discussed in later subsections of Section 3. The distribution of background O₃ hourly average concentrations will determine for each site the range of hourly average concentration values in the limit as emissions are reduced. The shape of the distribution of O₃ hourly average concentrations may appear to be Gaussian-like with only the amplitude varying at each site. Thus, daily maximum hourly averaged O₃ concentrations will remain well above 0 ppb even if all anthropogenic emissions were eliminated everywhere. As described in Section 3.2.12, the severe reduction of emissions during the COVID-19 lockdown in many countries during the spring (Northern Hemisphere) and fall (Southern Hemisphere) of 2020, resulted in a “natural experiment” that provided confirmation on the behavior of hourly average O₃ concentrations at both the high end and low end of the distribution. This “natural experiment,” as well as models and empirical data collected over the years, provide us with important insight as emissions are reduced about the behavior of changes in the distribution of O₃ concentrations.

3.1 Patterns of Shifting of Hourly Average Concentrations as Emissions are Reduced

3.1.1 The Lower Ozone Concentrations Shift Upward as Emissions are Reduced

In the 2014 EPA Policy Assessment document (EPA, 2014a), the EPA noted in its modeling effort that as NO_x was reduced, the high end of the distribution shifted downward and the low-end of the distribution shifted upward. There was a compression of the distribution of concentrations. Figures 4-9 and 4-10 (pages 4-24 and 4-25 in the 2014 PA) presented in EPA (2014b) are reproduced here as Figs. 3-1 and 3-2. For the 12 urban-influenced sites described in the modeling results for the period April-October, the general pattern from the modeling effort is that as emissions reductions occur to attain 4th highest 8-h daily maximum (MDA8) values for alternative scenarios of 75, 70, 65, and 60 ppb, the individual daily MDA8 values at the high end of the distribution are reduced toward the center of the distribution. However, the lowest daily MDA8 values increase. The model predictions have been observed as noted earlier using actual observations resulting from emission reductions (Lefohn et al., 1998; Lefohn et al., 2010b; Simon et al., 2015; Lefohn et al., 2017, 2018). The shifting of the lower concentrations toward the mid-level values is associated with less NO titration of O₃ of the lower hourly average concentrations as reduction in NO_x emissions occurs (Lefohn et al., 1998; Lefohn et al., 2010b; EPA, 2014b, 2020b; Simon et al., 2015; Lefohn et al., 2017, 2018).

To attain the O₃ standards, a strategy of reducing NO_x emissions has been implemented to reduce the higher hourly average O₃ concentrations. As a result of the reduction of NO titration of O₃ by NO_x reductions, the lower hourly average concentrations at many locations are shifted upward as the more biologically important higher hourly average O₃ concentrations are shifted downward (Simon et al., 2015; Lefohn et al., 2017; Lefohn et al., 2018). Simon et al. (2015) discussed the effects of reducing O₃ precursors in the United States on O₃ concentrations. Using daily 8-h average concentrations, the authors reported that decreasing O₃ trends generally occurred in the summer, in less urbanized areas, and at the upper end of the O₃ distribution (i.e., the higher 8-h concentrations). Conversely, increasing O₃ trends generally occurred in the winter, in more urbanized areas, and at the lower end of the O₃ distribution. As noted in the second draft PA (EPA, 2023, page 2-20)

Simon et al. (2015) found that, similar to results presented in this section for DVs and annual 4th high MDA8 concentrations, the 95th percentile of summertime MDA8 concentrations decreased significantly at most sites across the U.S. between 1998 and 2013. In contrast, trends over that time period for the 5th percentile, median and mean of MDA8 varied with location and time of year. Similarly, Lefohn et al. (2017) reported that between 1980 and 2014 there was a compression of the distribution of measured hourly O₃ values with extremely high and extremely low concentrations becoming less common. As a result, O₃ metrics impacted by high hourly O₃ concentrations, such as the annual 4th highest MDA8 value, decreased at most US sites across this period. Concurrently, metrics that are impacted by averaging longer time periods of hourly O₃ measurements, such as the 6-month (April-September) average of daytime (8am-7pm) O₃ concentrations, were more varied with only about half of the sites exhibiting

decreases in this metric and most other sites exhibiting no trend (Lefohn et al., 2017).

By focusing on a specific level of the 4th highest MDA8 value that protects human health, the EPA's emissions reductions strategy forces the concentrations of concern (i.e., the highest values) downward toward the middle values. The frequency of the middle concentration values is increasing due to the downward shift from the higher values as indicated above, as well as the upward shift of the lower MDA8 concentrations toward the middle. In other words, the high end is coming down and the low end is coming up. Both meet in the middle of the distribution as described by Lefohn et al. (1998) and Simon et al. (2015).

On page 3C-101 (3C.7.2 Distribution of Hourly O₃ Concentrations) of the 2020 PA (EPA, 2020b), the Agency updated its 2014 modeling analyses. It appears that in that subsection of the PA, the figures illustrated in the 2020 PA (EPA, 2020b) and the figures shown in the same section in the second draft PA (EPA, 2023, page 3C-91) are identical. EPA (2023, page 3C-81) notes that

The precursor reductions used to estimate spatial and temporal patterns of O₃ concentrations for the three air quality scenarios were NO_x-only reductions. We focused on NO_x-only reductions in light of several key findings from analyses for the 2014 HREA that explored the use of both NO_x and VOC reductions versus NO_x-only scenarios (2014 HREA, Appendix 4D). There were several key findings from that comparison. First, in most of the urban study areas, the NO_x/VOC scenario did not affect O₃ response at the monitor having the highest design value in such a way to reduce the total required emissions cuts. Further, evidence in the literature has shown that locations in the U.S. have gotten more NO_x-limited since 2007 (the year modeled in the 2014 HREA) (Jin et al., 2017, Laughner and Cohen, 2019) and thus VOC reductions would be expected to have less impact on resulting O₃ concentrations in our scenarios for the 2016 modeling used here than they had in the previous analysis. Finally, the two areas (Denver and Chicago) in which VOC emissions had the most impact in the 2014 HREA were not included in the current analysis. For these reasons, NO_x-only reductions were the most appropriate scenarios for this analysis.

Koplitz et al. (2021) noted that anthropogenic NO_x and VOC emissions in the United States have declined substantially over the last decade, altering the NO_x-VOC chemistry and O₃ production characteristics of many areas. The authors used multiple air quality analysis tools to assess how these large reductions in NO_x and VOC affected O₃ production regimes across the United States between 2007 and 2016. Both the ambient and modeling data showed that more locations across the United States have shifted toward NO_x-limited regimes between 2007 and 2016. The model-based HDDM sensitivity analysis shows only a few locations remaining NO_x-saturated on high-O₃ days in 2016, including portions of New York City, Chicago, Minneapolis, San Francisco, and Los Angeles. The work by Koplitz et al. (2021) provides insights into the current state of O₃ production chemistry in large population centers across the United States, as well as how O₃ chemistry in these areas may evolve in the future.

The second draft PA (EPA, 2023) conclusions are similar to the conclusions reached in the 2014 PA (EPA, 2014a) document. Figs. 3-3 to 3-10 (reproduced from Fig. 3C-67, page 3C-93 through Fig. 3C-74, page 3C-100) display diurnal boxplots of hourly O₃ concentrations for 2015-2017 at monitor locations in each urban area. For each hour of the day, the rectangular box represents the 25th and 75th percentiles of the distribution, with a solid line representing the median of the distribution through the center. Each box has “whiskers” which extend up to 1.5 times the interquartile range (i.e., the 75th percentile minus the 25th percentile) from the box, and dots which represent outlier values. Black boxplots represent observed hourly O₃ concentrations, while blue boxplots represent hourly O₃ concentrations adjusted to meet the current standard of 70 ppb. Red boxplots represent hourly O₃ concentrations adjusted for the 75 ppb scenario, and green boxplots represent hourly O₃ concentrations adjusted for the 65 ppb scenario.

Eight cities were highlighted in the Agency’s modeling analyses (Atlanta, Boston, Dallas, Detroit, Philadelphia, Phoenix, Sacramento, and St. Louis). Ambient hourly O₃ monitoring data for years 2015 through 2017 in each of the eight urban study areas were adjusted using a model-based adjustment approach to create three different air quality scenarios. These scenarios included conditions that just meet the current O₃ standard (design value of 70 ppb), as well as conditions that just meet two alternative standards (design values of 75 ppb and 65 ppb). The figures below (Figs. 3-3 to 3-10) reproduced from the second draft PA (EPA, 2023) illustrate the shifting of the hourly average concentrations that result in the compression of the distribution as emissions are reduced to attain the three scenarios listed above. The EPA described the compression of the distribution of hourly average concentrations (high-end shifting downward and the low-end shifting toward the middle) in the modeling results in the second draft PA (EPA, 2023, page 3C-91) as follows:

The hourly plots show similar patterns in most of the urban study areas. O₃ concentrations during daytime hours decrease from observed values (black) to values adjusted to meet the current standard of 70 ppb (blue) and decrease further under the alternative scenario of 65 ppb (green). These daytime decreases are mainly seen on high O₃ days represented by outlier dots extending above the box and whiskers. Some study areas had observed 2015-2017 design values already meeting the alternative scenario of 75 ppb, therefore some plots show increases in O₃ concentrations while other study areas show decreases in O₃ concentrations for the 75 ppb scenario.

In some urban study areas O₃ concentrations on the mid-range days, represented by the 25th – 75th percentile boxes, remained fairly constant (e.g. Boston) while in other urban study areas O₃ on mid-range days decreased (e.g. Atlanta). Although daytime O₃ decreased, concentrations during morning rush-hour period generally increase. These increases are associated with VOC-limited and NO_x titration conditions near NO_x sources during rush-hour periods. Reducing NO_x under these conditions results in less O₃ titration and thus increases O₃ concentrations. Nighttime increases in O₃ as a results of NO_x reductions are often seen to a lesser extent than morning rush-hour period increases. Collectively these

features generally lead to a flattening of the diurnal O₃ pattern with smaller differences between daytime and nighttime concentrations as NO_x emissions are reduced. Urban study areas that required more substantial NO_x reductions for the 65 ppb scenario generally had more pronounced patterns of decreases in daytime O₃ and increases in nighttime O₃ leading to a flatter diurnal O₃ pattern (e.g., Sacramento in Fig. 3C-73).

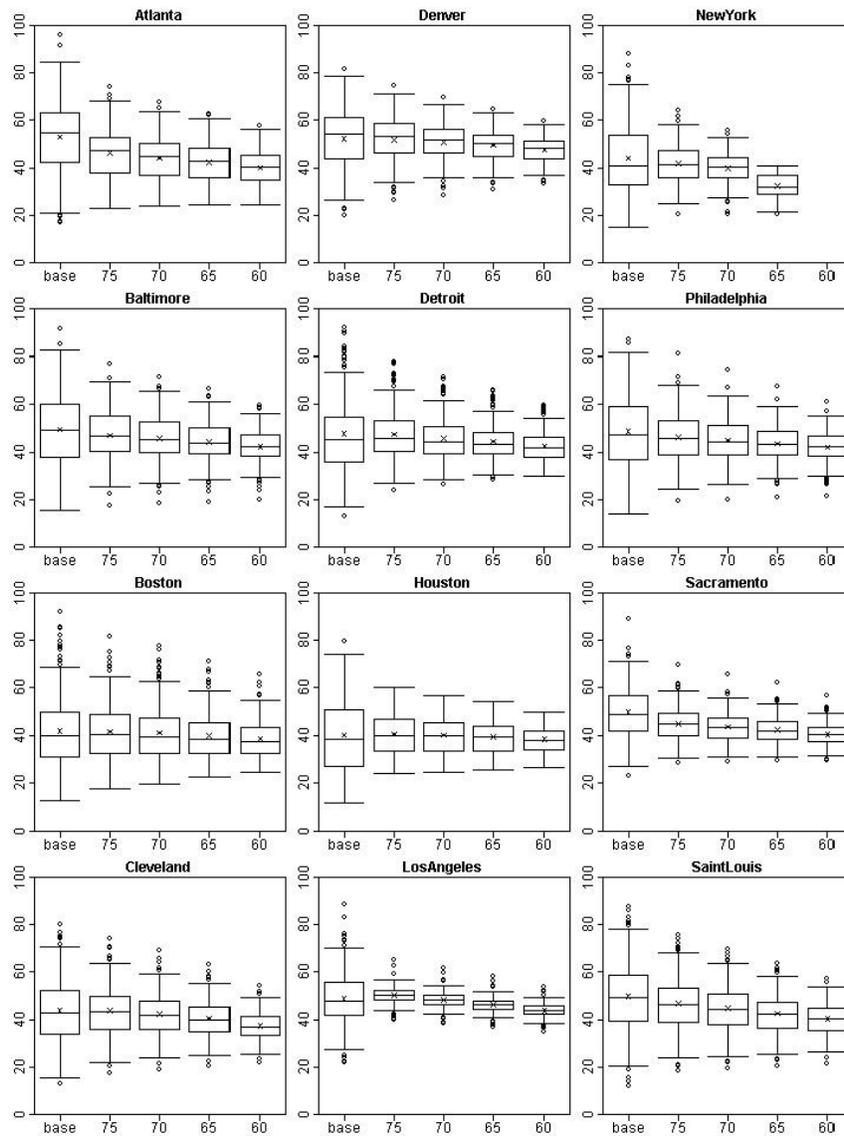


Figure 4-9. Distributions of Composite Monitor Daily Maximum 8-hr Average Values for the 12 Urban Study Areas in the Epidemiology-based Risk Assessment. Plots depict values based on ambient measurements (base), and values obtained with the HDDM adjustment methodology when just meeting the 75, 70, 65 and 60 ppb standards. Values shown are based on CBSAs for April-October of 2007. Note that the HDDM adjustment technique was not able to adjust air quality to just meet a 60 ppb standard in New York, so no boxplot is shown for that case. Boxes represent the median and quartiles, X's represent mean values, whiskers extend up to 1.5x the inter-quartile range from the boxes, and circles represent outliers.

Figure 3-1. Figure 4-9 from EPA (2014b).

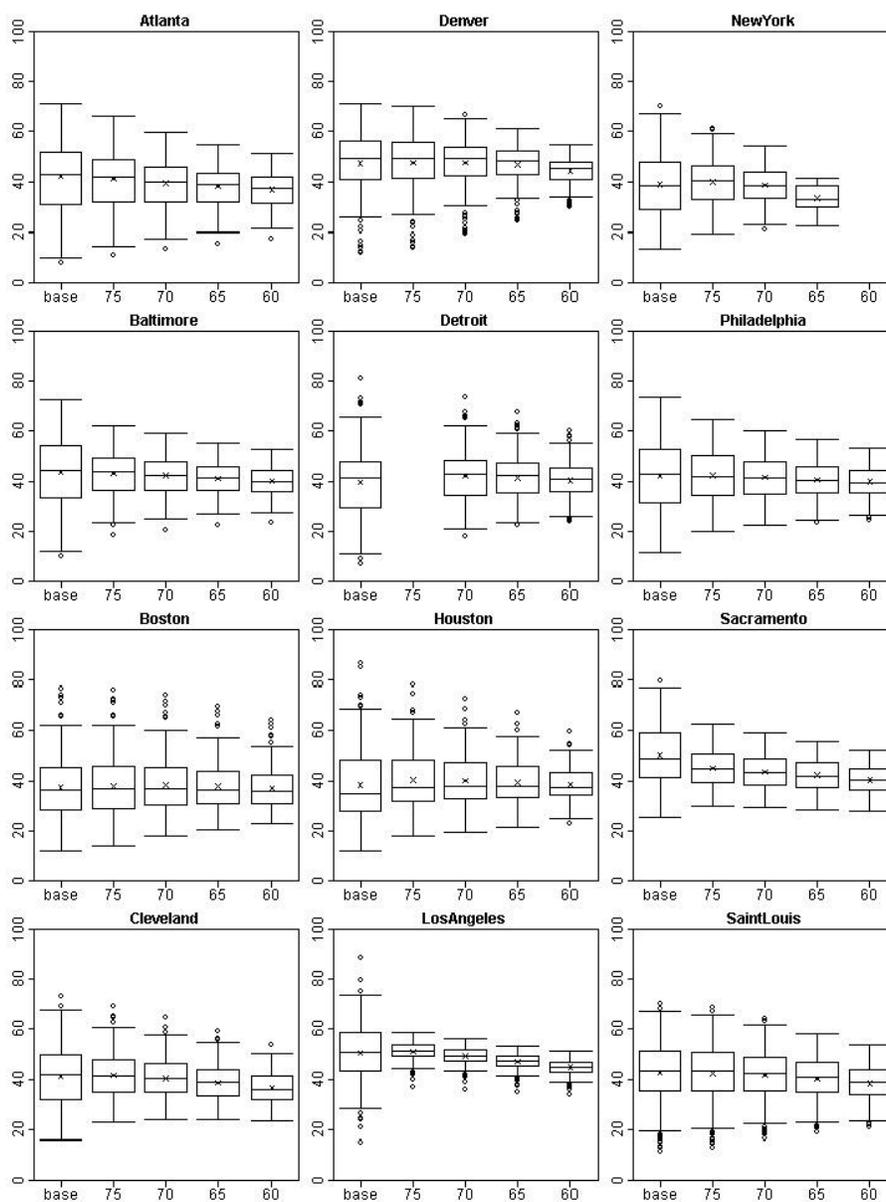


Figure 4-10. Distributions of Composite Monitor Daily Maximum 8-hr Average Values for the 12 Urban Study Areas in the Epidemiology-based Risk Assessment. Plots depict values based on ambient measurements (base), and values obtained with the HDDM adjustment methodology when just meeting the 75, 70, 65 and 60 ppb standards. Values shown are based on CBSAs for April-October of 2009. Note that Detroit air quality was meeting 75 ppb in 2008-2010, and the HDDM adjustment technique was not able to adjust air quality to just meet a 60 ppb standard in New York, so no boxplots are shown for those cases. Boxes represent the median and quartiles, X's represent mean values, whiskers extend up to 1.5x the inter-quartile range from the boxes, and circles represent outliers.

Figure 3-2. Figure 4-10 from EPA (2014b).

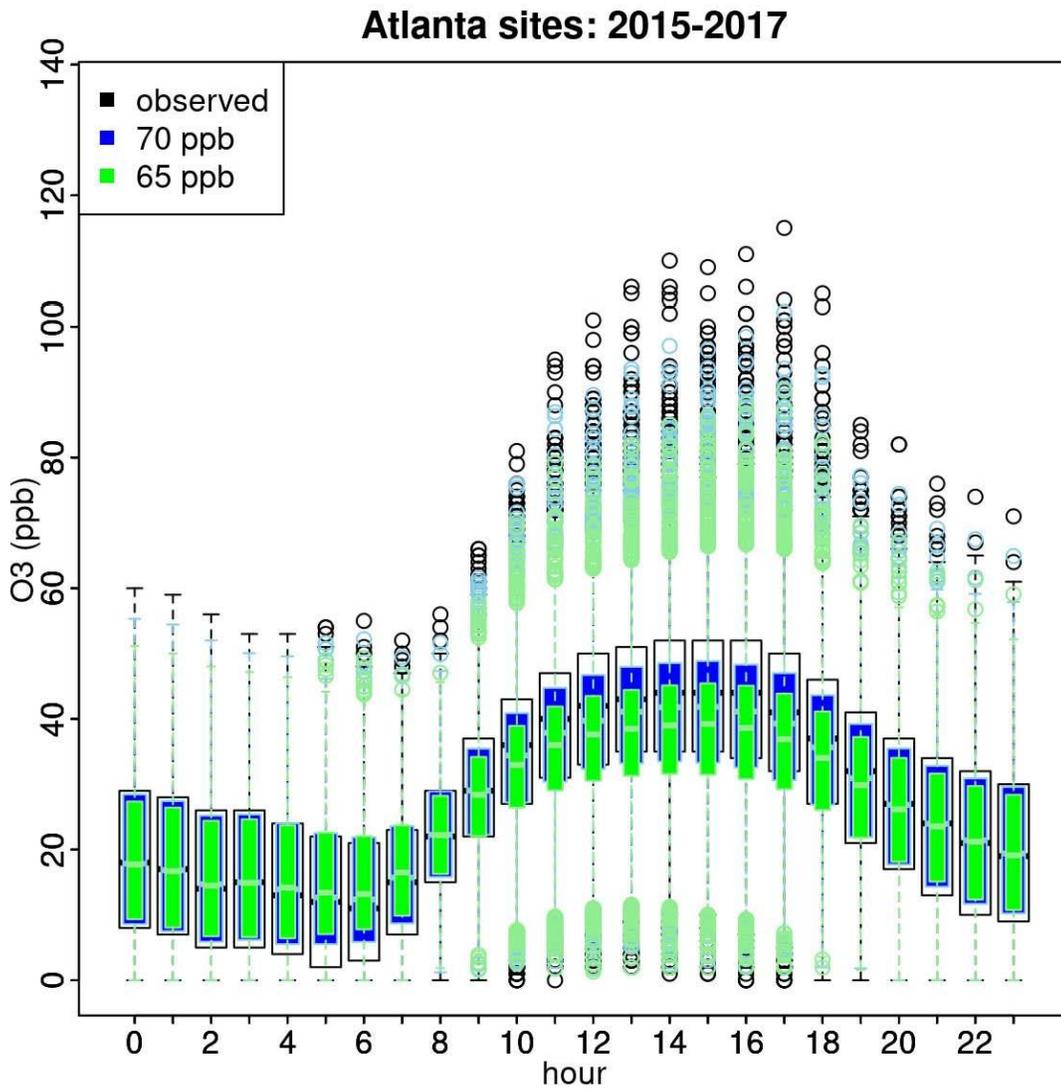


Figure 3C-67. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Atlanta.

Figure 3-3. Figure 3C-67 from EPA (2023).

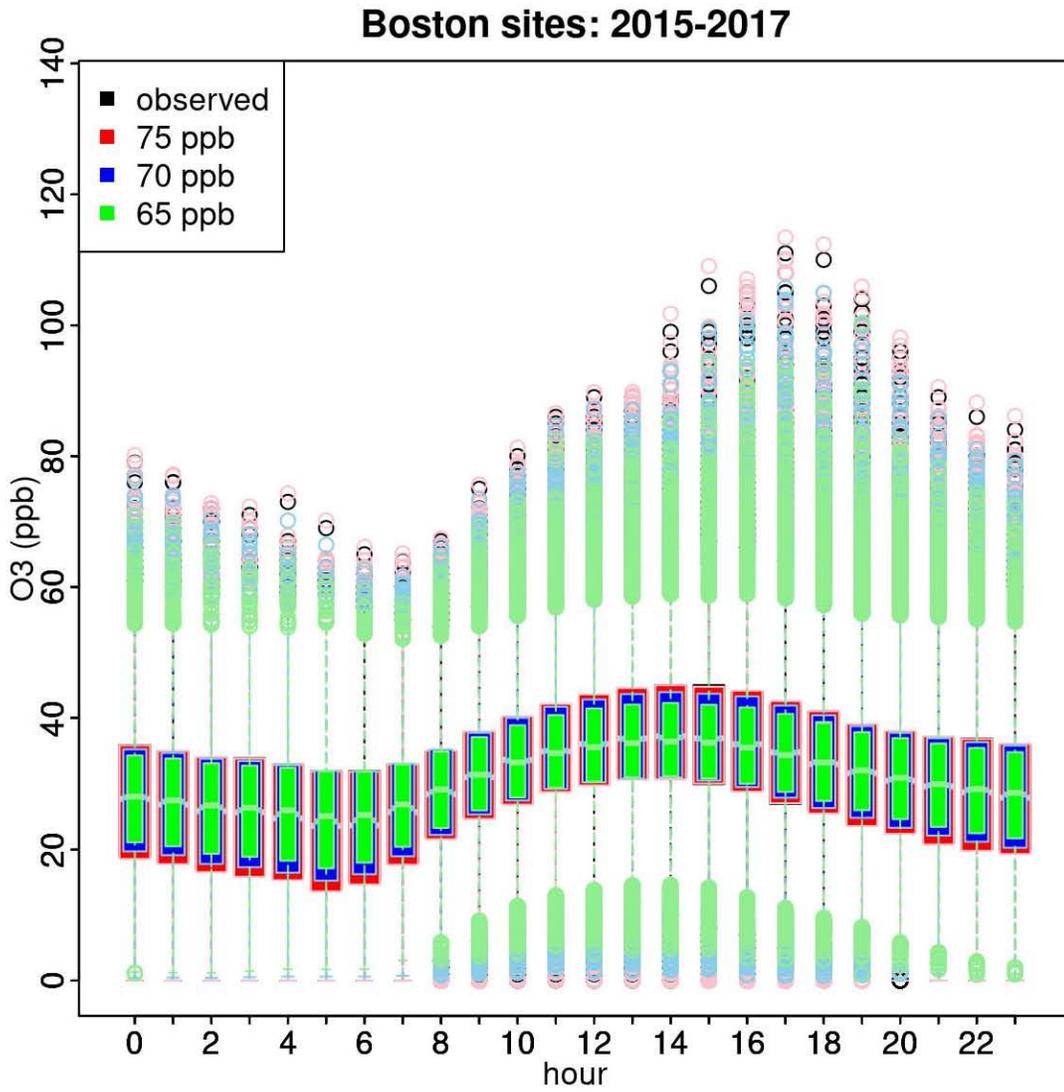


Figure 3C-68. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Boston.

Figure 3-4. Figure 3C-68 from EPA (2023).

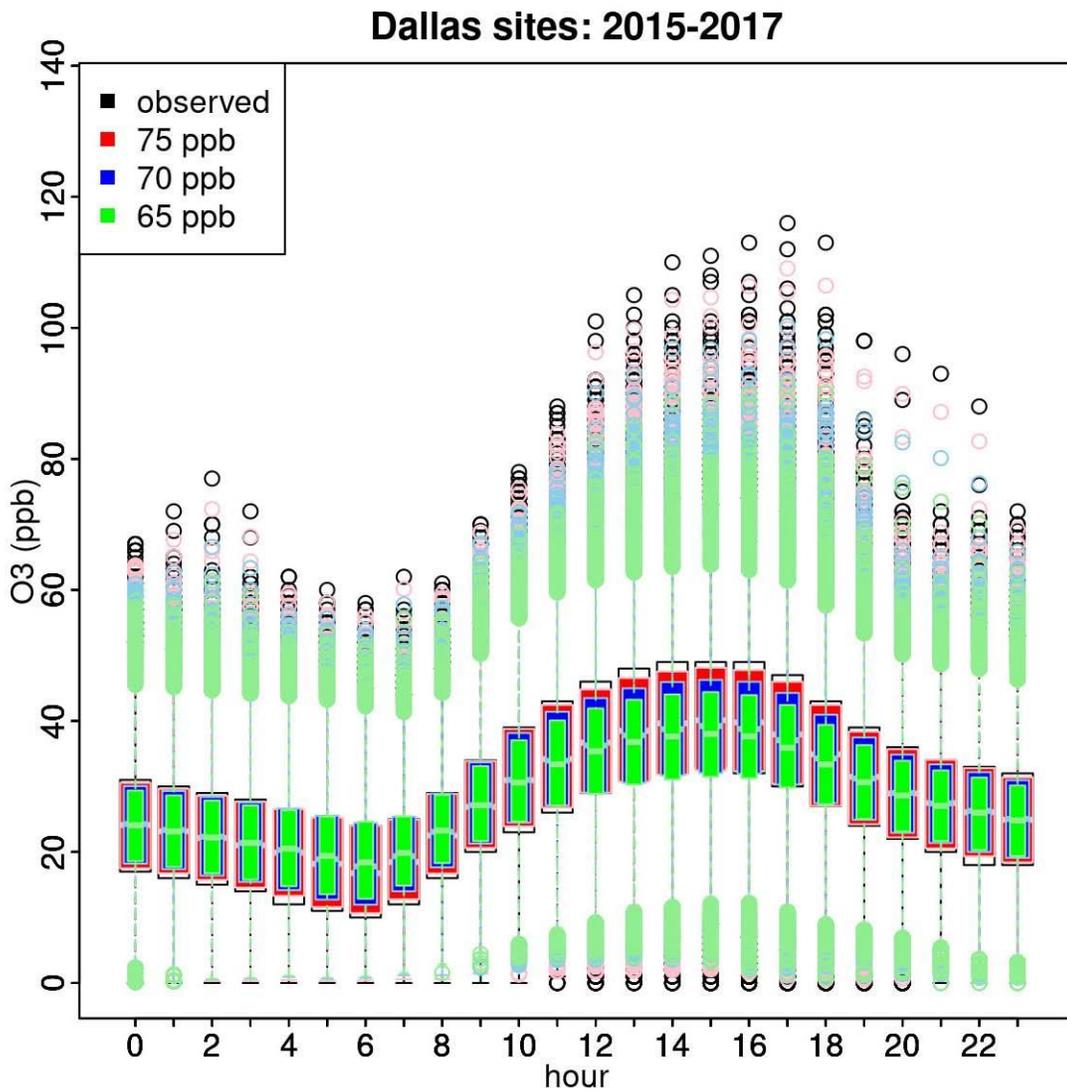


Figure 3C-69. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Dallas.

Figure 3-5. Figure 3C-69 from EPA (2023).

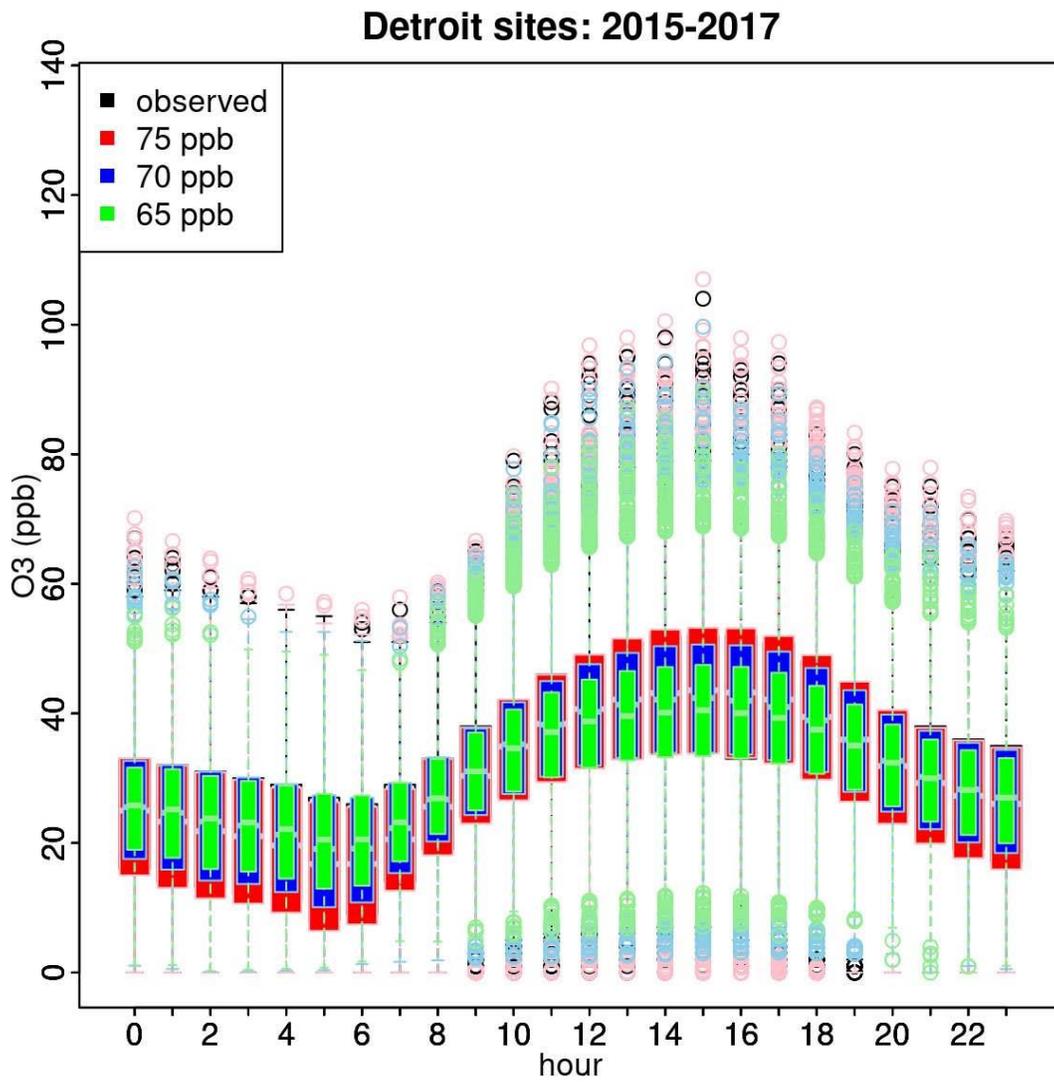


Figure 3C-70. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Detroit.

Figure 3-6. Figure 3C-70 from EPA (2023).

Philadelphia sites: 2015-2017

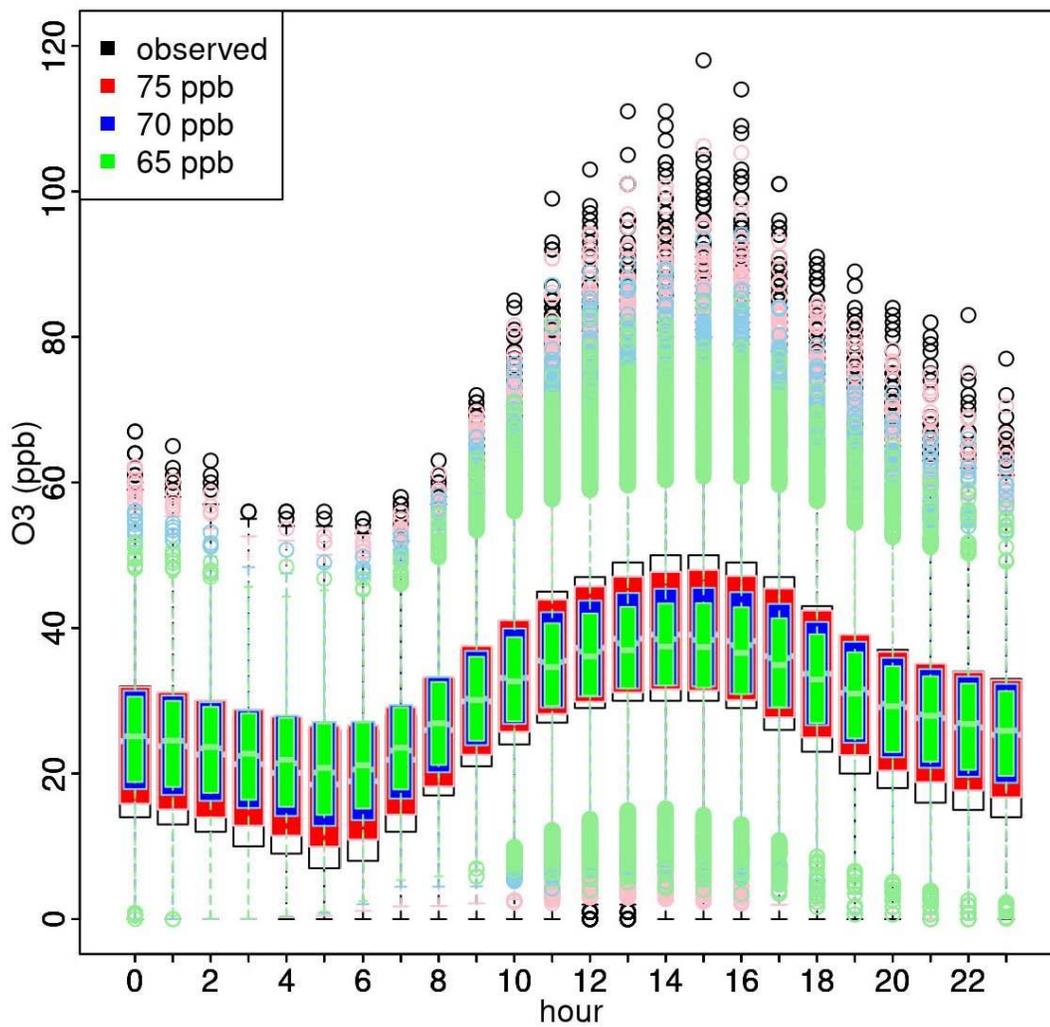


Figure 3C-71. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Philadelphia.

Figure 3-7. Figure 3C-71 from EPA (2023).

Phoenix sites: 2015-2017

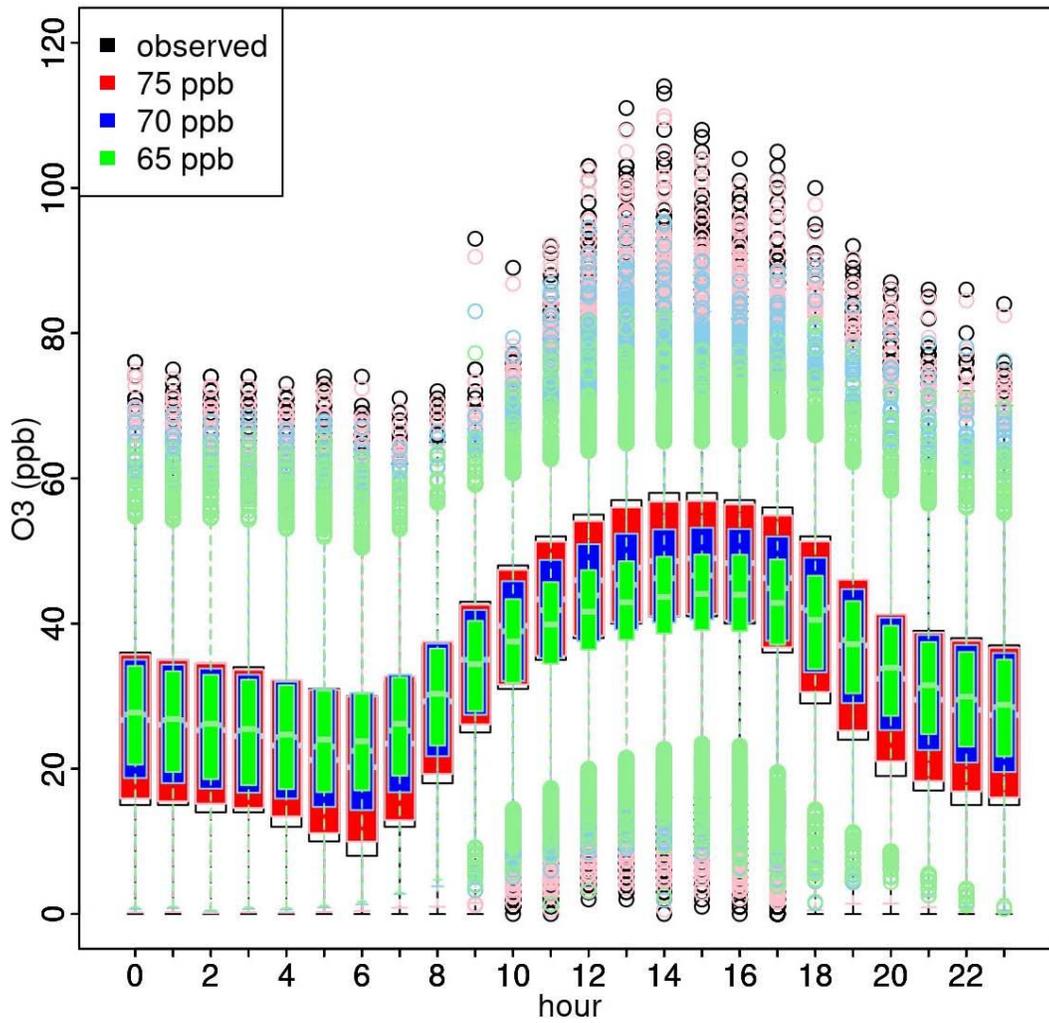


Figure 3C-72. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Phoenix.

Figure 3-8. Figure 3C-72 from EPA (2023).

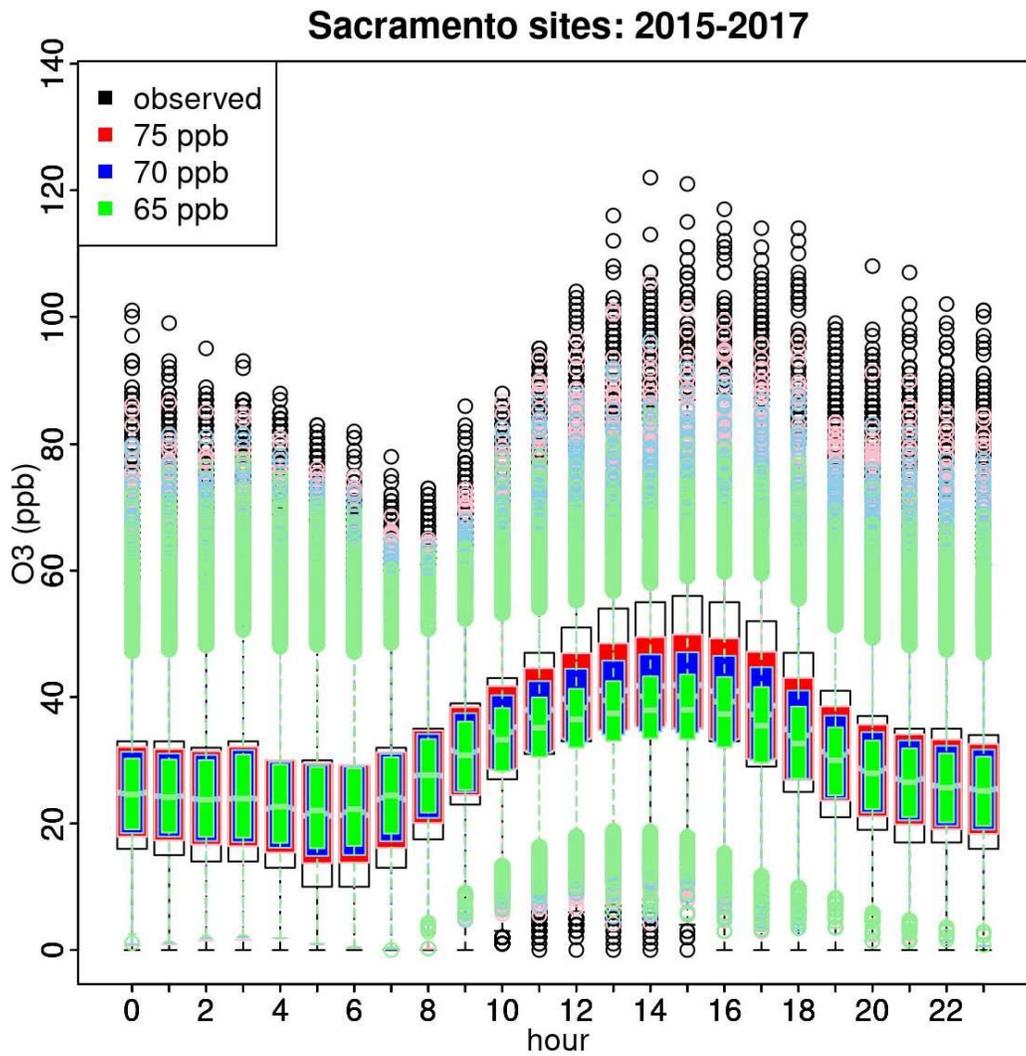


Figure 3C-73. Diurnal distribution of hourly O₃ concentrations at monitoring sites in Sacramento.

Figure 3-9. Figure 3C-73 from EPA (2023).

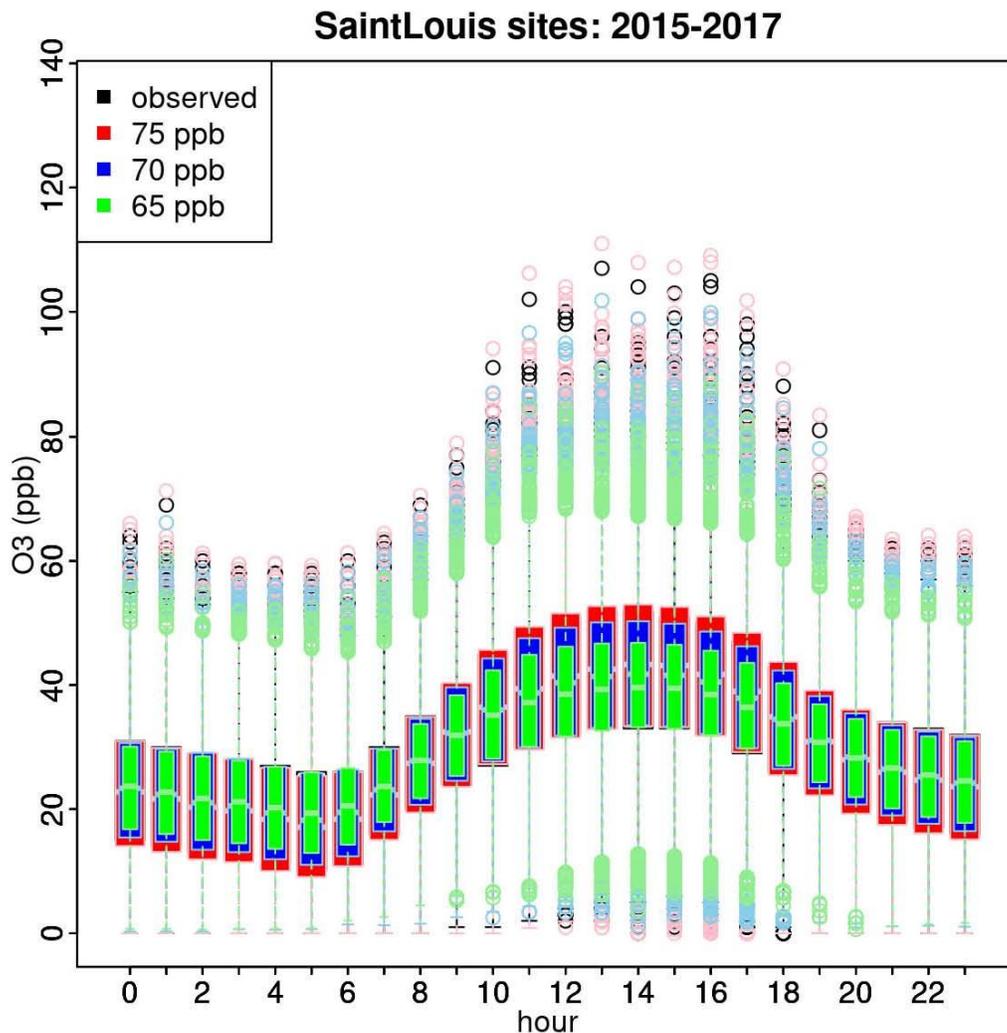


Figure 3C-74. Diurnal distribution of hourly O₃ concentrations at monitoring sites in St. Louis.

Figure 3-10. Figure 3C-74 from EPA (2023).

The second draft PA (EPA, 2023, page 3C-109) discusses the use of density scatter plots (Fig. 3C-83 through Fig. 3C-90) (Figs. 3-11 – 3-18) that show the change in MDA8 O₃ concentrations versus the observed concentrations based on the hourly VNA (Voronoi Neighbor Averaging) estimates in each study area. In each of these figures, the left-hand panel shows the observed MDA8 values (x-axis) versus the change in those values that occur when air quality is adjusted for the 75 ppb scenario (y-axis). The middle panel shows the MDA8 values for air quality adjusted to meet the 75 ppb scenario (x-axis) versus the additional change in those values that occur when air quality is adjusted to meet the current standard of 70 ppb (y-axis). Finally,

the right-hand panels show the corresponding changes from the current standard to the 65 ppb scenario. Within each panel, the x and y values are rounded to the nearest integer and colored to show the relative frequency of each 1 x 1 ppb square within the plot region. Values falling outside of the plot region were set to the nearest value within the plot region, and frequencies above the range in the color bar were set to the highest value within the color bar. The second draft PA (EPA, 2023, page 3C-110) concludes

In general, the density scatter plots show that the HDDM adjustment procedure predicts increases in MDA8 O₃ at low ambient air concentrations and decreases in MDA8 O₃ at high concentrations (Figure 3C-83 through Figure 3C-90). The vast majority of the increases in MDA8 O₃ occur at ambient air concentrations below 50 ppb. The relationship between the starting concentrations and the changes in these values based on the HDDM adjustments is fairly linear with strong negative correlation in all eight urban study areas. In some study areas, such as Philadelphia and Detroit, there is a bimodal pattern near the center of the distribution, which may be indicative of differing behavior near the urban population center versus the surrounding suburban areas.

The maps reveal consistent spatial patterns of O₃ changes across the urban study areas. The design values generally decreased when air quality was adjusted to meet the current standard of 70 ppb and continued to decrease when air quality was further adjusted for the 65 ppb scenario (Figure 3C-91 through Figure 3C-106). The design values tend to decrease more quickly in suburban and rural areas than in the urban population centers. The May-September “seasonal” average MDA8 values also followed this trend to some extent, although the behavior in the urban population centers varied slightly amongst the urban study areas (Figure 3C-107 through Figure 3C-114). In summary, these figures show that using the CAMx/HDDM adjustment methodology, peak O₃ concentrations are reduced in urban study areas with large domain-wide reductions in U.S. anthropogenic NO_x emissions.

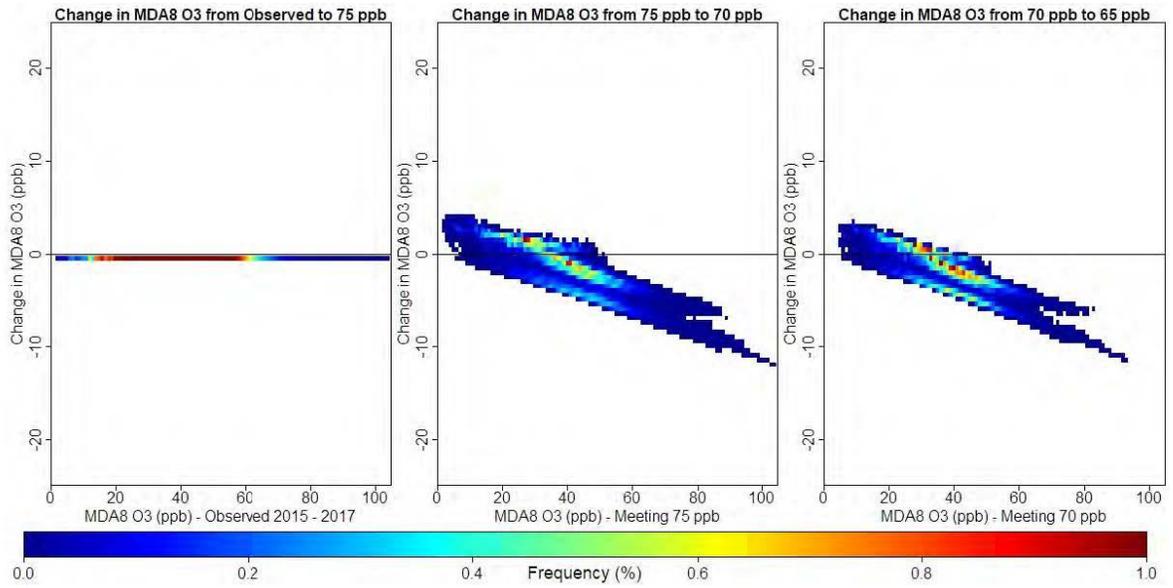


Figure 3C-83. Changes in MDA8 O₃ based on HDDM adjustments in the Atlanta study area.

Figure 3-11. Figure 3C-83 from EPA (2023).

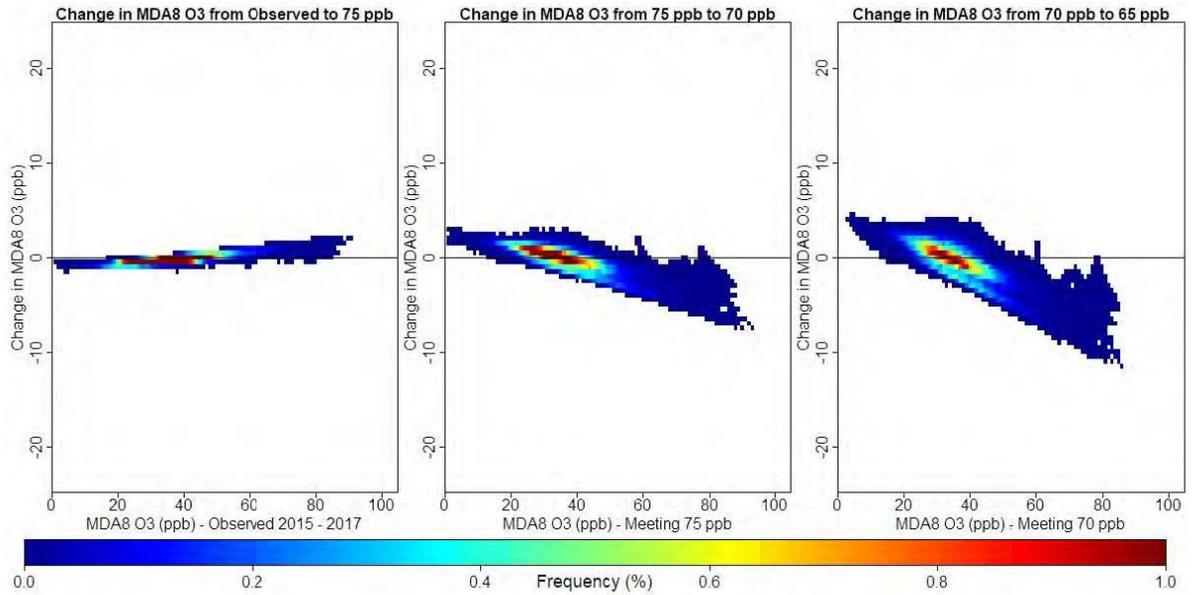


Figure 3C-84. Changes in MDA8 O₃ based on HDDM adjustments in the Boston study area.

Figure 3-12. Figure 3C-84 from EPA (2023).

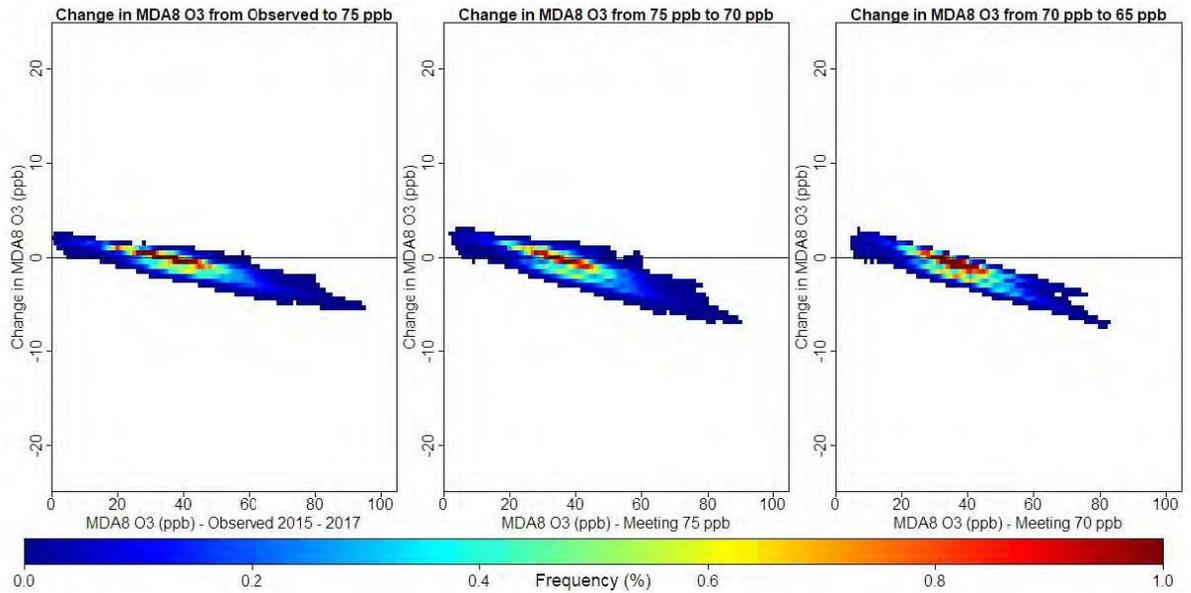


Figure 3C-85. Changes in MDA8 O₃ based on HDDM adjustments in the Dallas study area.

Figure 3-13. Figure 3C-85 from EPA (2023).

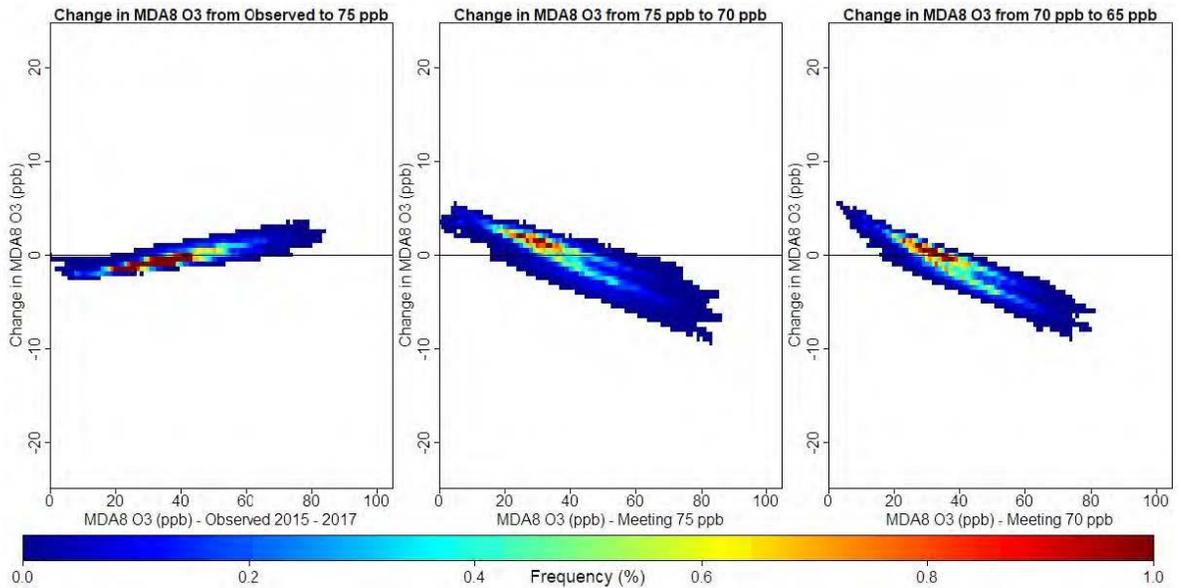


Figure 3C-86. Changes in MDA8 O₃ based on HDDM adjustments in the Detroit study area.

Figure 3-14. Figure 3C-86 from EPA (2023).

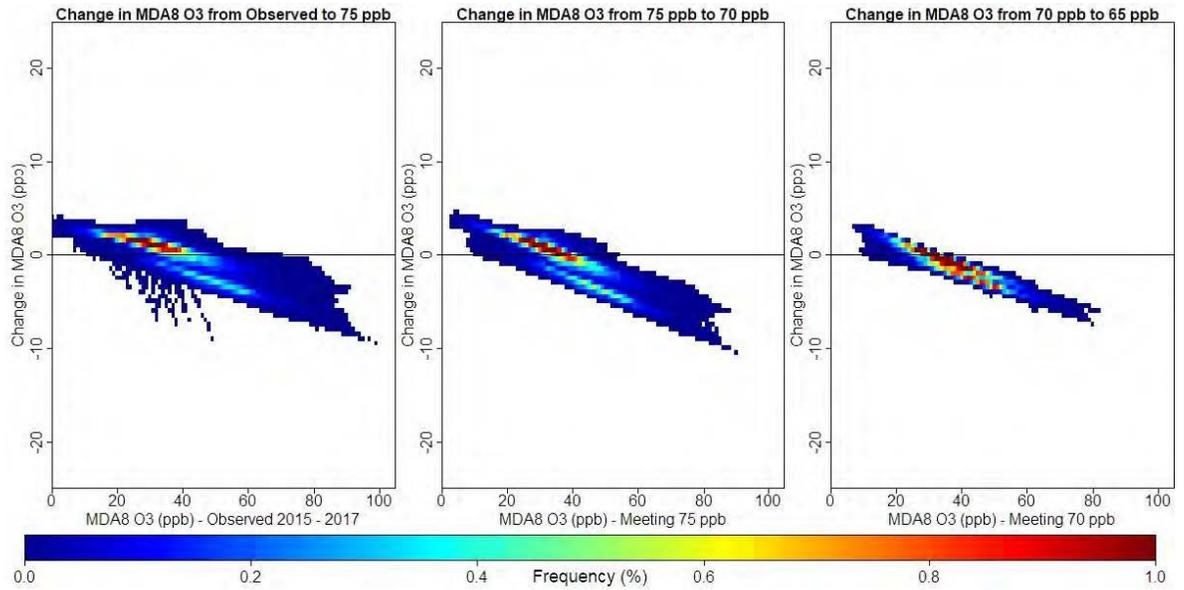


Figure 3C-87. Changes in MDA8 O₃ based on HDDM adjustments in the Philadelphia study area.

Figure 3-15. Figure 3C-87 from EPA (2023).

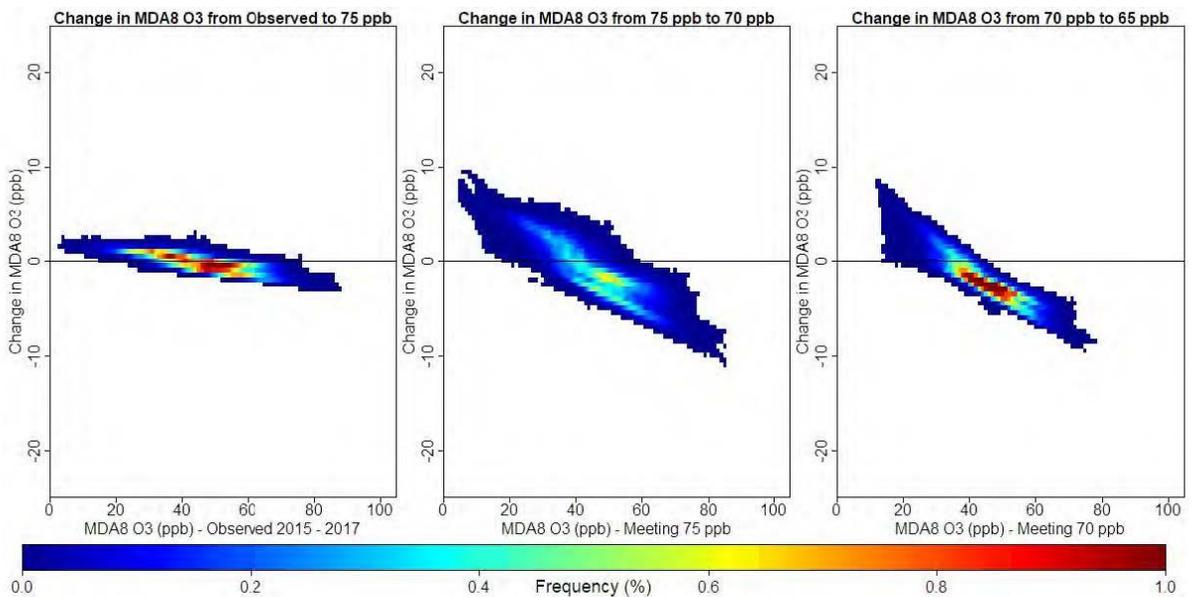


Figure 3C-88. Changes in MDA8 O₃ based on HDDM adjustments in the Phoenix study area.

Figure 3-16. Figure 3C-88 from EPA (2023).

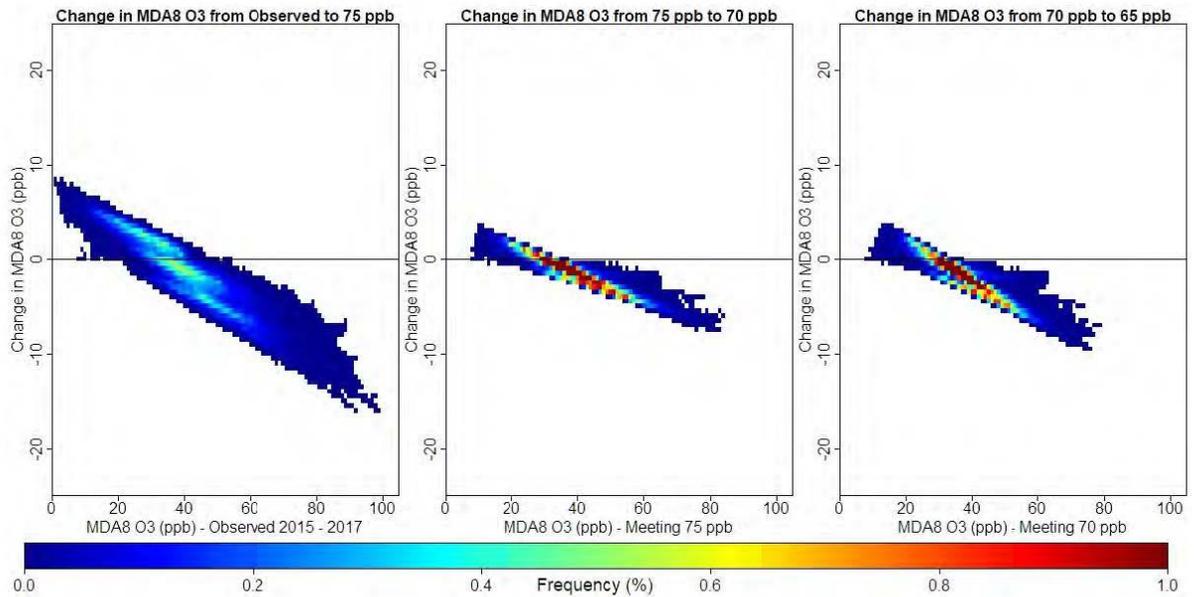


Figure 3C-89. Changes in MDA8 O₃ based on HDDM adjustments in the Sacramento study area.

Figure 3-17. Figure 3C-89 from EPA (2023).

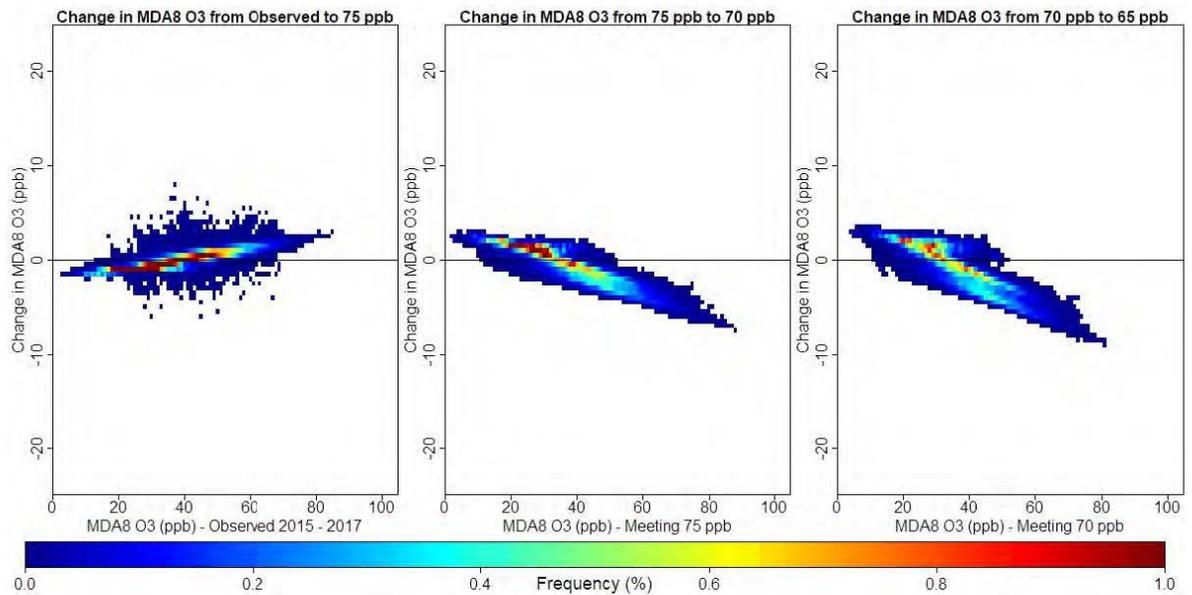


Figure 3C-90. Changes in MDA8 O₃ based on HDDM adjustments in the St. Louis study area.

Figure 3-18. Figure 3C-90 from EPA (2023).

Annual and summer mean and median hourly O₃ concentrations have been used by some researchers to characterize trends, assess human health long-term effects, and evaluate global

models. However, because emission reductions result in the high end shifting downward and the low end of the distribution of hourly average O₃ concentrations shifting toward the mid-level values, the average or median values *increase* at some sites. There are varying levels of agreement between trends in mean and median concentrations versus different metrics associated with the higher hourly or 8-h average concentrations. Lefohn et al. (2018) reported that trends in the 8-h average metric (an index associated with the high end of the distribution) bear the least resemblance to trends in the mean and median values with generally less than 50% of sites analyzed having trends in the same direction. Lefohn et al. (2018) reported overall that trends in the mean/median metrics were not representative of the trend behavior of those exposure metrics that focused on the higher end of the distribution. In Fig. 3-19 (reproduced from Lefohn et al., 2018), trend patterns for monthly average concentrations, annual SOMO35, and annual 4th highest daily maximum 8-h concentration (A4MDA8) exposure metrics at a suburban site in Philadelphia, Pennsylvania are compared. The SOMO35 is defined in the EU as the annual sum of the positive differences between the daily maximum 8-h O₃ average value and the cutoff value set at 35 ppb calculated for all days in a year. The monthly average concentrations significantly increased for seven of the 12 months, and were never estimated to decrease, while the SOMO35 and the A4MDA8 metrics, which focused on the higher hourly average concentrations, significantly decreased.

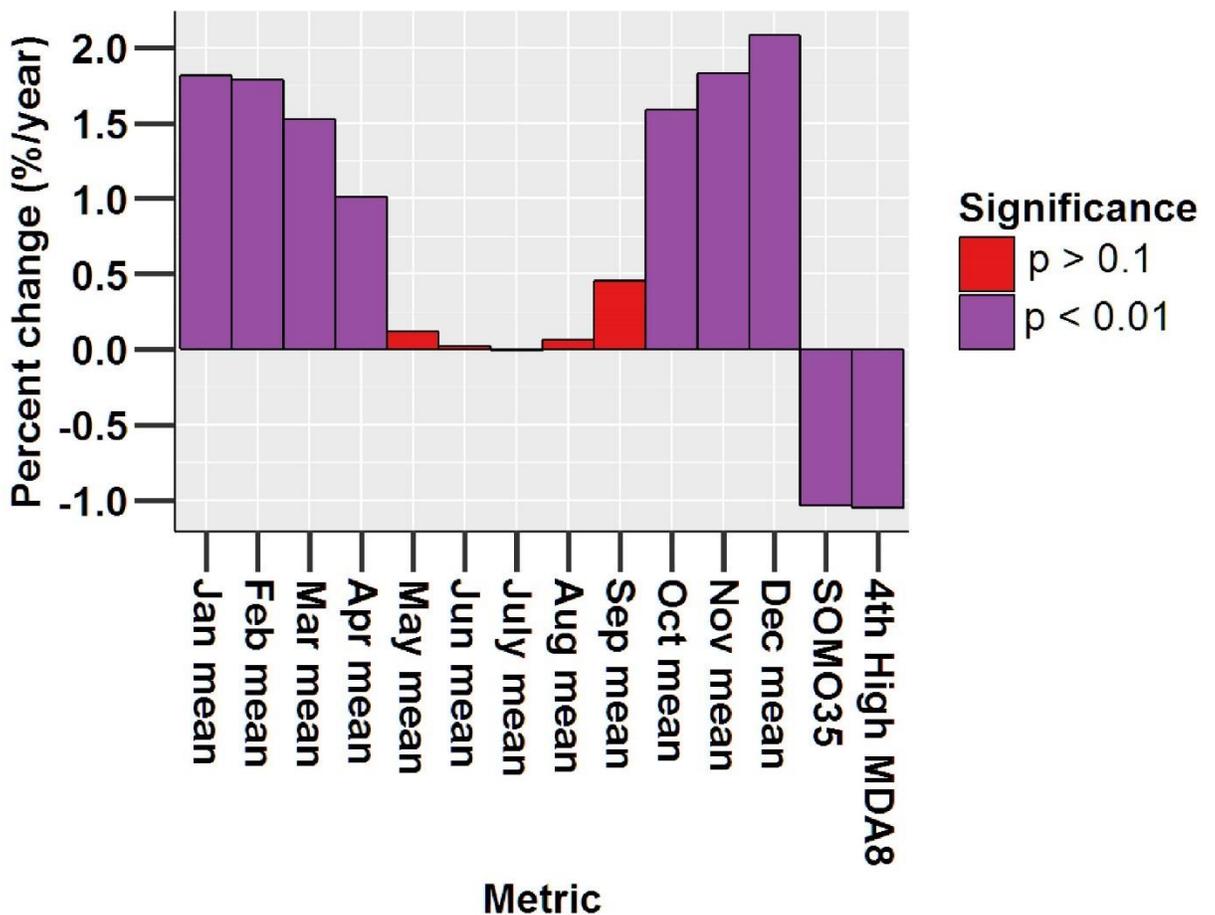


Figure 3-19. The Theil-Sen (%/year) trend in monthly average O₃ levels and the annual SOMO35 and 4th highest MDA8 human health metrics (A4MDA8) for a suburban site for 1980-2013 in Philadelphia, Pennsylvania (EPA AQS ID: 421010024-1). The $p < 0.05$ value was used to determine significance using the Mann-Kendall test. (Source: Lefohn et al., 2018).

Wells et al. (2021) noted that studies have shown that reductions in peak O₃ concentrations achieved by reducing precursor emissions often do not translate into reductions in seasonal mean O₃ concentrations, especially in urban areas (Simon et al., 2015; Lefohn et al., 2017). Accordingly, Wells et al. (2021) described improvements to the current method used by the EPA to adjust O₃ trends for meteorological influences by making refinements to the input data sources and by allowing the underlying statistical model to vary locally using a variable selection procedure. The authors explored the effects of meteorology on peak O₃ concentrations by developing quantile regression methods for adjusting trends in the 90th and 98th percentiles of the distribution of May to September MDA8 O₃ concentrations.

For the period 2000 – 2021, Figs. 3-20 and 3-21 illustrate a comparison of the annual 4th highest 8-h daily average concentrations with the annual averages of the hourly average concentrations for 5 sites (Simi Valley in southern CA; Queens, NY; Denali National Park, AK; Voyageurs National Park, MN; and Yellowstone National Park, WY. Note that the Simi Valley site in southern California does not experience O₃ exposures that are as high as the design values associated with sites located in San Bernardino County, where in most years the highest design values are experienced in the U.S. For the period 2019-2021, the design value was 0.114 ppm for San Bernardino County. In Fig. 3-20, as anticipated, the two urban sites (i.e., Simi Valley and Queens, NY) experience the highest annual 4th highest daily maximum 8-h average O₃ concentrations in comparison to the three rural National Park sites.

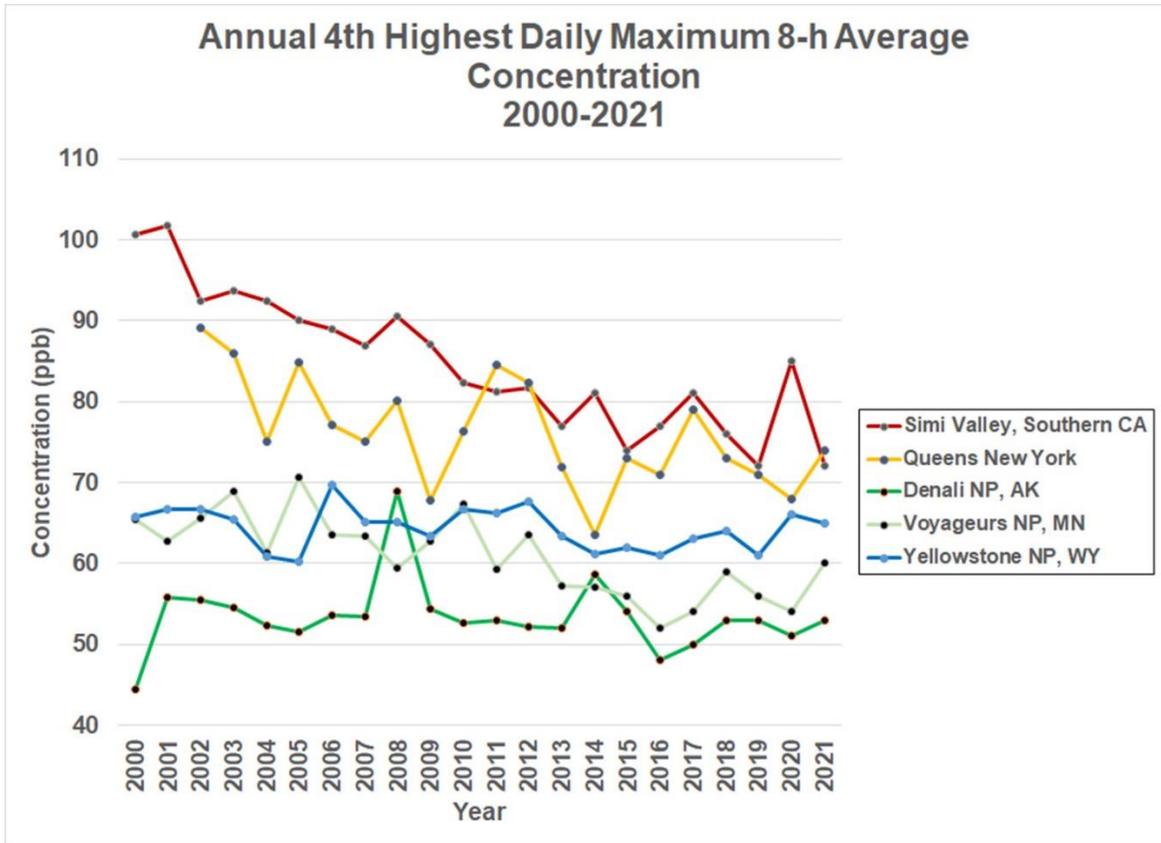


Figure 3-20. The annual 4th highest daily maximum 8-h average O₃ concentration for the period 2000-2021 for Simi Valley, CA (061112002), Queens New York, NY (360810124), Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011).

When the annual average of the hourly average concentrations is calculated for the 5 sites, the Yellowstone National Park site exhibits the highest average concentration values (Fig. 3-21). The annual average values for Voyageurs National Park, Denali National Park, and Simi Valley are similar in value. The Queens, NY site experiences the lowest annual average values, which are increasing over time.

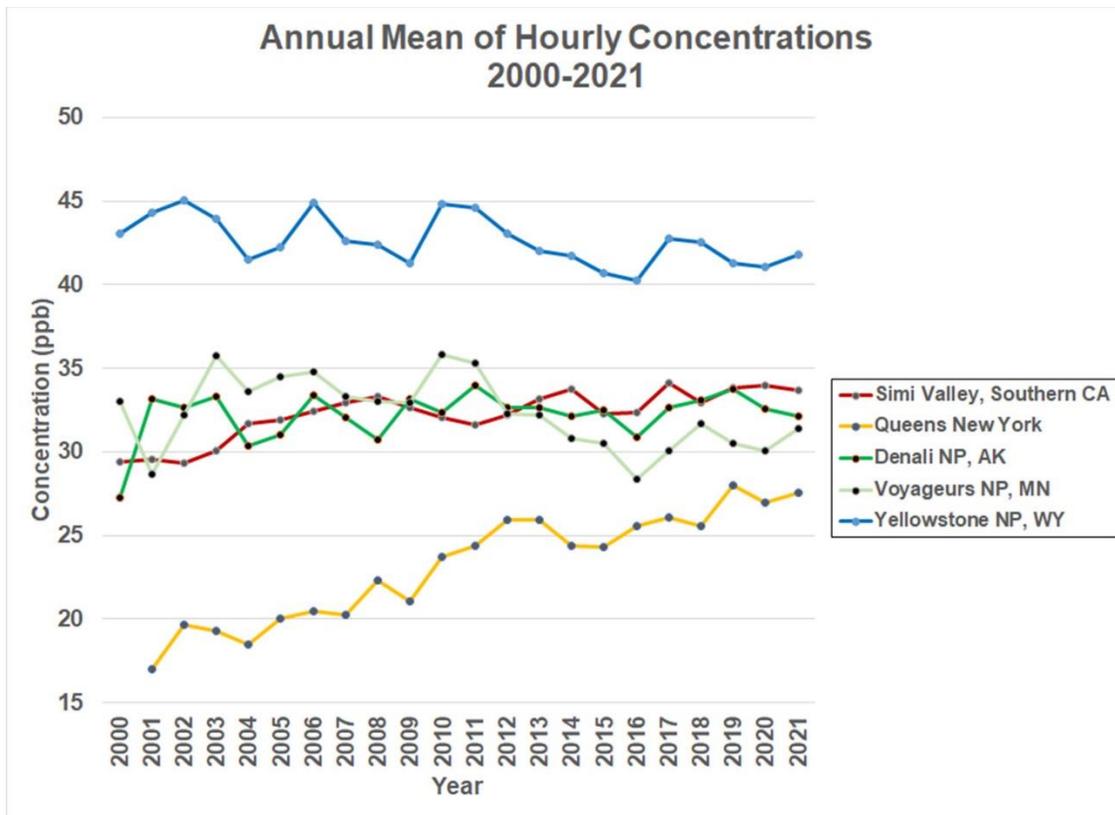


Figure 3-21. The annual mean of the hourly average O₃ concentrations for the period 2000-2021 for Simi Valley, CA (061112002), Queens New York, NY (360810124, Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011).

By calculating a long-term average concentration exposure metric, which combines all hourly values into one number, the highest hourly average concentrations in many cases no longer influence the resulting number because there are many more low- and mid-level values than the higher concentrations. The result of calculating a long-term average is that the annual average values at many of the O₃ monitoring sites have similar values, even though some sites experience elevated hourly O₃ average concentrations, and some do not. For example, in the annual mean figure (Fig. 3-21), the ordering of the sites from the highest to the lowest annual means would appear to be counter intuitive. While the Simi Valley site in southern California experiences the highest 8-h average O₃ exposures of the 5 sites (Fig. 3-20), the annual average concentration for the site is comparable to values for the National Park sites. The three National Park sites in the annual average figures do not experience high 8-h average concentration values comparable to many of the urban sites in the U.S. Based on the annual mean of hourly average concentrations, the high-elevation Yellowstone National Park site experiences much higher average values than any of the other 4 sites. The moderate (i.e., compared to many urban, suburban sites) hourly average O₃ concentrations experienced at Yellowstone National Park (WY) are influenced by frequent occurrences of stratospheric-tropospheric transport to the surface, which is a naturally occurring process that contributes to background O₃ levels (Lefohn

et al., 2001, 2011, 2012, 2014; EPA, 2014a). Fig. 3-22 illustrates for 2007 (modeled background O₃ data provided by EPA) the relationship between background O₃ levels (defined as apportionment-based USB and referred to as USB_{AB}), stratospheric-tropospheric transport to the surface at the site (STT-S), and the observed ambient daily maximum 8-hourly average concentrations. The term USB_{AB} will be discussed in Section 3.2. The frequency of STT-S trajectories that arrive at the surface at the site are greatest in the spring but occur throughout the year. Fig. 3-23 illustrates for the entire year (January-December 2006) similar results using Emission Influenced Background (EIB) estimates (see Lefohn et al., 2014) compared with the observed daily maximum 8-h average O₃ concentrations and daily STT-S trajectories. The trajectory model introduced by Wernli and Davies (1997) was used to identify days of high probability for STT trajectories to enhance surface O₃ at specific monitoring sites. The concept of EIB will be discussed as a measure for background O₃ in Section 3.2. An enhanced event occurred on 2 May 2006, when a maximum hourly average O₃ concentration of 89 ppb was measured at 19 UTC (Lefohn et al., 2011). The enhanced event can be seen in Fig. 3-23. There were over 140 STT trajectories that were estimated on that day to reach the surface at the O₃ monitoring site. Škerlak et al. (2019) have described the processes associated with this May 2006 event.

For the period 2000-2014, using data from the TOAR database (Schultz et al., 2017) and the Mann-Kendall nonparametric test, no statistically significant trends at the $p < 0.05$ value were observed at the Yellowstone National Park using either the seasonal 4th highest 8-h daily maximum concentration or the seasonal mean value metrics. Jaffe et al. (2018) in Fig. S1 (in their supplement), found no trend at Yellowstone NP for the April-September period for the 4th highest 8-h daily maximum concentration for the 2000-2014 period. No trend was detected at Yellowstone NP using the annual 4th highest 8-h daily maximum concentration metric for the 2000-2022 period (Lefohn, 2023 - private communication). It is important that metrics used for assessing trends at sites influenced by natural stratospheric process include the entire 24-h period. For example, when calculating a metric, such as the 4th highest 8-h daily maximum concentration, the entire 24-h period is required to capture the influence of stratospheric events that enhance O₃ concentrations, which at times occur in the late evening or very early morning hours. Reviewing the STT-S daily events for Yellowstone National Park, background O₃ (i.e., USB_{AB}) appears to play a predominant role in influencing the observed ambient levels of O₃.

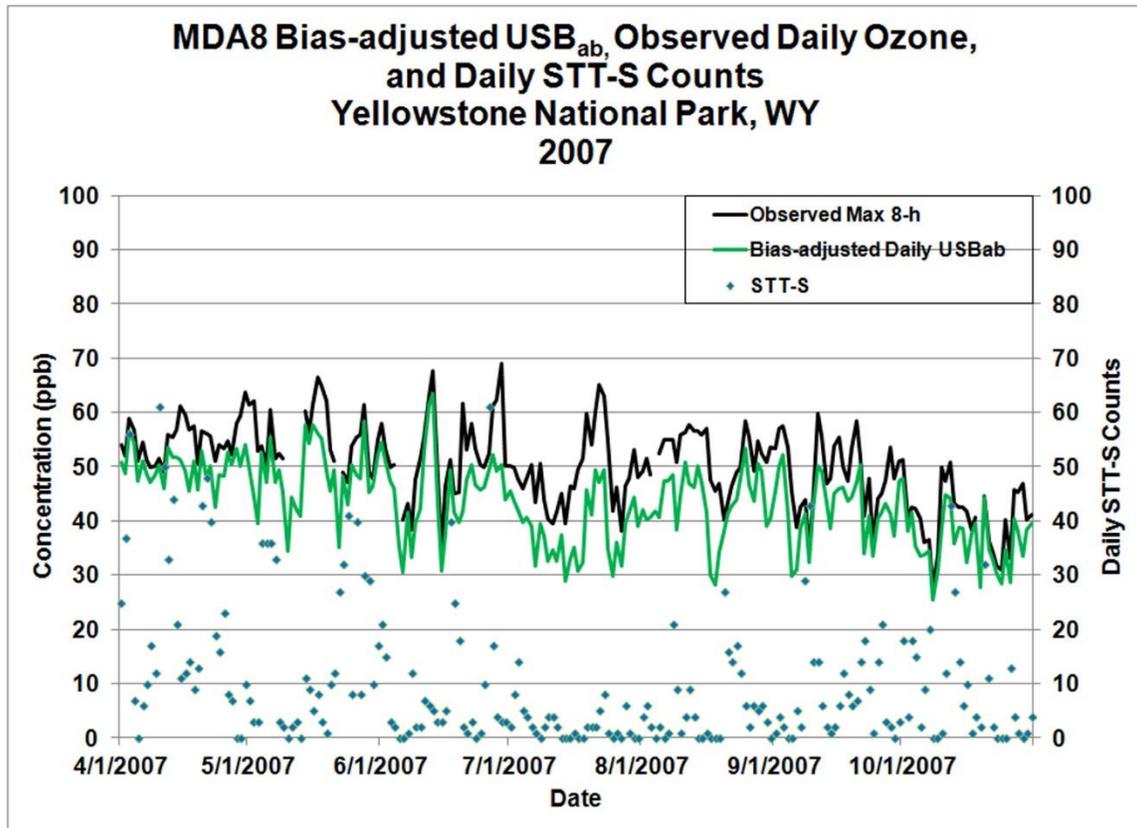


Figure 3-22. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for Yellowstone National Park (AQS ID 560391011) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details how the STT-S values are estimated.

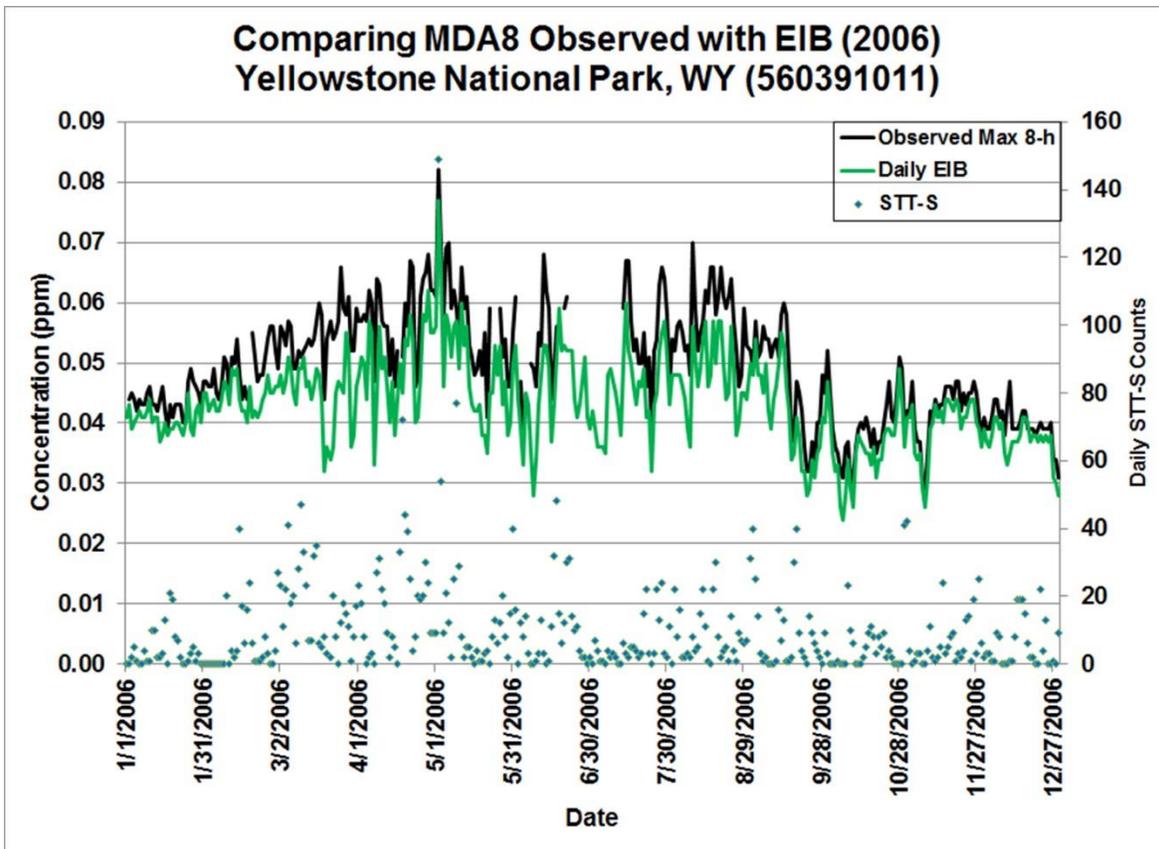


Figure 3-23. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted Emissions Influenced Background (EIB) 8-h daily maximum concentrations for Yellowstone National Park (AQS ID 560391011) for January-December 2006. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. See Lefohn et al. (2014) for details how the estimated Emissions Influenced Background (EIB) and STT-S values are estimated. Source: Lefohn et al. (2014).

Using EPA AQS hourly average O₃ concentration data, information from the TOAR database (see Schultz et al., 2017) indicates that of 406 U.S. sites that reported both annual and 4th highest MDA 8 values, 29% of the sites experienced increasing annual average concentrations for the period 2000-2014, while 65% had no trend, and 6% experienced decreasing annual average concentrations. The nonparametric Mann-Kendall test was used for testing trends. Thus, a substantial number of the AQS sites show increasing trends using the annual average concentration metric.

The calculation of the annual mean concentration includes the low, middle, and high hourly average concentrations. Lefohn et al. (1998), Simon et al. (2015) and Lefohn et al. (2017, 2018) have discussed the effects of NO_x emissions on hourly average O₃ concentrations. As pointed out in the second draft PA (EPA, 2023), the greater the amount of NO_x emissions, the

greater the frequency of high and low hourly average concentrations for many low-elevation monitoring sites. Ozone is titrated by NO, with the result that the frequency of low hourly average values increases, and the frequency of the higher hourly levels increases due to precursor production of O₃. Simon et al. (2015) discussed the effects of reducing O₃ precursors in the United States on O₃ concentrations. Using daily 8-h average concentrations, the authors reported that decreasing O₃ trends generally occurred in the summer, in less urbanized areas, and at the upper end of the O₃ distribution (i.e., the higher 8-h concentrations). Conversely, **increasing** O₃ trends generally occurred in the winter, in more urbanized areas, and **at the lower end of the O₃ distribution**. The authors noted that increasing fifth percentile trends (i.e., the trends in the lower end of the distribution) were more common in the more highly urbanized areas. Simon et al. (2015) indicated that as anthropogenic NO_x emissions have decreased, the O₃ distribution has been compressed (i.e., less frequent high and low values), leading to less spatial and temporal variability. Lefohn et al. (2017, 2018) noted that there is both modeling and observational evidence that the reductions in the frequency of low levels (i.e., shifts of the lower levels upward) are associated with emissions reductions resulting in less O₃ titration by NO. Based on this, one would anticipate that as emissions were reduced, that annual O₃ averages would increase, while the highest 8-h average concentrations would decrease. This pattern has been described in the literature (Lefohn et al., 2017, 2018). As noted in Section 3.2.12, the COVID-19 lockdowns that occurred in 2020 throughout the world during the spring (Northern Hemisphere) and fall (Southern Hemisphere) resulted in O₃ increases when averaging metrics were used and O₃ decreases when MDA8 metrics were applied.

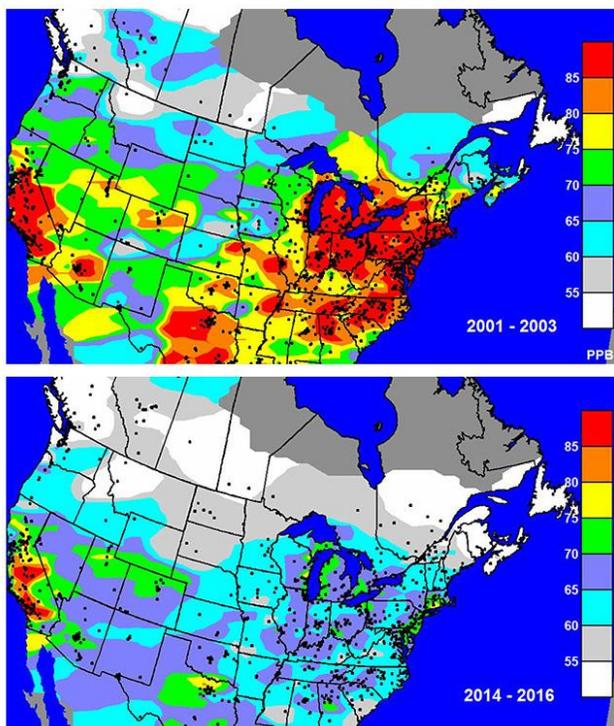
Zanis et al. (2022) presented an analysis of the effect of climate change on surface O₃ where they discussed the related penalties and benefits around the globe from the global modelling perspective based on simulations with five CMIP6 (Coupled Model Intercomparison Project Phase 6) Earth System Models. The authors noted that over regions remote from pollution sources, there was a robust decline in mean surface O₃ concentration on an annual basis, as well as for boreal winter and summer varying spatially from -0.2 to -2 ppb °C⁻¹, with strongest decline over tropical oceanic regions. Zanis et al. (2022) noted that O₃ increases over regions close to anthropogenic pollution sources or close to enhanced natural biogenic volatile organic compounds emission sources with a rate ranging regionally from 0.2 to 2 ppb C⁻¹, implying a regional surface O₃ penalty due to global warming. The authors observed that their multi-model analysis focused at large-scale responses of annual and seasonal surface O₃ to climate change on decadal climatic timescales. Zanis et al. (2022) noted that due to the non-linear nature of O₃ chemistry, their investigation of climate-driven changes in pollution episodes and O₃ extremes at hourly and daily temporal resolution was masked. They suggested that further work on O₃ extremes in selected polluted regions based on Earth System Models would be an asset for air pollution policies as heat waves and air pollution episodes pose a serious threat to human health and may worsen under future climate change depending on the scenario.

During the 2015 and 2020 O₃ rulemaking activities, EPA believed that both acute and chronic effects could be reduced by reducing the higher hourly average concentrations. As emissions were reduced, the higher part of the distribution of hourly average concentrations moved downwards toward the middle hourly average concentrations. By focusing on an emission reduction strategy to decrease the higher hourly average O₃ concentrations, the

potential for chronic and acute health and vegetation effects are reduced (Federal Register, 2015, 2020b).

3.1.2 Patterns of the Changes in Which Months the Highest O₃ Concentrations Occur as Emissions are Reduced

In the U.S., we have experienced significant reductions in O₃ levels. Figure 3-24 below compares the 3-year average of the annual 4th highest 8-h value between 2001-2003 with 2014-2016. As a result of emission reductions to attain reductions in O₃ exposures, important changes have occurred in the months when the highest O₃ concentrations are observed. Using models, Figs. 3-25 to 3-32 (EPA, 2023, Fig. 3C-75, page 3C-101 through Fig. 3C-82, page 3C-108) display the same information as Figs. 3-3 to 3-10 (Figure 3C-67 through Fig. 3C-74 in the second draft PA but for monthly rather than diurnal distributions. The figures illustrate the modeling results presented in the second draft PA document (EPA, 2023) for the shifting of the higher mid-range concentrations from the summer season toward the earlier months as emissions are reduced to attain the three scenarios discussed earlier.



Comparing the 3-year average of the annual 4th highest 8-h value for 2001-2003 versus 2014-2016. Analyses provided to AS Lefohn by Tom Dann, Canada.

A.S.L. & Associates,
Helena, MT

Figure 3-24. A comparison of the 3-year average of the annual 4th highest 8-h value for the period 2001-2003 with 2014-2016.

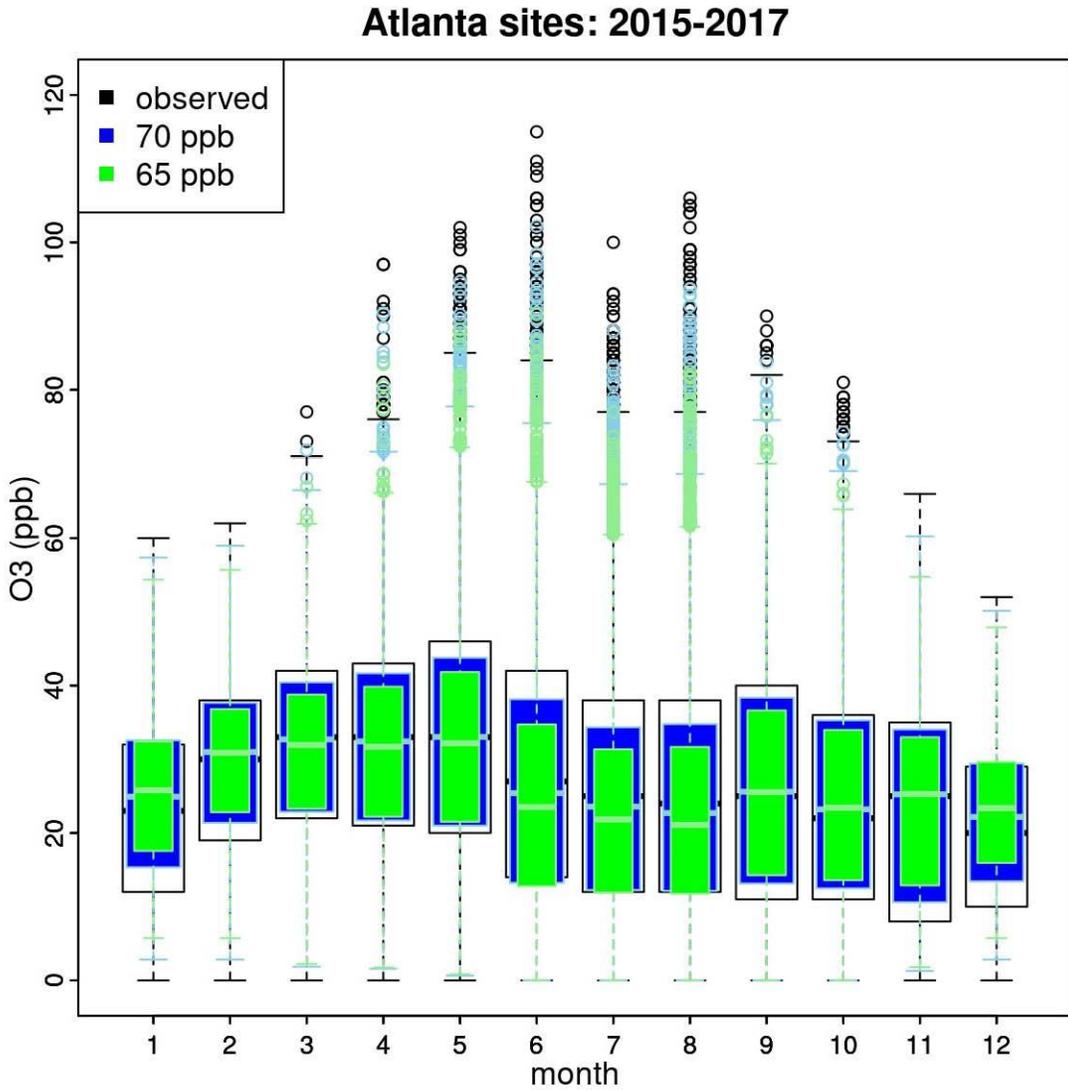


Figure 3C-75. Monthly distribution of hourly O₃ concentrations at monitoring sites in Atlanta.

Figure 3-25. Figure 3C-75 from EPA (2023).

Boston sites: 2015-2017

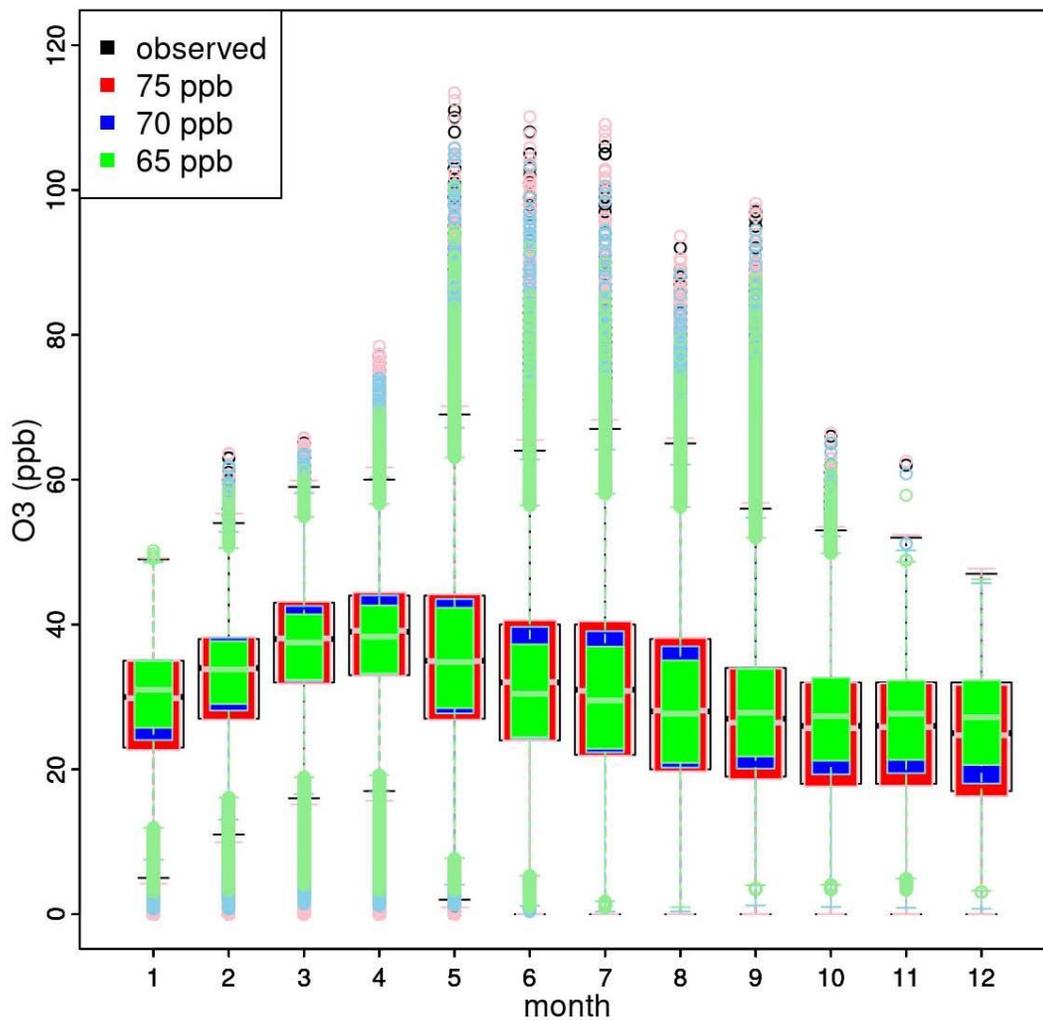


Figure 3C-76. Monthly distribution of hourly O₃ concentrations at monitoring sites in Boston.

Figure 3-26. Figure 3C-76 from EPA (2023).

Dallas sites: 2015-2017

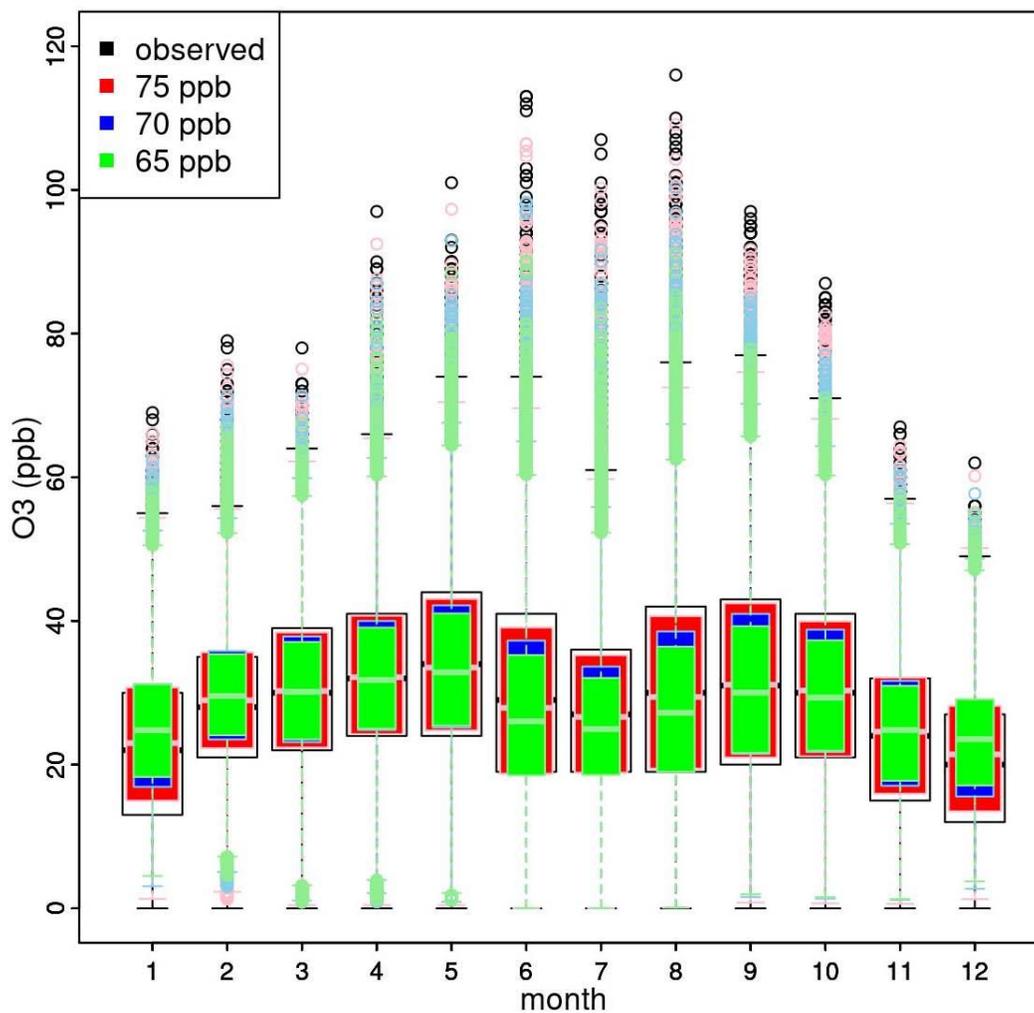


Figure 3C-77. Monthly distribution of hourly O₃ concentrations at monitoring sites in Dallas.

Figure 3-27. Figure 3C-77 from EPA (2023).

Detroit sites: 2015-2017

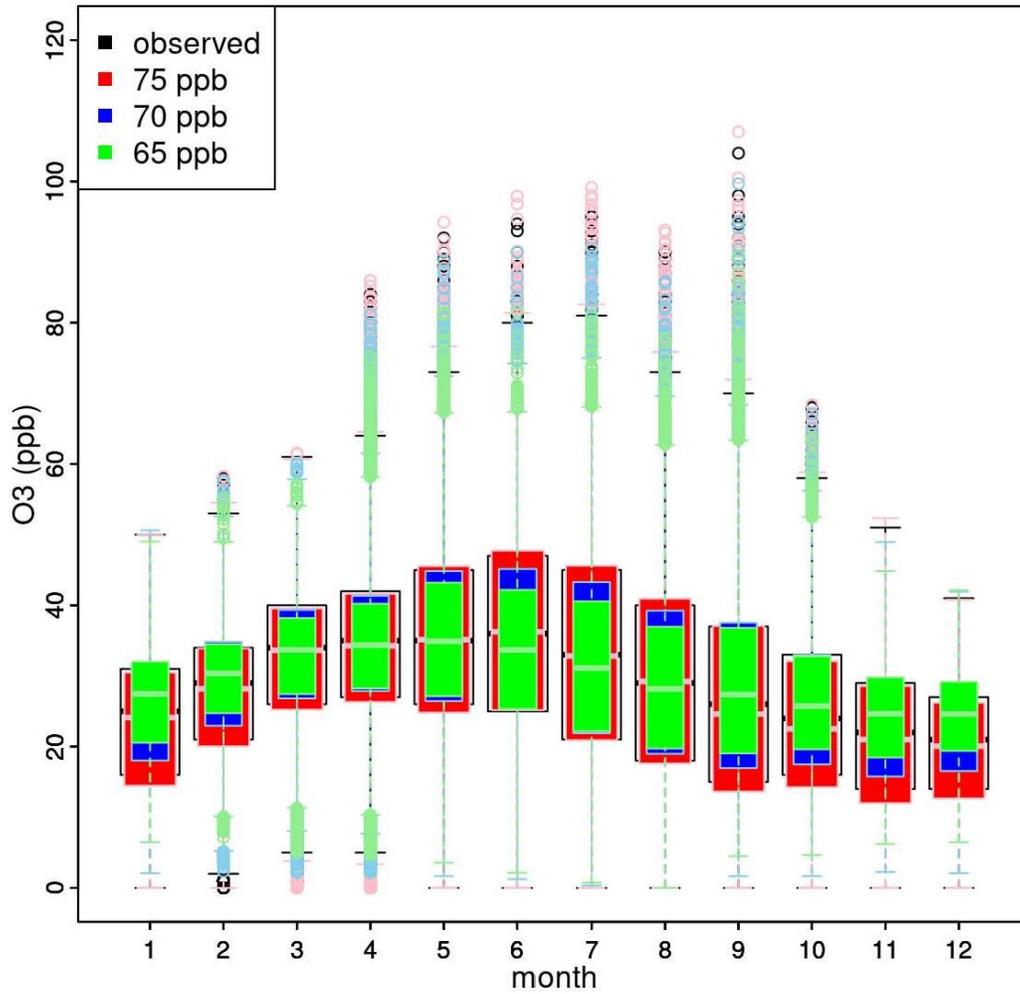


Figure 3C-78. Monthly distribution of hourly O₃ concentrations at monitoring sites in Detroit.

Figure 3-28. Figure 3C-78 from EPA (2023).

Philadelphia sites: 2015-2017

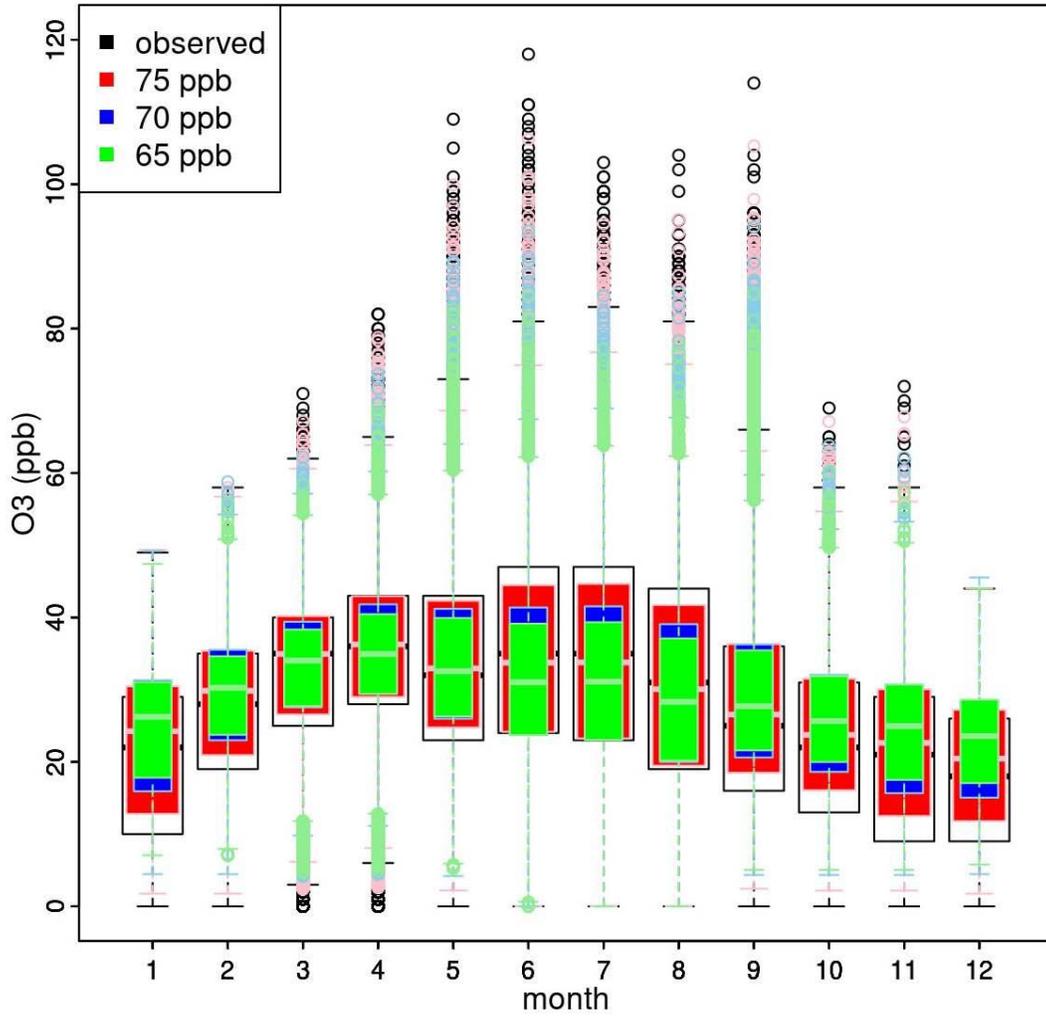


Figure 3C-79. Monthly distribution of hourly O₃ concentrations at monitoring sites in Philadelphia.

Figure 3-29. Figure 3C-79 from EPA (2023).

Phoenix sites: 2015-2017

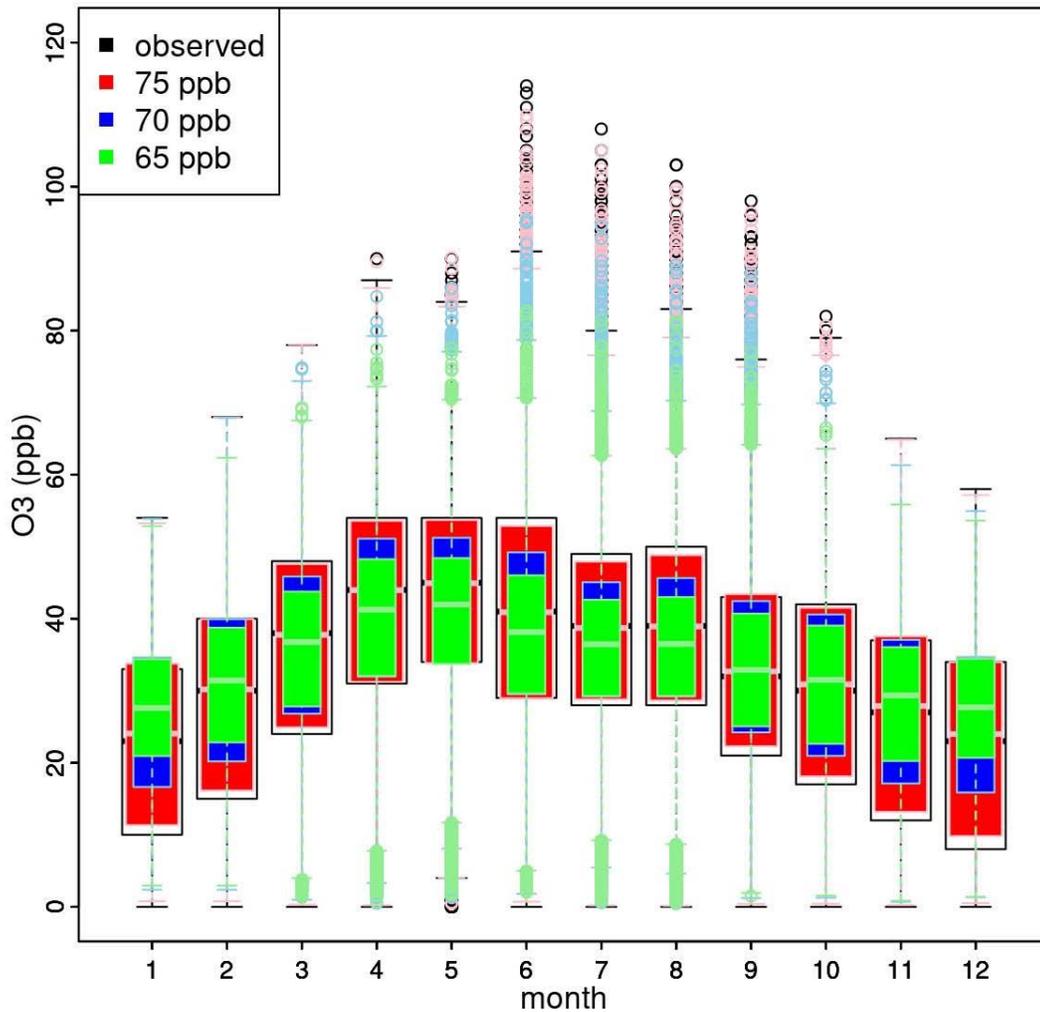


Figure 3C-80. Monthly distribution of hourly O₃ concentrations at monitoring sites in Phoenix.

Figure 3-30. Figure 3C-80 from EPA (2023).

Sacramento sites: 2015-2017

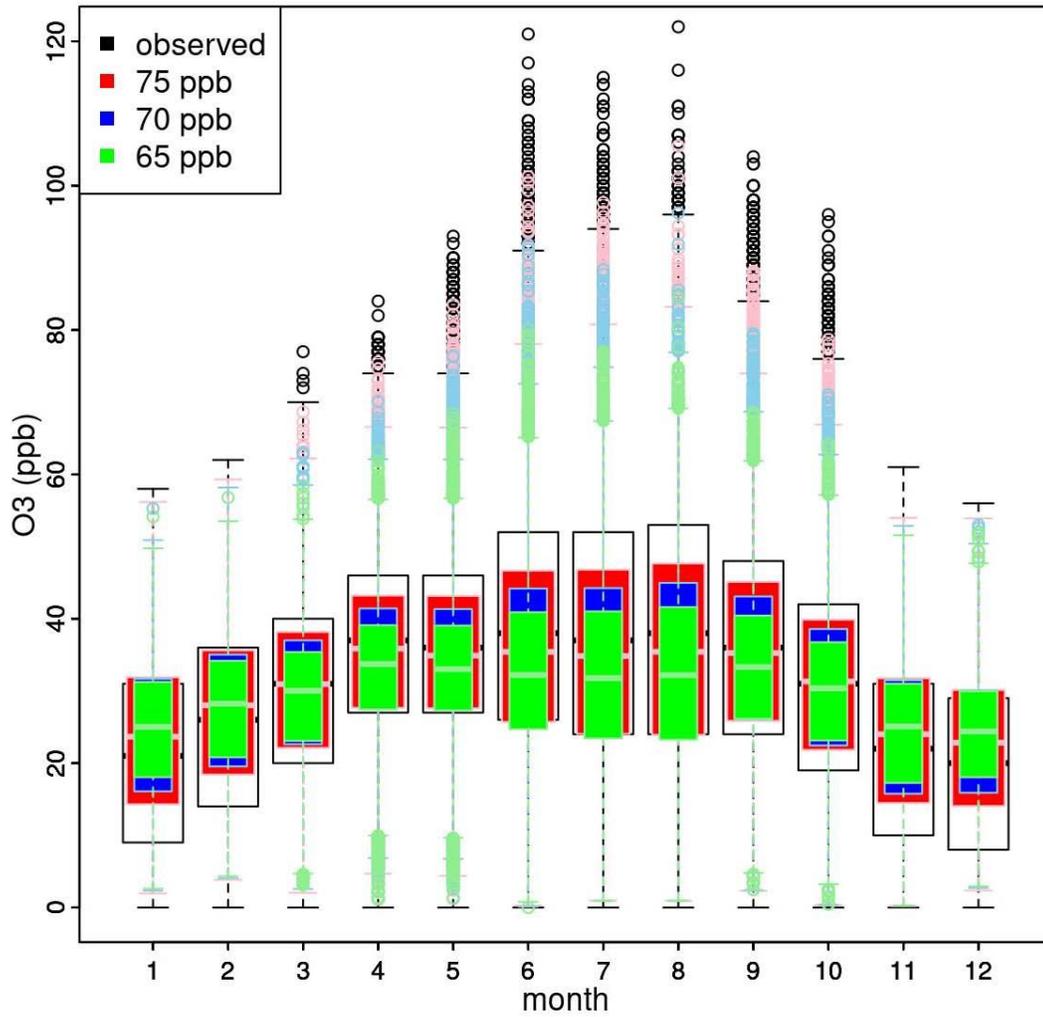


Figure 3C-81. Monthly distribution of hourly O₃ concentrations at monitoring sites in Sacramento.

Figure 3-31. Figure 3C-81 from EPA (2023).

SaintLouis sites: 2015-2017

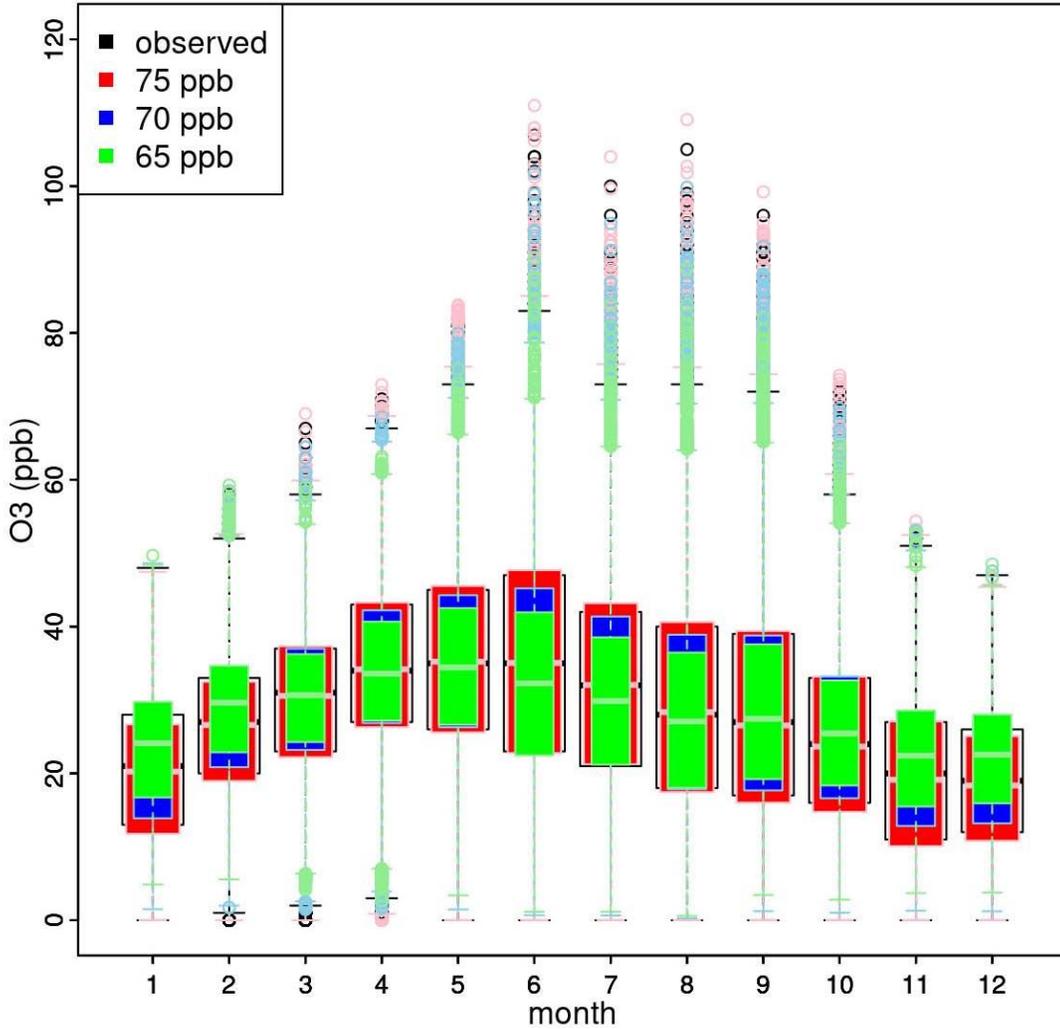


Figure 3C-82. Monthly distribution of hourly O₃ concentrations at monitoring sites in St. Louis.

Figure 3-32. Figure 3C-82 from EPA (2023).

Using its models, for the monthly plots for the 8 cities, the EPA notes in the second draft PA (EPA, 2023, page 3C-92) the following:

Similar to the diurnal plots, the seasonal distributions become flatter when adjusted to meet the 70 ppb and 65 ppb scenarios, especially on the highest O₃ days. This is due to more O₃ decreases during summer months and more O₃ increases in winter months. The O₃ increases in the winter are consistent with the understanding that solar insolation rates are lower in the winter reducing total photochemical activity and shifting the net

effect of NO_x emissions on O₃ which can both create O₃ through photochemical pathways and destroy O₃ through titration. In addition, the decreases on the highest O₃ days and increases on the lowest O₃ days show a visible compression of the O₃ distribution in these plots, similar to what was seen in the diurnal plots.

In the second draft PA (EPA, 2023), the modeling results showed changes for midrange O₃ days for a pattern of shifting higher mid-range O₃ from the summer months to earlier in the year. While in most cities, the highest interquartile O₃ concentrations in the recent conditions occur in the summer months, in many areas the highest interquartile O₃ concentrations shift to spring months (April-May) for the adjustment scenarios. This pattern can be seen above in Detroit (Fig. 3-28), Philadelphia (Fig. 3-29), Phoenix (Fig. 3-30), and St. Louis (Fig. 3-32). In the previous analysis in the EPA Risk Exposure Assessment (EPA, 2014b), a similar pattern was observed in Atlanta, Baltimore, Boston, Denver, Los Angeles, New York, Philadelphia, Sacramento, and Washington D.C. This pattern is consistent with a greater contribution from non-U.S. anthropogenic sources at lower projected standard levels than under recent observed conditions. Two of these non-U.S. anthropogenic sources, stratospheric intrusion and international transport, have been shown to peak during the spring months as discussed in the ISA (EPA, 2020a, page IS-15).

While the modeling results show the pattern of the shifting of when the higher mid-range concentrations occur from the summer months toward the spring months as emissions are reduced, it is important to explore if modeling predictions are confirmed using actual hourly concentration data from O₃ monitoring sites. Actual data do show that the highest O₃ exposures occur at some sites across the U.S. during the springtime (March to mid-June). Using hourly average O₃ data from 57 National Park Service Parks, the EPA in the 2014 PA (EPA, 2014c) (Welfare Appendix, page 7A-12) provided the highest 3-month W126 values and the timeframe corresponding to those W126 exposures for the Parks with O₃ monitors for the period 2006-2010. Table 7A-2 is provided in the pages below. Note that several of the O₃ monitors in the Parks experienced their highest 3-month W126 exposures during the spring months (defined as March, April, May or April, May, June) period. While the months of April, May, and June are not entirely a spring period (the first half of June is still spring), the time of year when the frequency of stratospheric intrusions (i.e., a natural process) to the surface is greatest at many sites during the March – June window. Lefohn et al. (2011, 2012) reported that stratosphere-to-troposphere transport to the surface (STT-S) frequently coincides with “enhanced” surface O₃ concentrations (≥ 50 ppb) at both high- and low-elevation monitoring sites across the U.S. during specific months, especially the spring. Dr. Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich in Switzerland, as a part of our research effort, applied a Lagrangian method, based on the approach introduced by Wernli and Bourqui (2002), to identify stratosphere-to-troposphere transport (STT) events down to the surface (i.e., STT-S events). The trajectory model introduced by Wernli and Davies (1997) was used to identify days of high probability for STT trajectories to enhance surface O₃ at specific monitoring sites. It is important to note that the analysis of stratospheric intrusions (SI) and calculation of the SI parameter as described in Lefohn et al. (2011) captures the frequency and vertical penetration of the intrusions; it does not provide information about the O₃ concentration within the intrusion. The O₃ concentration in stratospheric intrusions down to the lower troposphere was expected to be

highly variable due to concentration differences in the stratospheric origin and in chemical and mixing processes during the descent. As noted in Lefohn et al. (2011), this variability can strongly affect any statistical associations between the enhanced hourly average concentrations ≥ 50 ppb used in their analysis and the number of stratospheric intrusions. Because of this variability, Lefohn et al. (2011) selected the coincidence table approach, which summarized the frequency of daily intrusions and the daily maximum hourly average O₃ concentrations, and then applied appropriate statistical tests. For the high-elevation sites in the western and eastern U.S., the STT-S coincidences occurred most frequently during spring. However, Lefohn et al. (2012) noted that coincidences between STT-S and enhanced O₃ concentrations occurred at times during the summer, fall, and late winter.

Great Smoky Mountains National Park (GRSM) is the most visited National Park in the United States. It is a relatively small park (~210,433 ha), but topographically complex, with an elevational range of 1757 m. The Park is in parts of North Carolina and Tennessee. The name "Smoky" comes from the natural fog that often hangs over the range and presents as large smoke plumes from a distance. This fog is caused by the vegetation emitting volatile organic carbon chemicals that have a high vapor pressure and easily form vapors at normal temperature and pressure (Naranjo, 2011). The Park has historically been subject to elevated levels of pollutants, including SO₂, O₃, and NO_x. Neufeld et al. (2019) analyzed O₃ trends from 1989 to 2016 for six monitoring sites in and adjacent to GRSM and ranging in elevation from 564m to 2030m. The W126 exposures increased between the years 1989–~2002 and substantially decreased afterwards. Similar to the pattern described in the modeling results in the second draft PA (EPA, 2023), as emissions were reduced, at most of the six sites analyzed by Neufeld et al. (2019), the maximum 3-month W126 exposures shifted from mid-summer to the April–June period. Neufeld et al. (2019) noted in their paper that for the sites analyzed in their analysis, the shift in seasons following reduction of the peak concentrations may reflect a greater percentage of the remaining higher O₃ exposures being associated with background processes, such as stratospheric intrusions. Decreases in W126 exposures were correlated with lowered NO_x emissions from regional TVA power plants.

Besides the National Parks, a review of the data in EPA's AQS database indicates that there are many O₃ monitoring sites at both high and low elevations across the U.S. that exhibit highest exposures during the spring months. There are sites where maximum concentrations have shifted from summer to spring months, which confirms the pattern predictions of the models. There are also sites where shifts may not have occurred because the maximum concentrations continue in most years to occur during the spring months. At sites influenced by STT-S, there is a tendency for the highest O₃ exposures to occur during the spring months, but some sites may experience STT-S contributions throughout the year (Lefohn et al., 2011, 2012, 2014). Figs. 3-22 and 3-23, shown previously, illustrate that while the highest O₃ exposures at the high-elevation Yellowstone National Park site may occur during the spring months, STT-S enhanced O₃ levels exist throughout the entire year.

Table 7A-2 Ozone Exposure in 57 O₃ Monitors Located in Parks*

Monitor site ID	Park Name	W126					3-Month Timeframe for W126				
		2006	2007	2008	2009	2010	2006	2007	2008	2009	2010
230090102	Acadia National Park	10.59	7.89	7.64	7.02	5.24	MJJ	AMJ	MJJ	MAM	MAM
230090103	Acadia National Park	6.37	6.41	4.72	5.21	4.13	MJJ	AMJ	MJJ	MAM	MAM
311651001	Agate Fossil Beds National Monument	--	8.27	12.76	5.85	--	--	JAS	MJJ	JJA	--
460710001	Badlands National Park	--	--	2.23	2.54	3.85	--	--	JAS	AMJ	JJA
460711001	Badlands National Park	16.74	8.01	--	--	--	JJA	JJA	--	--	--
480430101	Big Bend National Park	11.62	10.60	10.55	8.62	8.47	AMJ	MAM	MAM	MAM	MAM
370110002	Blue Ridge Parkway	9.88	11.46	8.81	4.71	8.19	AMJ	AMJ	AMJ	AMJ	AMJ
490370101	Canyonlands National Park	18.06	16.93	17.06	12.23	13.24	MJJ	MJJ	AMJ	MAM	AMJ
250010002	Cape Cod National Seashore	13.47	13.16	12.89	5.25	7.03	MJJ	MJJ	MJJ	AMJ	MJJ
350153001	Carlsbad Caverns National Park	--	8.65	17.50	11.37	7.09	--	AMJ	AMJ	MJJ	AMJ
160310001	City of Rocks National Reserve	--	--	--	--	6.02	--	--	--	--	JJA
80771001	Colorado National Monument	--	11.61	15.04	4.13	8.75	--	JJA	MJJ	JAS	AMJ
450790021	Congaree National Park	12.31	10.78	9.45	3.97	6.32	MAM	MAM	MAM	FMA	MAM
450210002	Cowpens National Battlefield	14.30	7.87	16.05	3.24	8.81	MJJ	AMJ	JJA	FMA	MAM
160230101	Craters of the Moon National Monument	--	10.17	10.88	5.68	7.82	--	JJA	MJJ	MAM	JAS
210131002	Cumberland Gap National Historical Park	--	18.36	10.12	3.58	7.31	--	MJJ	MJJ	MJJ	MJJ
60270101	Death Valley National Park	29.18	32.55	25.57	15.30	10.61	MJJ	MJJ	MJJ	JJA	JAS
560111013	Devil's Tower National Monument	--	--	7.09	5.42	5.44	--	--	JAS	JAS	JJA
490471002	Dinosaur National Monument	--	10.33	13.34	8.39	13.80	--	MJJ	MJJ	MJJ	MJJ
300298001	Glacier National Park	2.90	2.29	3.98	3.53	2.44	JJA	MAM	MAM	AMJ	AMJ
300351001	Glacier National Park	--	--	--	4.91	3.93	--	--	--	MJJ	MJJ
40058001	Grand Canyon National Park	21.66	18.68	17.02	10.10	14.95	MJJ	AMJ	AMJ	JJA	AMJ
320330101	Great Basin National Park	15.54	15.79	16.94	10.19	11.44	JJA	MJJ	MJJ	AMJ	AMJ
370870036	Great Smoky Mountains National Park	11.46	13.35	11.50	4.59	7.89	AMJ	AMJ	AMJ	AMJ	AMJ
470090102	Great Smoky Mountains National Park	12.97	12.69	10.44	5.31	10.27	AMJ	MAM	AMJ	MAM	MAM
471550101	Great Smoky Mountains National Park	18.87	20.66	14.15	9.03	15.16	AMJ	AMJ	MJJ	MAM	MAM
471550102	Great Smoky Mountains National Park	19.59	23.51	16.23	7.32	11.94	MJJ	JJA	MJJ	MJJ	ASO

7A-12

Source: EPA (2014c).

Monitor site ID	Park Name	W126					3-Month Timeframe for W126				
		2006	2007	2008	2009	2010	2006	2007	2008	2009	2010
180890022	Indiana Dunes National Lakeshore	8.79	12.21	3.66	2.42	3.91	JJA	AMJ	JAS	MJJ	JJA
60650008	Joshua Tree National Park	24.36	19.97	27.43	19.66	23.39	AMJ	AMJ	MJJ	AMJ	AMJ
60651004	Joshua Tree National Park	--	26.37	30.05	18.81	20.47	--	MJJ	AMJ	JJA	JJA
60719002	Joshua Tree National Park	55.48	52.46	50.99	39.93	43.92	MJJ	MJJ	JJA	JJA	JJA
60893003	Lassen Volcanic National Park	18.97	15.10	18.98	7.64	9.63	JAS	JJA	MJJ	JJA	JAS
80830101	Mesa Verde National Park	23.44	17.57	13.41	15.05	11.94	MJJ	MJJ	AMJ	JJA	AMJ
60711001	Mojave National Preserve	--	28.50	38.92	19.91	19.39	--	MJJ	MJJ	JAS	JJA
530530012	Mount Rainier Wilderness	3.19	3.30	1.18	2.20	1.86	MAM	MAM	JAS	FMA	MAM
530090016	Olympic National Park	--	--	--	--	0.52	--	--	--	--	JAS
530091004	Olympic National Park	--	0.28	0.93	--	--	--	JAS	JAS	--	--
482731001	Padre Island National Seashore	--	8.19	3.66	--	--	--	AMJ	AMJ	--	--
40170119	Petrified Forest National Park	19.16	16.60	19.40	9.04	12.71	AMJ	AMJ	AMJ	AMJ	AMJ
60690003	Pinnacles National Monument	17.14	14.85	19.78	11.41	9.79	JAS	AMJ	MJJ	JAS	JAS
40190021	Saguaro National Park	19.57	17.06	20.13	11.01	15.31	MJJ	MJJ	AMJ	MAM	AMJ
360910004	Saratoga National Historical Park	6.68	10.38	9.26	5.40	5.98	JJA	MJJ	AMJ	MAM	MJJ
311570005	Scotts Bluff National Monument	--	--	--	--	6.20	--	--	--	--	JJA
61070006	Sequoia-Kings Canyon National Park	50.09	53.38	57.24	29.13	26.93	JJA	JJA	JJA	JAS	JAS
61070009	Sequoia-Kings Canyon National Park	66.07	62.88	56.91	55.51	53.79	JAS	JJA	MJJ	JAS	JAS
511130003	Shenandoah National Park	16.43	14.40	12.07	7.63	10.84	AMJ	AMJ	AMJ	MAM	JAS
380070002	Theodore Roosevelt National Park	7.71	5.54	5.55	3.95	4.19	JAS	JJA	AMJ	AMJ	AMJ
380530002	Theodore Roosevelt National Park	9.45	6.29	6.31	4.22	5.17	JJA	JJA	MJJ	AMJ	MAM
40070010	Tonto National Monument	26.39	23.24	25.40	13.67	16.90	MJJ	MJJ	AMJ	AMJ	AMJ
271370034	Voyageurs National Park	5.33	5.19	3.86	4.94	7.66	AMJ	AMJ	MAM	MAM	MAM
460330132	Wind Cave National Park	20.52	12.20	5.92	5.75	5.61	JJA	JJA	JJA	JJA	JAS
560391011	Yellowstone National Park	12.98	9.96	8.84	7.63	11.54	AMJ	AMJ	MAM	MAM	AMJ
60430003	Yosemite National Park	33.78	29.68	42.51	25.70	27.34	JJA	MJJ	JJA	JAS	JAS
60431002	Yosemite National Park	--	12.60	10.03	--	--	--	AMJ	MJJ	--	--
60431003	Yosemite National Park	--	11.61	--	--	--	--	JAS	--	--	--
60431004	Yosemite National Park	--	6.95	15.52	6.58	9.43	--	MJJ	JJA	JAS	JAS
60431005	Yosemite National Park	--	--	27.83	5.18	14.28	--	--	JAS	JAS	JAS

*Nine parks have more than 1 monitor

Source: EPA (2014c).

3.2 Background Ozone

3.2.1 Why is the Quantification and Spatial Distribution of Background O₃ Important?

The EPA (2015) recognized that periodically, in some locations in the U.S., sources other than domestic manmade emissions of O₃ precursors can contribute appreciably to monitored O₃ concentrations. EPA (2015) clarified the specific definition of background O₃ that the Agency wished to use and the sources and processes that led to background O₃ across the U.S.

The EPA revised the primary O₃ NAAQS to a level of 0.070ppm (70ppb) on October 1, 2015. The Administrator selected the final level of the NAAQS from the upper end of the range of proposed levels without considering the issue of proximity to background O₃ concentrations in some areas. However, the EPA considered the extent and importance of background O₃ throughout the NAAQS review process. EPA (2015) noted that Agency considered background O₃ to be any O₃ formed from sources or processes other than U.S. manmade emissions of nitrogen oxides (NO_x), volatile organic compounds (VOC), methane (CH₄), and carbon monoxide (CO). This definition of background was specifically referred to as U.S. background (USB). USB does not include intrastate or interstate transport of manmade O₃, which can also influence O₃ concentrations in downwind areas, but which can be addressed by certain provisions of the CAA. EPA (2015) noted that

Away from the earth's surface, O₃ can have an atmospheric lifetime on the order of weeks. As a result, background O₃, and to a lesser extent background O₃ precursors, can be transported long distances in the upper troposphere and be available to mix down to the surface when conditions are favorable. One of the largest natural sources of O₃ originates from production of O₃ in the stratosphere through interactions between ultraviolet light and molecular oxygen. O₃ exists in large quantities in the stratosphere and natural atmospheric exchange processes can transport stratospheric air into the troposphere. During certain meteorological conditions, discrete plumes of stratospheric air can be displaced far into the troposphere and impact ground-level O₃ concentrations. These events are called stratospheric intrusions and can result in relatively high USB levels of O₃ at the surface, especially at higher-elevation sites. Other natural sources of O₃ precursor emissions include wildfires, lightning, and vegetation. Biogenic emissions of methane, which can be chemically converted to O₃ over relatively long time (sic) scales, can also contribute to USB O₃ levels. Finally, manmade precursor emissions from other countries can contribute to the global burden of O₃ in the troposphere and to increased USB O₃ levels.

USB O₃ levels can vary considerably in space and time. When assessing USB O₃ concentrations, it is important to clarify the averaging time being considered. From a broad characterization perspective, it can be useful to identify annual or seasonal mean concentrations by location. However, from an air quality management perspective, it is more important to consider background concentrations on specific high O₃ days when concentrations may approach or exceed the NAAQS.

In the second draft PA (EPA, 2023, page 2-28), the authors note:

In this reconsideration, as in past reviews, the EPA generally characterizes O₃ concentrations that would exist in the absence of U.S. anthropogenic emissions as U.S. background (USB). An alternative phrasing for USB is the O₃ concentrations created collectively from global natural sources and from anthropogenic sources existing outside of the U.S. Such a definition helps distinguish the O₃ that can be controlled by precursor emissions reductions within the U.S. from O₃ originating from global natural and foreign precursor sources that cannot be controlled by U.S. regulations (ISA, section 1.2.2).

EPA (2015) noted that

Several modeling studies have attempted to estimate background O₃ levels by assessing the remaining O₃ in a model simulation in which certain emissions were removed. This basic approach, which is often referred to as “zero-out” modeling (i.e., U.S. manmade emissions are removed) or “emissions perturbation” modeling, has been used to estimate USB O₃ levels. Another modeling technique, referred to as “source apportionment” modeling, can also be used to estimate the sources that contribute to modeled O₃ concentrations. This approach estimates the contribution of certain source categories (e.g., natural sources, non-U.S. manmade sources) to modeled O₃ at each model grid cell on an hourly basis.

A great deal of the focus by EPA in its background O₃ modeling effort (pages 1-52 and 1-53 of the ISA (EPA, 2020a)) appears to be on the following:

- Applications of chemical transport models (CTMs) to estimate USB O₃ have found that USB concentrations are relatively constant with increasing total O₃ concentration, indicating that days with higher O₃ concentrations generally occur because of higher U.S. anthropogenic contributions (Dolwick et al., 2015).
- Based on these considerations, this section emphasizes USB on days with high O₃ concentration as the most relevant for discussing USB O₃, and wherever possible, the focus is on estimates of USB under these conditions because they are most relevant for evaluating the potential for a role of USB O₃ in contributing to the highest O₃ concentrations. Discussion of seasonal and monthly means of hourly data are also included because longer averaging times are relevant to assessments of health and ecological effects.

Continuing to focus on the contribution of USB on high O₃ days, page 1-66 in the ISA (EPA, 2020a) states:

- There is consistent evidence across several studies using different background measurement approaches that USB or other background concentration estimates on most days with high O₃ concentrations have been generally predicted to be

similar to or smaller than seasonal mean USB O₃ estimates in the eastern U.S. and in urban and low-elevation areas of the western U.S., and an inverse relationship between relative USB contribution and total O₃ concentration in these areas has been consistently predicted. This contrasts with high-elevation locations in the western U.S., where USB and NAB have been consistently predicted to increase with total O₃ concentration.

Further, in the second draft PA (2023, pages 2-66 and 2-67), the Agency's focus on USB on days when high O₃ concentrations occur continues:

- The USA contributions that drive predicted MDA8 total O₃ concentrations above 70 ppb are predicted to typically peak in summer. In this typical case, the predicted USB is overwhelmingly from Natural sources. The most notable exception to the typical case is reflected by predictions for an area near the Mexico border where the modeling indicates that a combination of Natural and Canada/Mexico contributions can lead to predicted MDA8 USB concentrations 60-80 ppb, on specific days, which is consistent with the O₃ PA prepared for the 2015 review (2014 PA, Section 2.4).⁴⁴
- Predicted international contributions, in most places, are lowest during the season with the most frequent occurrence of MDA8 concentrations above 70 ppb. Except for the near border areas, the International contribution requires long-distance transport that is most efficient in Spring.
- Days for which MDA8 total O₃ concentrations are predicted to be above 70 ppb tend to have a substantially higher model-predicted USA (anthropogenic) contribution than other days in both the West and the East.

From an air quality management perspective, the EPA's focus on the contribution of background O₃ during periods when elevated O₃ levels occur appears to be reasonable. Hogrefe et al. (2020) note that

Viewed from an air quality management perspective, the magnitude of O₃ formed from locally controllable anthropogenic precursor emissions relative to background ozone (BG O₃) levels—that is, O₃ formed from non-local anthropogenic precursor emission sources and local and non-local natural sources, regardless of where the O₃ formation occurs—has generally decreased over time. This increases emphasis on representing the processes controlling BG O₃ and characterizing its temporal and spatial fluctuations, especially for O₃ exceedance events. The latter is particularly important because although the relative magnitude of typical average BG O₃ concentrations with respect to the NAAQS has increased, BG O₃ contributions to specific exceedance events have not increased in some cases.

Langford et al. (2022) and Li et al. (2020) have discussed that increasing background O₃ sources, together with year-to-year variability in stratospheric influence, will leave little margin for O₃

produced from local and regional emissions, posing challenges to achieving a potentially tightened O₃ NAAQS in the southwestern U.S.

While the EPA appears to be mainly focused on how much of current O₃ exposures can be attributed to sources other than U.S. anthropogenic sources on days when ambient levels exceed the NAAQS, there are other considerations in the rulemaking process that are of equal importance to quantifying background O₃. As discussed earlier, EPA states on page 1-53 of the ISA (EPA, 2020a) that background seasonal and monthly means of hourly data are also included because longer averaging times are relevant for assessments of health and ecological effects. However, this statement is inaccurate. In many cases, assessment of human health and ecological effects are not based on longer averaging times.

The second draft PA (EPA, 2023, page 2-32) incorrectly defines the vegetation related W126 exposure index as follows:

³⁰ W126 is a daytime weighted average concentration where higher concentrations are given greater weight based on a sigmoidal curve (see Chapter 4).

The W126 exposure index is not a daytime weighted average concentration. For calculating the W126 metric, hourly average concentrations are weighted and accumulated over a specific period for assessing vegetation effects. The second draft PA (EPA, 2023, page 4-2) correctly defines the W126 exposure metric in footnote one as follows:

¹ The W126 index is a cumulative seasonal metric described as the sigmoidally weighted sum of all hourly O₃ concentrations during a specified daily and seasonal time window, with each hourly O₃ concentration given a weight that increases from zero to one with increasing concentration (80 FR 65373-74, October 26, 2015). The units for W126 index values are ppm-hours (ppm-hrs). More detail is provided in section 4.3.3.1.1 below.

For assessing vegetation risk, hourly average background O₃ concentrations contribute to the observed concentrations and therefore, contribute to the cumulative risk. For some human health risk assessments, daily maximum 8-h average concentrations are used in a time series. Daily maximum 8-h average concentrations contain background O₃ concentrations, which contribute to the estimated human health risk. As pointed out by Langford et al. (2022) and Li et al. (2020), background O₃ sources, including the year-to-year variability in stratospheric influence, play an important role in influencing O₃ levels in the southwestern U.S.

Background O₃ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total O₃ levels and the lower and mid-level concentrations influence mortality and morbidity risk estimates. It is important to quantify the importance of background O₃ in the low- and mid-range concentrations. Fig. 3-33 (reproduced from Fig. 9-8 on Page 9-32 from the 2014 HREA (EPA, 2014b)) illustrates the percent reduction in exposures and risks after just meeting alternative standards relative to just meeting the 2008 NAAQS of 75 ppb. In this plot, each row represents one of the key analytical results and each column gives the results for 2007 and 2009 for each urban study area. The scales are the same

between analyses, and as such, it is informative to examine both the overall patterns of change between alternative standards, and the absolute value of the percent reductions in risk metrics between analyses. The top row is the Exposure > 60 ppb; the second row is the Lung Function Risk (dFEV₁ > 10%); the third row is Mortality; the fourth row is Hospital Admissions. The risks associated with mortality and hospital admissions are much less than the risks associated with Exposure > 60 ppb and Lung Function Risk (dFEV₁ > 10%). This is because mortality and hospital admission risk metrics are based on non-threshold, approximately linear C-R functions, and therefore are sensitive to changes in O₃ along the full range of O₃ concentrations (page 9-30 of the 2014 Health REA (EPA, 2014b)). As explained by the Agency (EPA, 2014b), because O₃ in the lower concentration range may shift upward as the result of NO_x emission reductions, this can lead to increases in risk on some days, which can lead to a net increase or decrease in risk over the entire year, depending on whether the days with increased risk exceed days with decreased risk (generally due to a preponderance of days with lower O₃ concentrations).

Simon et al. (2016) note that some measures of risk, such as lung function responses derived from controlled human exposure studies, are most sensitive to reductions in high O₃ concentrations, and thus, risk metrics associated with these concentrations will show improvement with decreases in NO_x. Simon et al. (2016) point out that in contrast, the epidemiology-based risks that are currently equally responsive to changes at low and high concentrations are more ambiguous in response to NO_x decreases and ultimately depend on the direction and magnitude of the O₃ change in places with the highest population densities. Simon et al. (2016) suggest that because increases in O₃ occur at low O₃ concentrations at times when NO_x emission reductions occur, it is increasingly important for health studies to evaluate whether the shape of the concentration–response relationship changes at these lower O₃ levels.

Fig. 3-34 illustrates the percent of short-term mortality attributable to O₃ concentrations in the 25-55 ppb range for 2007. The data to create the figure were obtained from EPA (2014d) in Fig. 7-B1 on page 7-B3. In some cases, 90% or more of the accumulated risk is associated with the mid-range concentrations for cities across the U.S. The different colors represent the different standard scenarios considered in the 2014 Health REA (EPA, 2014b). Results shown from the 2014 Health REA (EPA, 2014b) were similar for all 12 cities used in the 2014 epidemiological risk analyses. The mid-range concentrations (25-55 ppb) as emissions are reduced is where background O₃ will predominate. Fig. 3-35 illustrates the contribution of background to ambient levels of O₃ for 2006 in Houston, Texas (see Lefohn et al., 2014 for further details). A large percentage of the observed concentrations in the 25-55 ppb range are associated with background O₃ at the measured levels in 2006. As indicated in earlier discussions in this document, as emissions are reduced, for some sites, a compression of the distribution of concentrations shifts the lower concentrations upward and the higher concentrations downward. Thus, as shown in Fig. 3-35, background O₃ concentrations will be expected to increase their domination of the cumulative mortality health risk estimates.

Background O₃ concentrations become more and more important in influencing ambient levels as emission reductions are implemented. It is recognized by the Courts that NAAQS O₃ levels are set to protect public health and welfare and that background O₃ is not a consideration in setting these levels. In the second draft PA (EPA, 2023, page 1-11), the EPA notes the following:

In the August 2019 decision, the court additionally addressed arguments regarding considerations of background O₃ concentrations, and socioeconomic and energy impacts. With regard to the former, the court rejected the argument that the EPA was required to take background O₃ concentrations into account when setting the NAAQS, holding that the text of CAA section 109(b) precluded this interpretation because it would mean that if background O₃ levels in any part of the country exceeded the level of O₃ that is requisite to protect public health, the EPA would be obliged to set the standard at the higher nonprotective level (id. at 622-23). Thus, the court concluded that the EPA did not act unlawfully or arbitrarily or capriciously in setting the 2015 NAAQS without regard for background O₃ (id. at 624).

However, for practical purposes, if the standard were to be set at a level that ambient O₃ levels consisted of almost all background O₃, then the setting of such a level under these circumstances would not be a standard, but a goal that would not necessarily be achievable. As noted in the second draft PA (EPA, 2023, page 2-52), when episodic natural events contribute to elevated O₃ concentrations documented in the air quality monitoring data to such an extent that they result in a regulatorily significant exceedance or violation of the NAAQS, the data can be addressed via the Exceptional Events Rule (40 CFR 50.14). Given the current level of the 8-h daily maximum concentration of 70 ppb for the O₃ NAAQS, the Exceptional Events Rule was applied by the states in several instances. For example, for the period 2017-2019, Arizona, California, Colorado, Idaho, Louisiana, Montana, Nevada, Utah, and Wyoming requested that specific hourly average O₃ concentrations reported for specific sites in their states be considered under the Exceptional Events Rule.

While background O₃ currently is not a consideration in the setting of the level of the O₃ standard, background O₃ is an important consideration for assessing human health effects risks. The risk assessments play an important role in the margin of safety determinations. Background O₃ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total ambient O₃ levels and potentially can influence human health risk assessments associated with margin of safety determinations. An adequate margin of safety is a policy choice left specifically to the Administrator's judgment. The greater the contribution of background O₃ to the human health risk assessment, the greater the uncertainty will be to the input into the margin of safety consideration. Thus, currently because of its importance in affecting the human health risk assessments used in the margin of safety determination, background O₃ is an important consideration to be quantified for the 8 cities highlighted in the Agency's modeling analyses (i.e., Atlanta, Boston, Dallas, Detroit, Philadelphia, Phoenix, Sacramento, and St. Louis) in the second draft PA (EPA, 2023). Because background O₃ estimates for the 8 cities were not provided in the second draft PA (EPA, 2023), no information is available at this time in the current O₃ NAAQS reconsideration rulemaking process for assessing the contribution of background O₃ to the human health effects risks. This information is important in margin of safety considerations for the setting of the NAAQS for O₃.

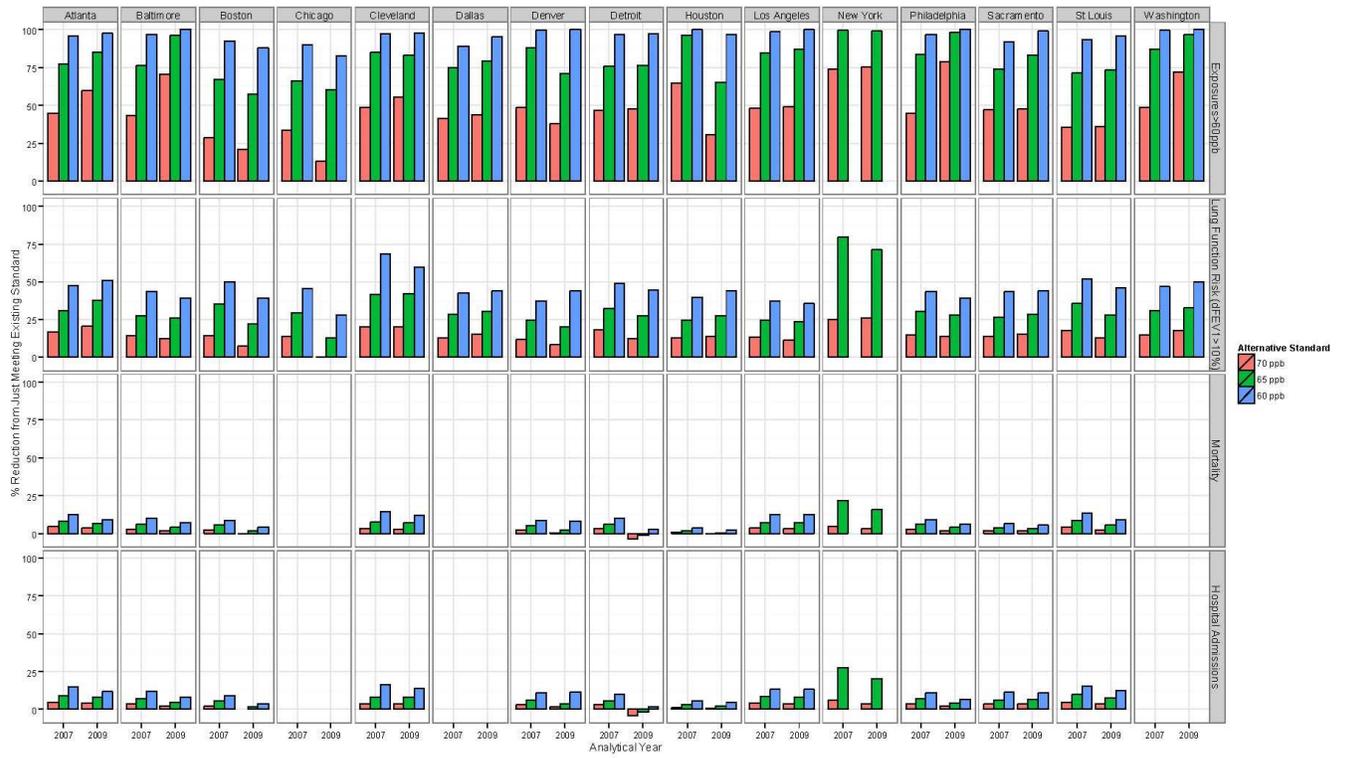


Figure 9-8. Comparison of the Percent Reduction in Key Risk Metrics for Alternative Standard Levels Relative to Just Meeting the Existing 75 ppb Standard.

Figure 3-33. Figure 9-8 from EPA (2014b).

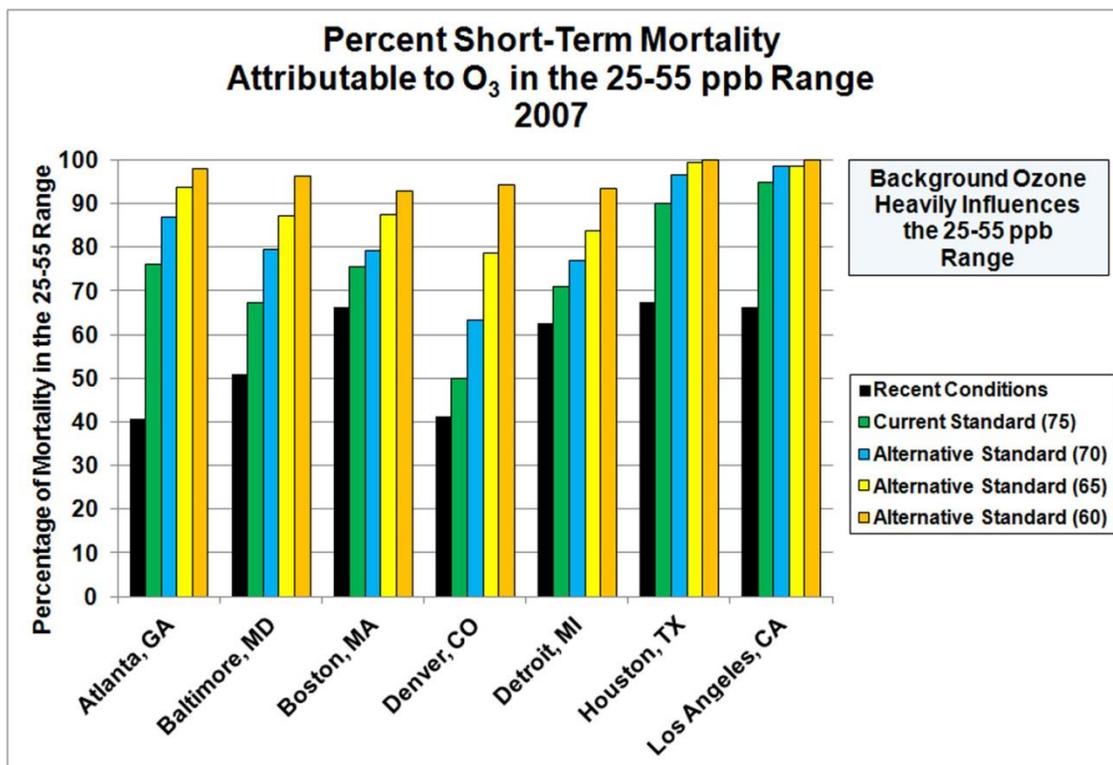


Figure 3-34. Percent short-term ozone-attributable mortality in the 25-55 ppb range for various exposure conditions for 2007 for 7 of 12 cities. (Source: Data from Fig. 7-B1 on page 7-B3 of EPA, 2014d).

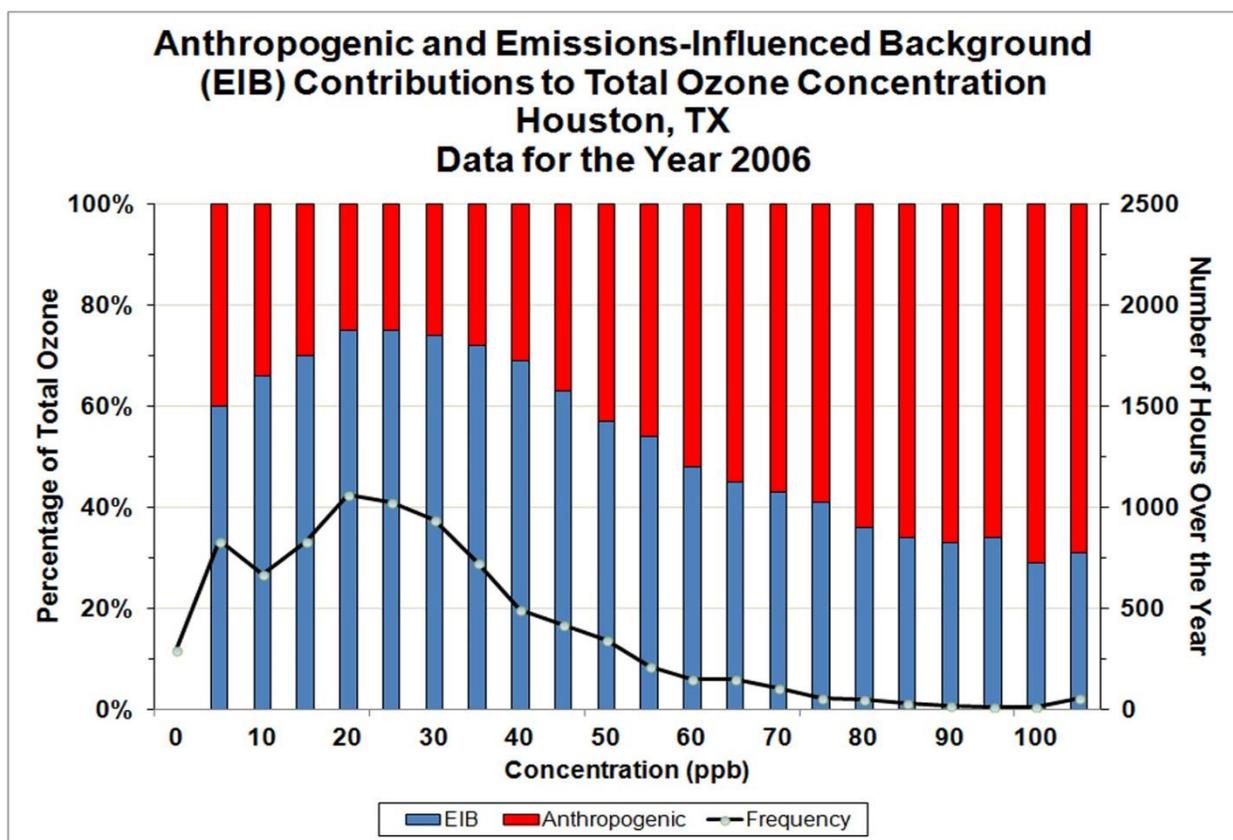


Figure 3-35. Binned (5 ppb) frequency distribution of observed hourly total O₃ (black curve; right axis) and average relative binned contributions of hourly maximum Emissions Influenced Background (EIB) and anthropogenic O₃ (bars; left axis) for Houston, TX (AQS ID 482010055) in 2006. Lefohn et al. (2014).

3.2.2 Background O₃ Can be Defined in Different Ways

Based on the discussion in Section 3.2.1, it appears that the EPA desires to answer the question “**How much of the current O₃ can be attributed to sources other than U.S. anthropogenic sources?** While the Agency is focused on how much of the current O₃ can be attributed to sources other than U.S. anthropogenic sources on days when ambient levels exceed the standard, as discussed previously, quantifying background O₃ is also important for assessing human health and ecological effects risks and how background O₃ potentially influences the margin of safety for the setting of the O₃ NAAQS.

While background O₃ cannot be measured directly, estimating it accurately by utilizing both empirical and modeling approaches is important. Under a variety of meteorological conditions, background O₃ can make a substantial contribution to levels that result in exceedances of Federal standards when (1) *episodic events* occur under relatively clean conditions (Zhang et al., 2011; Lin et al., 2012; Emery et al., 2012; Lefohn et al., 2014; Dolwick et al., 2015; Jaffe et al., 2018) and (2) local photochemical production combines with background levels that *enhance* the ambient concentrations (Lefohn et al., 2014; Dolwick et al.,

2015). Background O₃ is of interest because (1) at times background O₃ is associated with high concentrations experienced in the U.S. Intermountain West that affect attainability of O₃ air quality standards (Lefohn et al., 2001; Langford et al., 2009; McDonald-Buller et al., 2011; Lin et al., 2012; Dolwick et al., 2015; Li et al., 2020; Langford et al., 2022) and (2) background O₃ contributes on a continuous basis to observed concentrations that influence human health and vegetation risk estimates, whose values influence recommended levels for Federal O₃ standards (McDonald-Buller et al., 2011; EPA, 2014a,b). While considerable discussion in the U.S. has focused on background contributions to the human health Federal O₃ primary standard, elevated background concentrations associated with stratospheric intrusions can affect vegetation (Skelly, 2000; FLAG, 2010).

The term “background O₃” in the United States over the years has not been defined consistently (McDonald-Buller et al., 2011; Lefohn et al., 2014; EPA, 2014a). EPA (2006) defined North American background (NAB) O₃ (previously referred to as Policy-Relevant Background by the EPA) to include contributions from global anthropogenic and natural sources in the absence of North American (i.e., U.S., Canada, and Mexico) anthropogenic emissions. NAB O₃ is the range of concentrations that an air quality model estimates would exist in the absence of North American anthropogenic emissions. In 2013, the 2013 ISA (EPA, 2013) modified its definition of background by introducing the terminology U.S. background (USB) O₃ concentrations. The level of USB O₃ is defined to include contributions from global anthropogenic and natural sources in the absence of U.S. anthropogenic emissions. In other words, the USB O₃ concentration is defined as the O₃ concentration that would occur *if all U.S. anthropogenic O₃ precursor emissions were removed* (EPA, 2020a, Page ES-3). The difference associated with the hypothetical estimates using models of NAB O₃ and USB O₃ is small (EPA, 2013).

Recognizing that NAB O₃ and USB O₃ are estimated background concentrations that reflect hypothetical *zero* U.S. anthropogenic emissions, Lefohn et al. (2014) believed it was important to define a new metric that estimated background O₃ levels under *current anthropogenic emission* conditions. The authors, using source-apportionment based modeling, referred to these background estimates as Emissions-Influenced-Background (EIB) O₃. The authors defined Emission-influenced Background (EIB) O₃ to include contributions from natural sources throughout the globe and from anthropogenic sources outside of North America. EIB O₃ estimates the impact of background sources, even in situations in which local O₃ has been influenced by U.S. anthropogenic emissions. In August 2014 in its PA, the EPA (2014a) described estimates of source-apportionment U.S. Background (USB_{AB}). The EPA (2014a) defined source-apportionment based U.S. Background (USB_{AB}) in a similar manner as EIB O₃, except that USB_{AB} O₃ included anthropogenic sources from Canada and Mexico. Similar to EIB, USB_{AB} estimated the impact of background sources, even in situations in which local O₃ had been influenced by U.S. anthropogenic emissions (see Dolwick et al., 2015 for further discussion).

An important advantage in estimating either EIB O₃ or USB_{AB} background is that policymakers have an indication of (1) the relationship between current daily background levels and daily observed O₃ concentrations and (2) the level of O₃ concentration that may occur because of implementing emissions reductions strategies. For example, if EIB O₃ or USB_{AB} O₃

concentrations have a large relative contribution to observed O₃ concentrations at a specific location, one would anticipate that emissions reductions on a regional scale would not have much impact on the concentrations at that site.

In a previous PA (EPA, 2014a), research results based on natural background were presented. Natural background O₃ is defined as the O₃ concentrations that would occur if all anthropogenic emissions were removed worldwide. Processes that contribute to natural background O₃ include O₃ transport from the stratosphere and O₃ formed from precursor emissions originating from wildfires, lightning, natural methane sources, plants, and other natural VOC and NO_x emissions.

On page 1-6 of the ISA (EPA, 2020a), the authors mention **Baseline ozone** as an alternative metric for USB and NAB. Baseline O₃ has been defined as the measured O₃ concentration at rural or remote sites that have not been influenced by recent, local emissions (Jaffe et al., 2018). In contrast to USB, baseline O₃ is directly measured. Baseline measurements are typically from monitors in locations that are minimally influenced by local anthropogenic sources, and samples used as baseline measurements are limited to those monitored during meteorological conditions consistent with the relative absence of local contamination. Baseline O₃ can include the O₃ produced from U.S. emissions that circle the globe and may also include effects of same-state emissions. An example of the latter would be O₃ from U.S. emissions near the West Coast or Gulf Coast that is transported over the Pacific Ocean or Gulf of Mexico, respectively, and then transported back onshore. In some cases, sources that impact baseline O₃ may not similarly impact O₃ in populated locations. For instance, baseline O₃ measured on a mountaintop may include stratospheric influences not representative of contributions in nearby lower elevation locations.

The ISA (EPA, 2020a) points out (page 1-7) that there are several reasons why baseline O₃ measurements cannot be used as a proxy to estimate USB O₃ levels in urban areas. As previously described, baseline O₃ can include contributions from U.S. emissions. Additionally, baseline O₃ monitors can be very distant from urban sites, and O₃ measured at the baseline site can be destroyed through surface deposition or chemical reactions during transport from the baseline site to a downwind monitor. In addition, atmospheric conditions may not favor transport of baseline O₃ from the monitor location to populated areas at lower elevations. The ISA (EPA, 2020a) also points out that another reason why baseline O₃ measurements cannot be used as a proxy for USB O₃ levels (using the zero-out methodology) in urban areas is that meteorological conditions that favor mixing from the free troposphere to ground level have strong ventilation and are not conducive to photochemical O₃ episodes that produce the highest urban O₃ concentrations. However, as noted in the ISA (EPA, 2020a), stratospheric intrusion events are an exception. The ISA (EPA, 2020a) concludes (page 1-7) that while baseline O₃ measurements cannot be used directly to estimate USB (zero-out methodology) O₃, baseline O₃ data are useful for evaluating the CTMs that are used to provide model estimates of USB O₃.

In summary, the following terms have been used in the ISA (EPA, 2020a) to describe background O₃:

- USB is defined to include contributions from global anthropogenic and natural sources in the absence of U.S. anthropogenic emissions.
- NAB has been defined as the O₃ concentration that would occur in the U.S. in the absence of anthropogenic emissions in continental North America (EPA, 2013). NAB has also been referred to as policy-relevant background (PRB) in earlier publications (EPA, 2007).
- Emissions-influenced background (EIB) has been defined as another measure of background O₃ estimated from source apportionment modeling approaches while including chemical interactions with anthropogenic emissions (Lefohn et al., 2014).
- Source-apportionment U.S. Background (USB_{AB}) is the amount of O₃ formed from sources other than U.S. anthropogenic sources as estimated via an apportionment technique (Dolwick et al., 2015). USB_{AB} O₃ includes anthropogenic sources from Canada and Mexico.
- Natural background O₃ is defined as the O₃ concentrations that would occur if all anthropogenic emissions were removed worldwide. Processes that contribute to natural background O₃ include O₃ transport from the stratosphere and O₃ formed from precursor emissions originating from wildfires, lightning, natural methane sources, plants, and other natural VOC and NO_x emissions.
- Baseline O₃ has been defined as the measured O₃ concentration at rural or remote sites that have not been influenced by recent, local emissions (Jaffe et al., 2018). The ISA points out (page 1-7) that there are several reasons why baseline O₃ measurements cannot be used as a proxy to estimate USB O₃ levels in urban areas.

USB, as well as USB_{AB}, is a model construct that cannot be measured using ambient monitoring data. The ISA (EPA, 2020a) notes that this approach is consistent with the 2006 Ozone Air Quality Criteria Document (AQCD) (EPA, 2006a) and the 2013 Ozone ISA (EPA, 2013), which also used modeled estimates of background O₃. Reliance on atmospheric modeling for USB, as well as USB_{AB} concentrations estimates, continued in the 2013 Ozone ISA (EPA, 2013). In earlier assessments, O₃ estimates were based on measurements at monitoring sites with low concentrations that appeared to be isolated from anthropogenic sources (Altshuller and Lefohn, 1996; Trainer et al., 1993).

3.2.3 EPA's Preference for the Use of USB Rather than Other Definitions of Background

EPA has preferred to use the USB methodology for estimating background O₃. In the EPA White Paper (EPA, 2015) that is referred to in the second draft PA (EPA, 2023, page 2-66), the Agency noted that

For the purposes of this white paper and the continuing discussion of background O₃ issues in the NAAQS implementation context, the EPA considers background

O₃ to be any O₃ formed from sources or processes other than U.S. manmade emissions of nitrogen oxides (NO_x), volatile organic compounds (VOC), methane (CH₄), and carbon monoxide (CO). This definition of background is specifically referred to as U.S. background (USB).

The EPA in its White Paper (EPA, 2015) apparently made a unilateral decision to use USB rather than USB_{AB} in defining the term “background O₃.” EPA provided no clear rationale for this decision.

On page 1-5 of the ISA (EPA, 2020a), the EPA notes that modeling approaches for estimating background O₃ can be classified as either source-sensitivity or source-apportionment approaches. USB was originally estimated using source-sensitivity approaches (e.g., “zero-out” modeling). Apportionment-based USB (USB_{AB}) has been defined as the amount of O₃ formed from sources other than U.S. anthropogenic sources as estimated via an apportionment technique (Dolwick et al., 2015). In the 2014 Policy Assessment (EPA, 2014a), the Agency discussed and used both USB and USB_{AB}.

The ISA (EPA, 2020a) notes on page 1-56:

The zero-out approach is more suited for answering the question “what ozone levels would exist in the absence of all U.S. emissions?” while the source apportionment approach is more suited for answering the question “what amount of current ozone comes from background sources?” The difference between USB and USB_{AB} is small in remote areas most strongly affected by USB sources, but can be substantial in urban areas strongly affected by anthropogenic sources that influence both production and destruction of ozone (Dolwick et al., 2015).

Given that the EPA made the decision to use USB rather than USB_{AB}, it must be remembered that USB estimates will represent a quantity never to occur in the real atmosphere (EPA, 2014a). As noted in the 2014 PA (EPA, 2014a), sensitivity approaches (i.e., USB) can be unreliable for evaluating mass contributions to O₃ production because of nonlinearity in the chemistry.

The EPA in its 2014 PA (EPA, 2014a) noted that the strength of the source-apportionment approach (i.e., USB_{AB}) is that it provided a direct estimate of the amount of O₃ contributed by each source category, while avoiding artifacts caused by non-linearity in the chemistry, which is a potential with the zero emissions (i.e., zero-out) modeling used to estimate USB O₃ concentrations. Table 3-1 below (original labeled Table 2-1 on page 2-15 in EPA, 2014a) is reproduced from the EPA 2014 PA (EPA, 2014a). The table compares the two model methodologies used to characterize USB (i.e., zero-out) and USB_{AB} (apportionment based).

As noted in the ISA (EPA, 2020a) (Page IS-16):

Both approaches are essential and complementary for understanding and estimating USB ozone. The zero out approach is suited for determining what ozone levels would have existed in recent modeled years in the absence of all U.S.

emissions, while the source apportionment approach is suited for determining the fraction of current ozone originating from background sources in recent modeled years.

As noted above, a key point made in the ISA (EPA, 2020a, page 1-56) is that the difference between USB and USB_{AB} is small in remote areas most strongly affected by USB sources but *can be substantial in urban areas strongly affected by anthropogenic sources that influence both production and destruction of O₃* (Dolwick et al., 2015).

Estimation Methodology	Question addressed	Background Quantities	Strengths and Limitations
Zero-out	How much ozone would remain if controllable emissions were completely removed?	NB / NAB / USB	<p><u>Strength:</u> The approach is simple to implement and provides an estimate of the lowest O₃ levels that can be attained by eliminating all U.S. anthropogenic emissions.</p> <p><u>Limitation:</u> Estimates are based on a counterfactual, represents a quantity never to occur in real atmosphere. Additionally, sensitivity approaches can be unreliable for evaluating mass contributions to O₃ production because of non-linearity in the chemistry.</p>
Source Apportionment	How much of the current ozone can be attributed to sources other than U.S. anthropogenic sources?	Apportionment-based USB	<p><u>Strength:</u> Provides a direct estimate of the amount of O₃ contributed by each source category while avoiding artifacts caused by non-linearity in the chemistry.</p> <p><u>Limitation:</u> While this approach identifies important sources that contribute to O₃, it does not predict quantitatively how O₃ will respond to specific emissions reduction scenarios.</p>

Table 2-1 Comparison of the two model methodologies used to characterize background ozone levels.

Table 3-1. A comparison of the two model methodologies used to characterize background O₃ levels. Source: EPA (2014a).

The distinction between USB and USB_{AB} is important because apportionment techniques for estimating USB_{AB} are designed to realistically treat nonlinear and nonadditive interactions of USB and U.S. anthropogenic emissions that affect both production and destruction of O₃. In contrast, source-sensitivity modeling approaches originally used for estimating USB are not designed to address these interactions. As pointed out in the ISA (EPA, 2020a, page 1-5), USB and USB_{AB} are **not** the same quantity estimated with different approaches but are *estimates of conceptually different quantities. While USB is an estimate of O₃ concentrations that could be achieved if U.S. anthropogenic sources were eliminated, USB_{AB} is an estimate of how much O₃ can be attributed to background sources when those anthropogenic sources are present.*

As pointed out in Table 3-1 above (Table 2-1 in the 2014 PA), the USB approach is simple to implement and provides an estimate of the lowest O₃ levels that can be attained by eliminating all U.S. anthropogenic emissions. The table also notes that the USB estimates are based on a counterfactual, representing a quantity never to occur in the real atmosphere. As noted in the 2014 PA document (EPA, 2014a), sensitivity approaches can be unreliable for evaluating mass contributions to O₃ production because of nonlinearity in chemistry. The USB_{AB} approach provides a direct estimate of the amount of O₃ contributed by each source category, while avoiding artifacts caused by non-linearity in the chemistry. As noted in the 2014 PA (EPA, 2014a), while the approach identifies important sources that contribute to O₃, it does not predict quantitatively how O₃ will respond to specific emissions reductions scenarios. However, it is reasonable to assume that as emissions were reduced with the result that current O₃ levels at the high end were reduced and the lower levels shifted toward the mean due to reduction in NO_x levels, the USB_{AB} would *increase* to levels above the USB_{AB} estimates based on current O₃ levels. *In other words, background O₃ would make up a higher fraction of the levels observed in future O₃ levels achieved with emission reductions.* This is an important concept. By estimating USB_{AB} concentrations, one could obtain a lower estimate of background O₃ and quantify the percentage contribution of background O₃ to the total concentration. The percentage would provide important information to the EPA Administrator about the influence of background levels to those human health risk estimates that affect margin of safety determinations. As indicated earlier, a quantitative understanding of background O₃ is essential for air quality management (Li et al., 2020, Hogrefe et al., 2021, Langford et al., 2022). This is especially true given the lowering of the NAAQS O₃ levels and the associated increasing relative importance of background O₃ as domestic precursor emissions decrease. Using the information provided in Table 3-1 (Table 2-1 in the 2014 PA), it appears that USB_{AB} might have been a better modeling methodology to apply rather than USB to answer the question “How much of the current O₃ can be attributed to sources other than U.S. anthropogenic sources?” The USB_{AB} estimate would provide a lower estimate of background O₃ in most cases than that provided from USB estimates.

As noted in the second draft PA (EPA, 2023, page 2-38), the zero-out approach provides simplicity for interpretation and consistency purposes. However, the use of background O₃ estimates associated with the USB methodology (i.e., the zero-out modeling and its non-linearity chemistry problems) may provide less than optimum estimates when compared with those estimates derived from the USB_{AB} methodology. As noted in the second draft PA (EPA, 2023,

page 2-38), in urban areas, the zero-out approach will estimate higher natural and USB contributions than total O₃ when NO_x titration is present. **The second draft PA (EPA, 2023, page 2-38) notes that the USB estimate, therefore, is an estimate of what concentrations could be without U.S. anthropogenic emissions and not the fraction of observed O₃ that is USB.**

3.2.4 EPA's Conclusion on Background O₃ Modeling Estimates

EPA ISA (2020a, page 1-65) notes that while the seasonal mean USB concentration patterns may be important for identifying atmospheric processes leading to high USB concentrations and for understanding total O₃ exposures over long periods, *they are less relevant for estimating USB concentrations on days with high MDA8 concentrations and for understanding the role that hourly average background O₃ concentrations play in affecting human health and vegetation risk estimates.* On page 1-5 of the ISA (EPA, 2020a), the EPA notes that the averaging time of a USB estimate is intended to match the averaging time of the total O₃ concentration measured. In other words, if the EPA desired to quantify the percentage of background O₃ associated with the top ten daily maximum 8-h concentrations during the year at a specific monitoring site, the background O₃ should be reported in daily maximum 8-h concentrations.

Focusing on the seasonal means, whose values may be of interest to scientists attempting to understand atmospheric processes, on Page 1-65 in the ISA (EPA, 2020a), the EPA notes that the 2013 Ozone ISA (EPA, 2013) reported higher mean USB and NAB concentration estimates in spring than in summer for most regions of the U.S, and these results are consistent with earlier modeling estimates. However, EPA notes that some new results are consistent with this pattern (e.g., Lefohn et al., 2014), while other results suggest that summer USB and baseline O₃ concentrations can be comparable to (Jaffe et al., 2018) or greater than (Guo et al., 2018) spring concentrations. Guo et al. (2018) reported region-wide seasonal mean USB concentrations greater in summer than spring for most U.S. regions. The authors proposed that improvement of isoprene-NO_x chemistry was the reason for the difference in results compared to previous results that indicated springtime was the period of greatest background O₃ contributions.

EPA in the ISA (EPA, 2020a, page 1-65) believes the disagreement among researchers to be significant because numerous studies of USB and other measures of background O₃ have focused on spring as the season with the greatest USB concentrations. The ISA (EPA, 2020a, page 1-65) notes that

1. Recent publications have come up with conflicting conclusions about seasonal trends in USB. Higher seasonal mean USB concentrations in spring than in winter were reported for intermountain western sites (Fiore et al., 2014).
2. Fiore et al. (2014) reported higher seasonal mean NAB concentrations in spring than in summer at high-elevation western U.S. sites, consistent with the 2013 Ozone ISA (EPA, 2013).

3. Region-wide seasonal mean USB concentrations greater in summer than spring were reported for most U.S. regions (Guo et al., 2018). Improvement of isoprene-NO_x chemistry was proposed as the reason for the difference in results compared with earlier modeling results like those of Fiore et al. (2014).
4. Jaffe et al. (2018) reported comparable median spring and summer baseline O₃ concentrations at elevations >1 km in the western U.S., while below 1-km baseline O₃ concentrations were higher in spring.

EPA in its ISA (EPA, 2020a) did not resolve the conflicting conclusions about when seasonal mean background O₃ is greatest. EPA should have addressed the inconsistencies in its own latest modeling results concerning the seasonal patterns to assess the adequacy of the latest model predictions. The ISA (EPA, 2020a) dismissed the inconsistencies by indicating that seasonal mean USB metrics are less relevant for estimating USB concentrations when focusing on days with high MDA8 concentrations, as well as for understanding the role that hourly average background O₃ concentrations play in affecting human health and vegetation risk estimates.

On page 17 of Jaffe et al. (2018), the authors note that model-calculated USB O₃ was greatest in March through June, which agrees with the observations when stratospheric contributions are greatest at many high- and low-elevation sites across the U.S. (Lefohn et al., 2011, 2012). Jaffe et al. (2018) summarized their findings as follows concerning the seasonal behavior of background O₃:

Model-calculated USB O₃ is greatest in March through June, with monthly mean MDA8 mole fractions at higher elevations in the west of up to 50 ppb and annual 4th highest MDA8 values exceeding 60 ppb at some locations. Lower elevation cities nationwide have monthly mean USB O₃ of 20–40 ppb during the O₃ season. Daily variations, particularly in spring and early summer, can be due to stratospheric intrusions mixed with Asian pollution, which can contribute to observed MDA8 values over 70 ppb.

Using baseline O₃ data (Fig. 2 of Jaffe et al., 2018), Jaffe et al. (2018) illustrated the vertical profiles of O₃ at 4 sites in the West (Trinidad Head, Cheeka Peak, Mt. Bachelor Observatory, and Chews Ridge). The authors noted that at low elevations at the four sites, mean spring O₃ levels were about 10 ppb higher than summer values, whereas above 1 km, median spring and summer values were comparable with summer.

Guo et al. (2018) note that their model may have exaggerated the relative importance of enhanced background O₃ resulting from soil NO_x and isoprene. The authors noted in their paper that substantial biases in the severity and timing of high-O₃ events occurred in their model and that the model underestimated the frequency of high events in spring that they indicated were possibly associated with stratospheric intrusions. These important uncertainty statements mentioned in their paper may help explain their finding that USB O₃ tended to be higher in the summer than in the spring in most regions.

In the second draft PA (EPA, 2023), the authors summarize their conclusions from the EPA's updated background O₃ modeling results for the year 2016. These conclusions are as follows:

- For this analysis we did not attempt to quantify the contributions from individual Natural sources (e.g., lightning, soil, fires, stratosphere) or to address exceptional events beyond basic screening to remove very large fire plumes. (page 2-66).
- The USA contributions that drive predicted MDA8 total O₃ concentrations above 70 ppb are predicted to typically peak in summer. In this typical case, the predicted USB is overwhelmingly from Natural sources. The most notable exception to the typical case is reflected by predictions for an area near the Mexico border where the modeling indicates that a combination of Natural and Canada/Mexico contributions can lead to predicted MDA8 USB concentrations 60-80 ppb, on specific days, which is consistent with the O₃ PA prepared for the 2015 review (2014 PA, Section 2.4). (2-66).
- Consistent with previous assessments, USB is higher in the West than in the East. (page 2-65).
- Days for which MDA8 total O₃ concentrations are predicted to be above 70 ppb tend to have a substantially higher model-predicted USA (anthropogenic) contribution than other days in both the West and the East. (page 2-67).
- Predicted international contributions, in most places, are lowest during the season with the most frequent occurrence of MDA8 concentrations above 70 ppb. Except for the near-border areas, the International contribution requires long-distance transport that is most efficient in Spring. (page 2-67).
- The West has higher predicted USB concentrations than the East, which includes higher contributions from International and Natural sources. Within the West, high-elevation and near-border areas stand out as having particularly high USB. The high-elevation areas have more International and Natural contributions than low-interior areas in the same region. The near-border areas in the West can have substantially more international contribution than other parts of the West. (page 2-66).
- The current analysis indicates that natural and USA O₃ contributions peak during the traditional O₃ season (May through September), while long-range intercontinental transport of international O₃ (i.e., contributions from China, India etc.) peaks in the spring (February through May). (page 2-64).
- The natural contribution has a single maximum in late summer in the West, whereas, in the East there is evidence of two peaks— the largest in late Spring and a second peak in early Fall. (page 2-48).

- The Natural component of USB exhibits the largest magnitude difference between the West and East. (page 2-66).
- The contributions from Canada/Mexico at near-border locations are associated with relatively short-range transport and the seasonality peaks during May through September, similar to USA anthropogenic O₃. (page 2-64).
- Ozone formed from anthropogenic emissions in Canada and Mexico can peak in late spring or early summer when total O₃ is high. (page 2-65).
- Long-range transport and USA anthropogenic contributions tend to peak at different times of the year, so the contribution of international is often at its minimum when local sources are the driving factor for high total O₃ during the May through September O₃ season. (page 2-65).
- On high O₃ days (greater than 70 ppb) the West-East differences are largely associated with international contributions in near-border areas and natural contributions at high-elevation locations. (pages 2-65 to 2-66).
- In the Northern Hemisphere, the natural NO_x sources with the largest emissions are lightning (9.4 megatonN/yr), soils (5.5 megatonN/yr), and wildland fires (~2.2 megatonN/yr). (page 2-66).
- Because NO_x is the limiting precursor at hemispheric scales, the emissions data suggests that lightning and soils contribute are most likely the largest contributors to Natural O₃. As noted by Lapina et al. (2014), a large contribution from lightning may be the result of lightning strikes outside the U.S. while the contribution from soil NO_x tends to largest from emissions within the U.S. The distant lightning source is likely to have its effect as part of the well-mixed background. The local soil NO_x emissions have a clear seasonal cycle and is known to have large local contributions. (page 2-66).
- Wintertime O₃ events can be associated with emissions from local oil and gas production in the Intermountain West. Even though these episodes can occur as early in the year as February, international emissions may not contribute to them substantially. The conditions associated with these events result in decoupling of the local air masses from the upper atmosphere, essentially isolating air in the mountain valleys from the atmosphere above and reducing the influence of long-range transport compared to other winter and early spring days. As a result, these unique wintertime O₃ episodes may have little relative influence from international emissions despite occurring at a time of year when long-range transport from Asia is efficient. (page 2-65).

In the second draft PA (EPA, 2023), the EPA provides summary figures that are output from the 2016 USB analysis. For Fig. 3-36 (reproduced from Fig. 2-23 in the PA), the authors note on pages 2-48 and 2-49 of the PA the following:

The temporal pattern in the regional average clearly shows that the seasonality of MDA8 predictions for each total O₃ component varies by region. The natural contribution has a single maximum in late summer in the West, whereas, in the East there is evidence of two peaks—the largest in late Spring and a second peak in early Fall. The somewhat lower MDA8 O₃ in summer in the East requires further analysis but may be related to the lack of lightning emissions within the regional domain. The seasonality international contribution predictions is more similar between the two regions. The international contributions in both the West and East are greatest in Spring, but the contribution in the West is larger both at its peak and its trough, compared to the East. The total international contribution and the separately analyzed long-distance components (e.g., China, India, international shipping) peak in spring when O₃ lifetimes favor long-range transport (see Appendix 2B, Figure 2B-29). However, the Canada/Mexico component of international contributions peaks in summer because of the relative proximity to the U.S. receptors. The predicted USA contribution increases in the summer for both the West and the East, but the USA contribution in the West is smaller than in the East. As mentioned previously, this “all cells” average is disproportionately rural in the West. The following analysis looks further at the different types of land in the West, including urban areas that are more representative of population centers that behave differently than the “all cells” analysis.

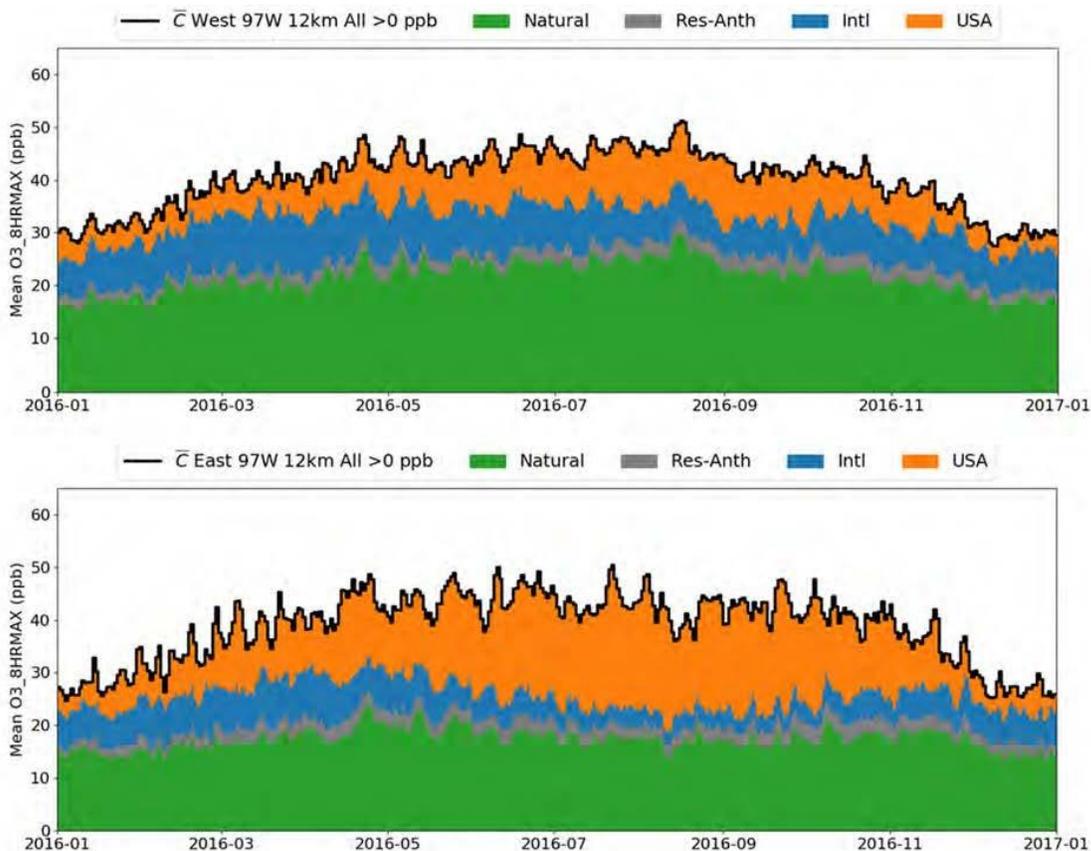


Figure 3-36. Annual time series of regional average MDA8 O₃ concentration and 8-h contributions for the West (top) and the East (bottom). See legend above from the original figure. Source EPA (2023 Fig. 2-23, page 2-49).

On page 2-50 of the second draft PA (EPA, 2023), the authors note that Fig. 2-24 (page 2-51 in the second draft PA) illustrates the contributions to the West split into three parts: the highest elevation areas, the near border areas, and Low/Interior areas with a weighted average focusing on urban areas. Fig. 3-37 is from the second draft PA (EPA, 2023). The authors note the following:

The urban area weighted average gives a larger weight to data in those urban areas that have dense emission sources (e.g., mobile). The urban area weighted average shows higher contribution from USA while Natural and International are lower compared to Figure 2-23. The differences between urban-weighted and non-weighted contributions are smaller in the East (not shown) than in the West (compare Figure 2-23 top and Figure 2-24 bottom). Compared to the West, the East has a larger fraction of land use that is urban (see Figure 2-22), which explains this difference. Thus, the non-weighted regional average contributions in the East includes the effects of urban areas much more so than the West. The seasonality of International is also different between the highest elevation areas, near border areas, and urbanized areas. At low/interior and at high-elevation sites, the simulated International contribution peaks earlier in the year than at border

sites. This earlier season peak is consistent with seasonality of O₃ lifetime necessary for long-range transport and a smaller contribution of long-distance sources (India, China, and Ships, see Appendix 2B, Figure 2B-30). At near-border sites, the seasonal cycle of predicted USB contributions from Canada/Mexico and from long-range transport combine to create a maximum later in the spring or early summer that is dominated by Canada/Mexico contributions (see Appendix 2B, Figure 2B-30, middle panel).

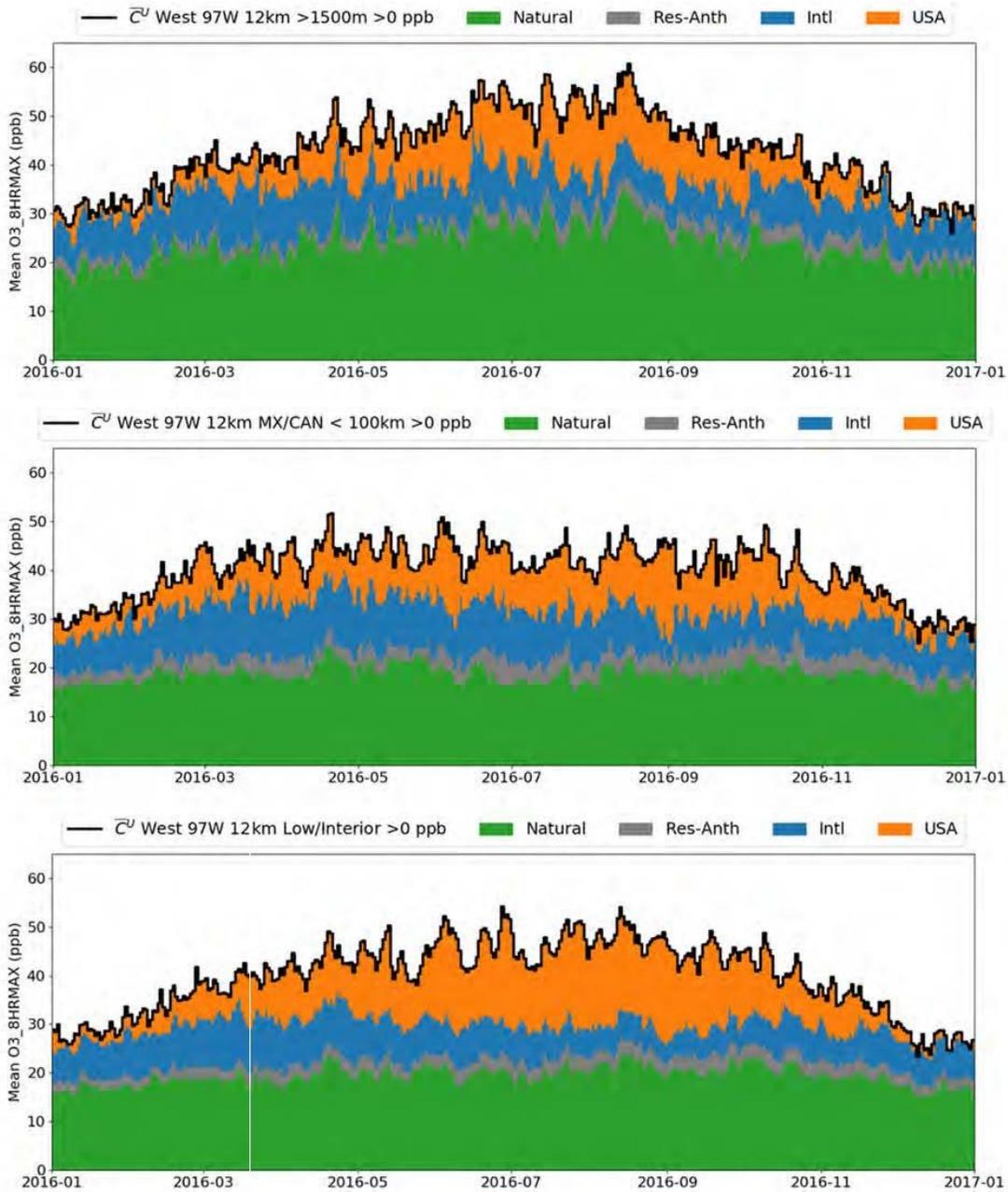


Figure 3-37. Annual time series of regional urban area-weighted average MDA8 O₃ concentration and 8-h contributions for the High-elevation West (top), near-boarder West (middle), and Low/Interior West (bottom). See legend above from the original figure. Source EPA (2023 page 2-51).

Earlier we discussed specific applications of background O₃ in the standard-setting process. While seasonal mean background O₃ estimates are of interest for identifying atmospheric processes, their use in assessing the role of background O₃ on exceedance days, as well as in human health and vegetation risk analyses, is limited. The time series of 8-h average daily maximum concentrations of background O₃ is important for some human health risk analyses and hourly average background O₃ concentrations are important for vegetation risk analyses. It appears in the reading the ISA (EPA, 2020a) and second draft PA (EPA, 2023) that the EPA is focused on understanding the percent contribution of background O₃ to current ambient levels on days when ambient levels exceed the standard. *However, as noted earlier, there are other considerations in the rulemaking process of equal importance for quantifying background O₃.* In many cases, assessment of human health and ecological risks is based on understanding the role that background O₃ plays throughout the distribution of hourly average concentrations and not just the role that background O₃ plays during periods of highest O₃ exposures. For the vegetation related W126 exposure index, each hourly average concentration is weighted and accumulated over a specific period for assessing vegetation effects. Hourly average background O₃ concentrations contribute to the observed concentrations and therefore, contribute to the cumulative risk. For some human health risk assessments, daily 8-h average concentrations are used in a time series. Daily maximum 8-h average concentrations contain background O₃ concentrations, which contribute to the estimated human health risk. Background O₃ affects the overall human health risk assessments, and therefore, can influence the margin of safety determinations required for establishing the O₃ NAAQS.

3.2.5 How Much Do Sources other than U.S. Anthropogenic Contribute to Current Ozone Exposures?

To answer the question of how much of the current O₃ exposure can be attributed to sources other than U.S. anthropogenic sources, both daily estimates of USB_{AB} (EPA, 2014a; Dolwick et al., 2015) and Emissions Influenced Background (EIB) (Lefohn et al., 2014) are used for illustrative purposes. Besides focusing on the highest hourly average O₃ concentrations (i.e., the upper end of the distribution of hourly values), it is important to understand the relative role that background O₃ plays over the entire distribution of total ambient O₃ concentrations. For example, for the human health risk associated with epidemiological models using no cutoff (i.e., no threshold), the lower end of the distribution plays an important role. Background O₃ contributes a large amount to the lower concentrations and cannot be reduced with reductions in emissions. For example, the high-elevation Yellowstone National Park site in Wyoming is dominated by background O₃ throughout the year with minor anthropogenic contributions (Lefohn et al., 2014). In Fig. 3-38 below, the relative comparison of EIB background levels (noted by blue) to anthropogenic (noted by red) within each concentration level shows that background contributes greater than 80%, including the mid-range concentrations (20-25 ppb),

which was an important concentration range that influenced EPA's human health risk estimates in 2014. As noted above, EIB O₃ for 2006 is similar to the USB_{AB} estimates for 2007 utilized by the EPA in its 2014 PA (EPA, 2014a). In comparison, Fig. 3-39 illustrates that for Denver the contribution of background within the mid-range concentrations is approximately 75 to 80%. For the Los Angeles area (Fig. 3-40), a site heavily influenced by anthropogenic emissions, background contributes 60-80% in the mid-range.

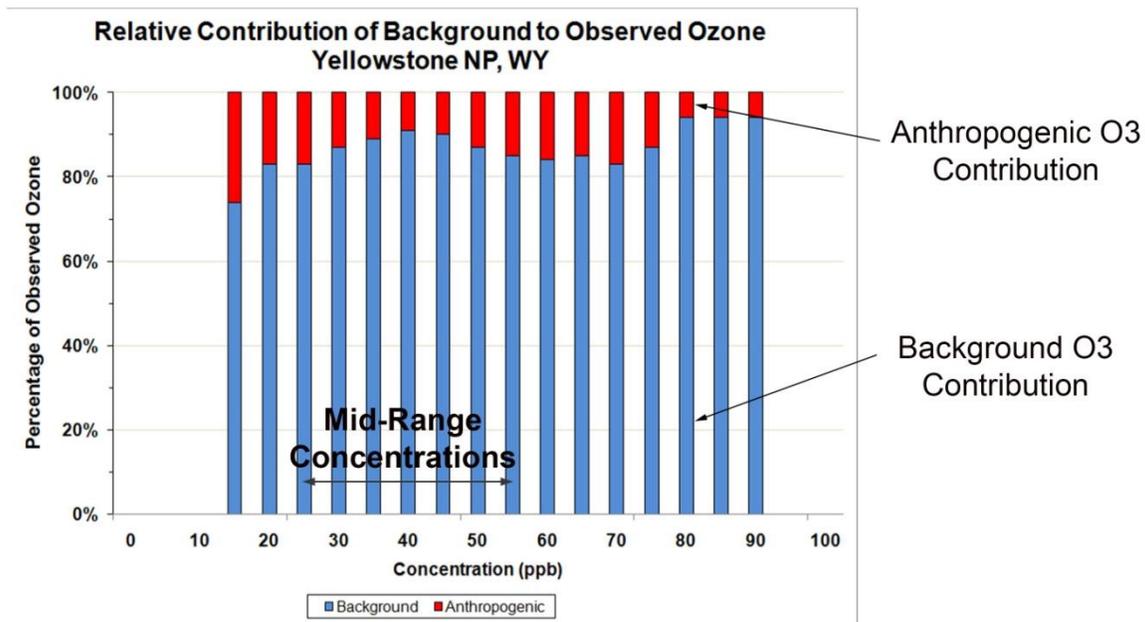


Figure 3-38. Average relative contributions of current hourly background (blue) and anthropogenic O₃ (red) for Yellowstone NP (WY) (AQS ID 560391011) in 2006. (Source: Lefohn et al., 2014).

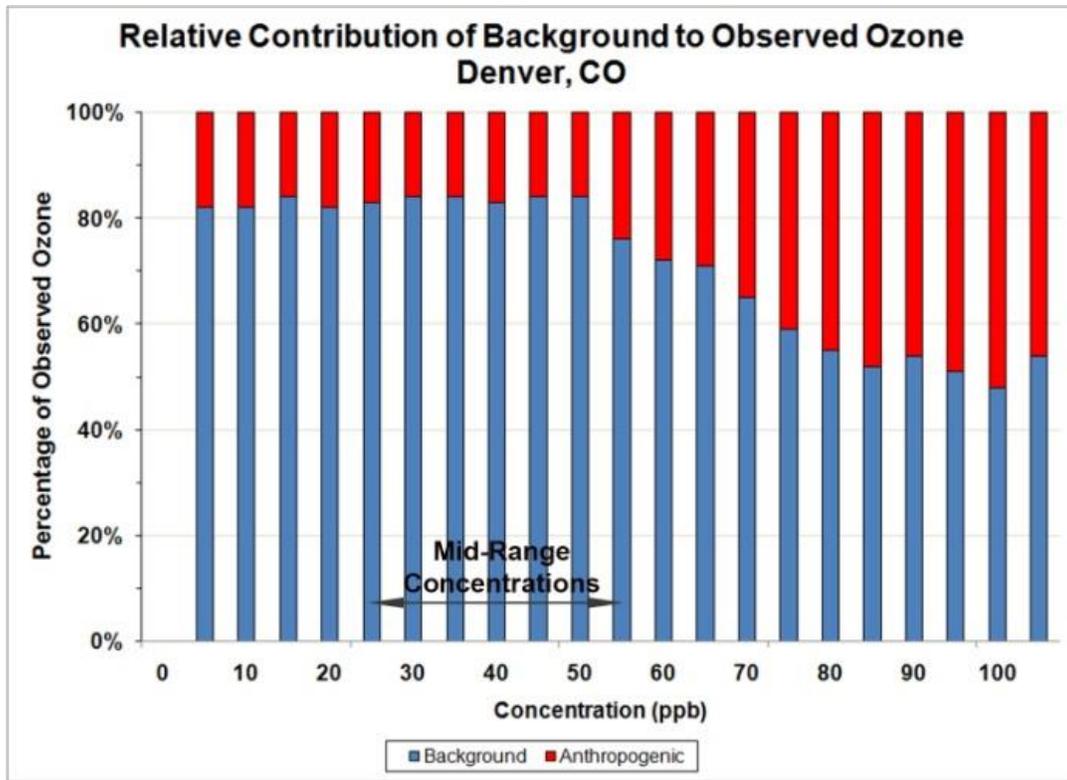


Figure 3-39. Average relative contributions of current hourly background (blue) and anthropogenic O₃ (red) for Denver (CO) (AQS ID 080590006) in 2006. (Source: Lefohn et al., 2014).

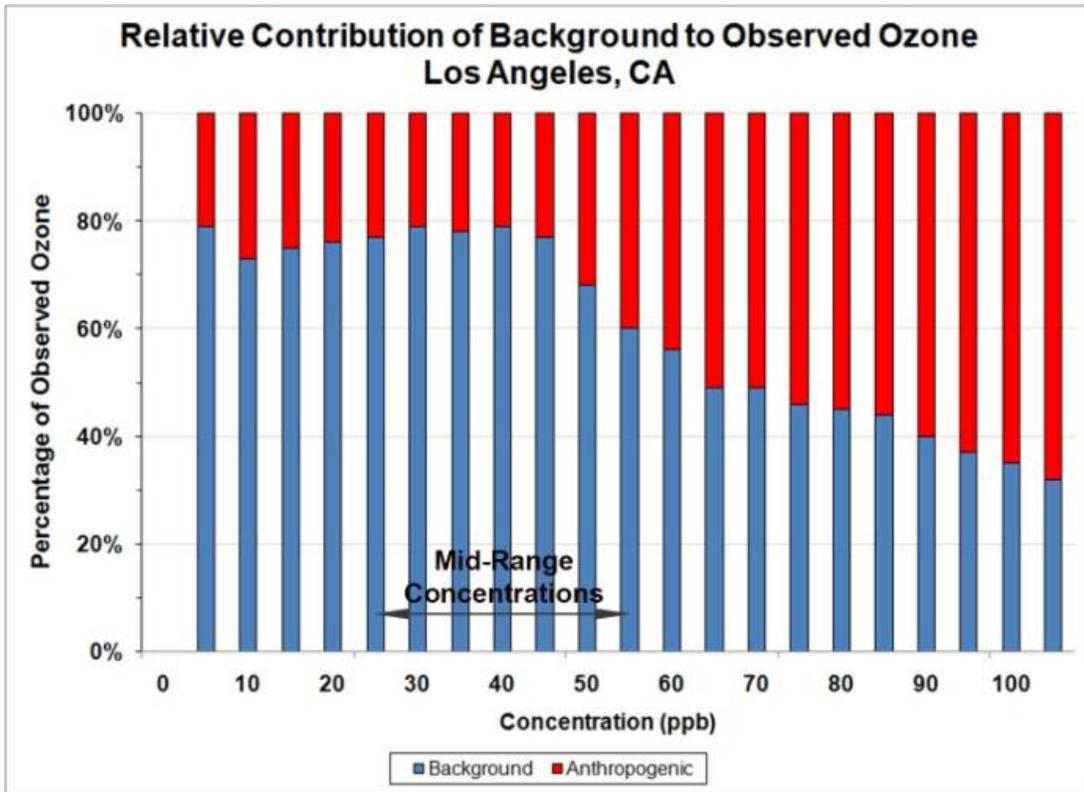


Figure 3-40. Average relative contributions of current hourly background (blue) and anthropogenic O₃ (red) for Los Angeles (CA) area (AQS ID 060719004) in 2006. (Source: Lefohn et al., 2014).

On page 2-23 of the EPA’s PA (EPA, 2014a), Fig. 3-41 below (Fig. 2-15 in EPA, 2014a) illustrates the distributions of the relative proportion of apportionment-based U.S. Background (USB_{AB}) to total O₃ (all site-days), binned by modeled MDA8 from the 2007 source apportionment simulation. The figure indicates that the USB_{AB} concentrations play an especially important role in both the low and mid-range total O₃ concentrations (EPA, 2014a).

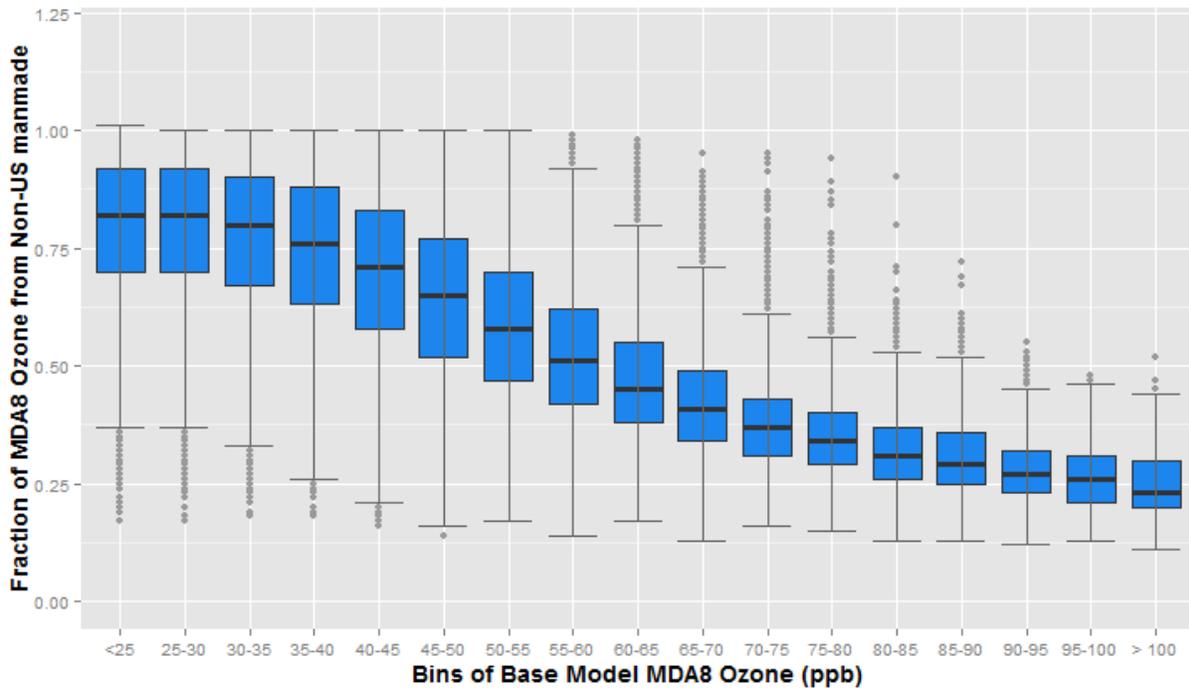
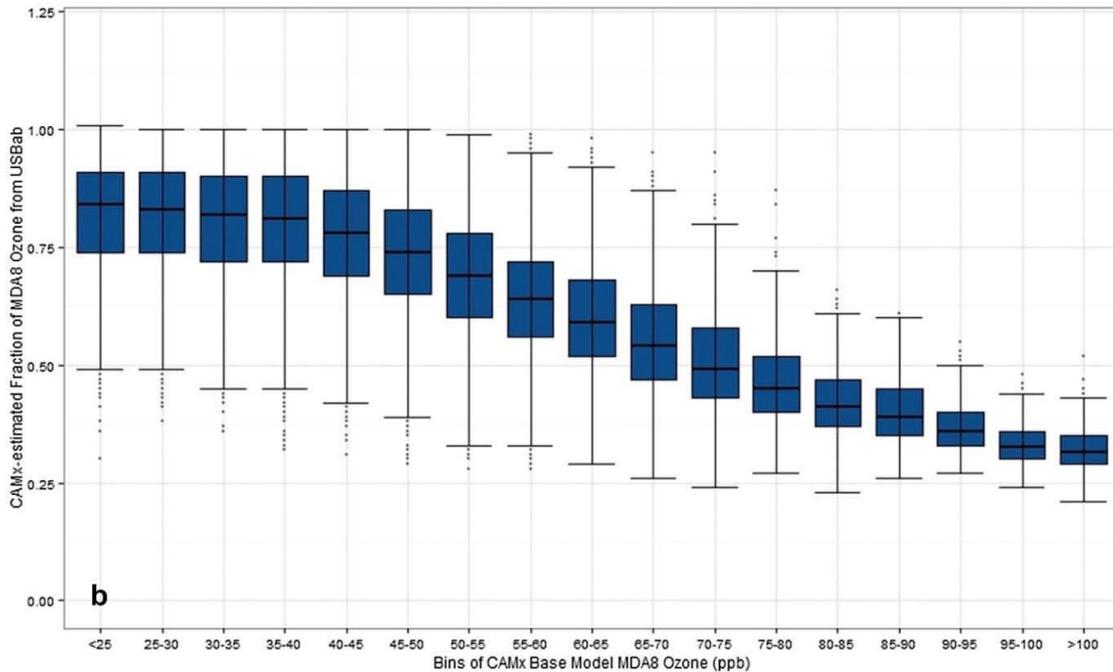


Figure 3-41. Distributions of the relative proportion of apportionment-based U.S. Background (USB_{AB}) to total O_3 (all site-days), binned by modeled MDA8 from the 2007 source-apportionment simulation. Source: Fig. 2-15 in EPA (2014a) with slight modification.

Fig. 3-42 below (Fig. 1-18b on Page 1-68 of ISA) illustrates CAMx estimates of daily distributions of bias-adjusted USB_{AB} O_3 fraction at monitoring locations across the western U.S. for the period April–October 2007, binned by base model MDA8 O_3 concentration ranges. Similar to the results presented in EPA’s 2014 PA (EPA, 2014a), the USB_{AB} concentrations play a very important role in both the low and mid-range total O_3 concentrations.



ppb = parts per billion.

Source: [Dolwick et al. \(2015\)](#).

Figure 3-42. CAMx estimates of daily distributions of bias-adjusted apportioned-based (USB_{AB}) O₃ fraction at monitoring locations across the western U.S. for the period April–October 2007, binned by base model MDA8 O₃ concentration ranges. Source: Figure from Dolwick et al. (2015) as reproduced in the ISA (EPA, 2020a) – page 1-68, Fig. 1-18b.

Lefohn et al. (2014) described a decreasing predicted relative contributions of background O₃ to total O₃ (identified by the authors as “Emissions Influenced Background”) with increasing total O₃ concentration. At low-elevation and urban sites in the western U.S., O₃ concentrations estimated as USB, USB_{AB}, NAB, or EIB contributions were also reported to be independent of overall O₃ concentration, resulting in a decreasing relative background contribution with increasing total O₃ concentration (Lefohn et al., 2014; Dolwick et al., 2015; Guo et al., 2018). However, model results do show increasing USB_{AB} and NAB predicted relative contributions of background O₃ to total O₃ with increasing O₃ concentration at high-elevation western U.S. sites (Fiore et al., 2014; Lefohn et al., 2014).

3.2.6 Sources of Background Ozone

Depending upon the part of the distribution of hourly average O₃ concentrations, U.S. background O₃ accounts for an important fraction of ambient O₃ concentrations because of stratospheric exchange, international transport, wildfires, lightning, global methane emissions, and natural biogenic and geogenic precursor emissions. As noted in the ISA (EPA, 2020a), as the literature on background O₃ has evolved, much of the discussion has focused on the relative

importance of stratospheric O₃ and intercontinental transport as the major sources of background O₃ (Page IS-14 of the ISA).

On Page IS-14 of the 2020 ISA (EPA, 2020a), the authors note that tropospheric O₃ derived from stratosphere-troposphere dynamics was described in detail in the 2013 Ozone ISA (EPA, 2013). Stratospheric air naturally rich in O₃ can be transported into the troposphere under certain meteorological circumstances, with maximum contributions observed at midlatitudes during the late winter and early spring. This process, known as “tropopause folding,” is characterized by episodic events typically lasting a few days from late winter through spring when deep stratospheric intrusions rich in O₃ can quickly and directly well into the troposphere and, more rarely, reach ground level (EPA, 2013). The 2013 Ozone ISA (EPA, 2013) also discussed the potential importance of deep convection, another form of stratosphere-troposphere exchange that occurs mainly in summer, as a mechanism for transporting stratospheric O₃ into the upper troposphere. Stratospheric intrusion events related to frontal passage and tropopause folding that reach the surface have less influence on surface O₃ during the summer months.

The relevance of stratospheric-to-tropospheric transport (STT) for influencing tropospheric O₃ concentrations has been well documented (e.g., Reed, 1955; Junge, 1962; Danielsen, 1968; Danielsen, 1974; Danielsen and Mohnen, 1977; Ludwig et al., 1977; Shapiro, 1980; Haagenson et al., 1981; Davies and Schuepbach, 1994; Lamarque and Hess, 1994; Schuepbach et al., 1999; Stohl et al., 2000; Lefohn et al., 2001; Cooper et al., 2005; Cristofanelli et al., 2006; Hocking et al., 2007; Ordóñez et al., 2007; Langford et al., 2009; Akriditis et al., 2010; Cristofanelli et al., 2010; Škerlak et al., 2014, 2019). Albers et al. (2018, 2022) have described the dynamical processes influencing the strength and frequency of anomalous STT of O₃. Lefohn et al. (2001) (authors A.S. Lefohn, S.J. Oltmans, T. Dann, and H.B. Singh) described the contribution of stratospheric O₃ to observed O₃ levels across the U.S. The authors attributed STT processes to the observation that hourly average O₃ concentrations ≥ 50 ppb occurred frequently during the photochemically quiescent months in the winter and spring at several rural sites across southern Canada and the northern U.S. In their paper, the authors described an STT event, where the stratosphere contributed on May 6, 1999, to enhanced O₃ concentrations in Boulder, Colorado. Data were provided courtesy of A. Langford of NOAA-Boulder. Eight years later, Langford et al. (2009) discussed in detail, using lidar and surface measurements, the May 6, 1999 contribution of stratosphere-to-troposphere transport to high surface O₃ along the Colorado Front Range.

In the paper by Langford et al. (2009), the authors indicated that others disputed the Lefohn et al. (2001) findings and used results from models to argue that the high-O₃ episodes described by Lefohn et al. (2001) could be explained by subsidence of free tropospheric air contaminated by North American anthropogenic sources and thus, did not represent true background values. Langford et al. (2009) noted that the modeling results were at odds with many other studies, which had presented evidence for significant stratospheric contributions to surface O₃ at both high-altitude sites (Schuepbach et al., 1999; Stohl et al., 2000) and near sea level (Cooper et al., 2005; Hocking et al., 2007). Langford et al. (2009) presented additional examples of deep STT contributing to high surface O₃ using lidar and surface measurements from the Front Range of the Colorado Rocky Mountains during the 1999 O₃ season (March–October). Their results showed that the stratospheric source was not only significant but could

directly lead to exceedances of the 2008 NAAQS standards in a major metropolitan area. Langford et al. (2009) described a deep tropopause fold that brought ~215 ppb of O₃ to within 1 km of the highest peaks in the Rocky Mountains on 6 May 1999. One-minute average O₃ concentrations exceeding 100 ppb were subsequently measured at a surface site in Boulder, and daily maximum 8-h O₃ concentrations greater or equal to the 2008 NAAQS O₃ standard of 0.075 ppm were recorded at 3 of 9 Front Range monitoring stations. Other springtime peaks in surface O₃ were also shown to coincide with passage of upper-level troughs and dry stable layers aloft. The authors noted that their results showed that the stratospheric contribution to surface O₃ was significant and could lead to exceedance of the 2008 NAAQS O₃ standards in a major U.S. metropolitan area.

Lefohn et al. (2011), using trajectory calculations, investigated the frequency of STT events and their associated enhancements on 12 surface O₃ monitoring sites in the western and northern tier of the U.S. The trajectory model introduced by Wernli and Davies (1997) was used to identify days of high probability for STT trajectories to enhance surface O₃ at specific monitoring sites. For most of the sites analyzed, Lefohn et al. (2011) indicated that contributions from stratosphere-to-troposphere transport to the surface (STT-S) were frequent during specific months and appeared to enhance the surface O₃ concentrations at both high- and low-elevation monitoring. Lefohn et al. (2012), quantified the frequency of STT events that result in O₃ concentration enhancements (i.e., hourly average concentrations \geq 50 ppb) observed at 39 high- and low-elevation monitoring sites in the U.S. during the years 2007-2009. They employed a forward trajectory-based approach to address the relationship between stratospheric intrusions and enhancements in hourly average O₃ concentrations. Their results indicated that STT down to the surface (STT-S) frequently contributed to enhanced surface O₃ hourly averaged concentrations at sites across the U.S., with substantial year-to-year variability. The O₃ concentrations associated with the STT-S events appeared to be large enough to enhance the measured O₃ concentrations during specific months of the year. Months with a statistically significant coincidence between enhanced O₃ concentrations and STT-S occurred most frequently at the high-elevation sites in the Intermountain West, as well as at the high-elevation sites in the West and East. These sites exhibited a preference for coincidences during the springtime and in some cases, the summer, fall, and late winter. Besides the high-elevation monitoring sites, low-elevation monitoring sites across the entire U.S. experienced enhanced O₃ concentrations coincident with STT-S events. Škerlak et al. (2014) noted that STT processes, which contribute to background O₃, affect the Intermountain West and other mountain ranges in the West year around, with a clear peak during the spring.

The STT-S counts, as has been estimated using the methodology described in Lefohn et al. (2011, 2012), have been compared with actual O₃ data. In the EPA AQS database, hourly average O₃ concentrations are at times marked with various coding by the state or tribe entity responsible for collecting the data. One specific code is “RO”. The code signifies that the governmental entity responsible for reporting the hourly O₃ data into the EPA’s AQS database plans to submit a demonstration that the value(s) should be excluded from the NAAQS calculations for attainment purposes. The Fig. 3-43 below illustrates for a site in the Denver area (AQS ID 080590006) for May 2012 the relationship between the STT-S trajectories described above and calculated by Professor Heini Wernli (Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland) and the “RO” codes embedded in the site’s hourly data file in the

AQS database. All hourly average O₃ concentrations in the AQS database from May 26 at 2000 through May 29 at 0100 local standard time (LST) were designated with an “RO” code. An hourly average O₃ concentration of 80 ppb was recorded on May 27, 2012 at 0300 LST, which would be 1000 GMT on the figure below. The STT-S counts appear to agree well with the “RO” designations in the AQS database for this site. Similar comparison agreements between STT-S trajectories and observed values have been described (Lefohn et al., 2011).

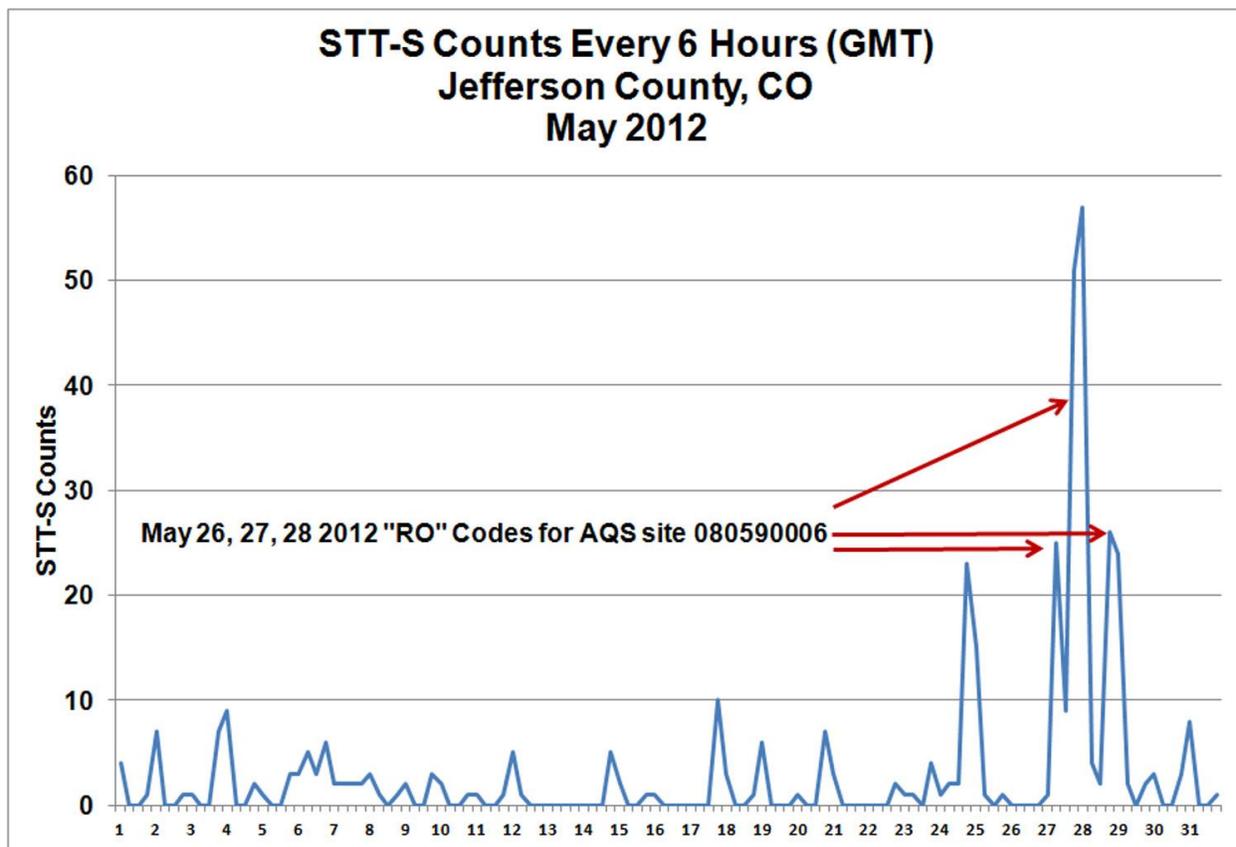


Figure 3-43. Relating STT-S counts for every six hours (GMT) with “RO” codes for a site in Jefferson County, Colorado (AQS ID 080590006) for May 2012. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

On page IS-15 of the ISA (EPA, 2020a), the authors state that international transport from Asia has also been identified as a major source of precursors that contribute about 5 to 7 ppb to USB O₃ concentrations over the western U.S. (EPA, 2013, 2006). Ozone precursor emissions from China and other Asian countries have been estimated to have more than doubled in the period 1990–2010, and an estimated increase of 0.3 to 0.5 ppb/year of mid-tropospheric O₃ USB in spring over the western U.S. in the two decades after 1990 was largely attributed to a tripling of Asian NO_x emissions. However, after this period, trends in NO_x emissions from China, the largest O₃ precursor source in Asia, have declined as confirmed by rapidly decreasing satellite derived tropospheric NO₂ column measurements over China since 2012. Stringent air quality standards implemented in 2013 within China have markedly reduced national emissions.

On page IS-15 of the ISA (EPA, 2020a), the authors note that other contributors (i.e., wildfires, lightning, global methane emissions, and natural biogenic and geogenic precursor emissions) to USB are either smaller or more uncertain than stratospheric and intercontinental contributions. EPA notes on page 1-1 of the ISA (EPA, 2020a) that an increasing trend of U.S. background concentration at high-elevation western U.S. sites before approximately 2010 now shows signs of slowing or even reversing, probably due to decreasing East Asian precursor emissions. However, as discussed later in this subsection, not all high-elevation western U.S. sites have experienced increasing O₃ trends.

One needs to explore further EPA's conclusion that trends at all western high-elevation sites are slowing or reversing due to decreasing East Asian precursor emissions. While emission reductions have occurred in China, O₃ concentrations continue to rise in some locations (Xu et al., 2020). Lefohn et al. (2017) analyzed trends using hourly average O₃ concentrations from monitoring sites in Hong Kong and mainland China and noted the following:

The changes in O₃ concentration distribution at the Chinese sites were most commonly associated with shifts towards higher concentrations, with the result that the metrics either increased in magnitude or showed no trend. Although NO_x emissions reductions occurred over a short time frame toward the end of the study period, mainland China and Hong Kong exhibited increasing trends in many of the exposure metrics. Although speculative, possible reasons for not observing significant trend reductions in the exposure metrics in China may be associated with the need for a longer period than six years (2010-2015) for emission changes to influence the metric trend patterns. In addition, the scarcity of monitoring stations could possibly contribute to lack of clear trend patterns. Year-to-year variability of meteorology could be a large factor in not observing decreases in the exposure metrics. In addition, further reductions in NO_x levels may be required before decreasing trends are observed. At many of the Chinese sites, O₃ formation is sensitive to VOCs rather than NO_x; VOCs have been increasing in mainland China (Ohara et al., 2007).

Lefohn et al. (2017) further noted

In contrast to decreasing emissions in the EU and US, emissions of NO_x have increased until recently in mainland China. Conversely, in Hong Kong, there have been large reductions in local emissions of both NO_x and VOC since 1997. However, peak ambient O₃ concentrations have not decreased due to the contribution of long-range transport from increasing O₃ levels from mainland China (Xue et al., 2014). NO_x emissions in China peaked around 2010-2011 and have since decreased (Duncan et al., 2016). Distribution changes at sites in China were most commonly associated with shifts towards higher concentrations.

Xu et al. (2020) presented an integrated analysis of long-term measurements of surface O₃ from eight sites distributed in the North China Plain (NCP) and Yangtze River Delta (YRD), the relatively underdeveloped region Northeast China, and the remote regions in Northwest and

Southwest China. Trends and present-day values for seven annual and five seasonal O₃ metrics were characterized through the year 2016. The authors discussed the long-term trends in the various O₃ metrics. Large and significant increases of O₃ were detected at the background site in the NCP, moderate increases at the global baseline site in western China, significant decreases at the northwestern edge of China, and nearly no trend at other sites. The summer O₃ metrics at the Shangdianzi background site in the NCP indicate increases at rates of more than 2%/yr during 2004–2016. In contrast, O₃ at the Lin'an (LA) background site in the YRD was constant over the period 2006–2016. Xu et al. (2020) note that although tropospheric NO₂ has been declining over the major air pollution regions in China, strong increases of O₃ at many of urban sites in China were observed during 2013–2017 (Lu et al., 2018). The authors noted that this implied that China's policy of emissions reduction in past years was unfavorable for controlling O₃ levels at urban sites, though it has stopped the increases of O₃ at some background sites and effectively lowered PM_{2.5} concentrations. Wang et al. (2022) note that limited long-term observations (i.e., >10 years) at several rural sites suggest that there have not been significant increases in O₃ concentrations in rural areas in the past decade.

Long-range transport from Asia has not influenced trend patterns at *all* western U.S. high-elevation O₃ monitoring sites. Not all high-elevation western U.S. sites have exhibited statistically significant trends during the springtime, when transport is anticipated to be highest from Asia. An evaluation of trend patterns of high-elevation western U.S. sites during the springtime, using the 4th highest daily maximum 8-h concentration exposure metric, shows that some sites did not experience increasing trends over the period 2000-2014. Using TOAR data (Schultz et al., 2017), Table 3-2 illustrates the trend patterns for spring (March-May) and summer (June-August) for 13 O₃ monitoring sites. Although not high-elevation sites, Glacier National Park (MT) and Denali National Park (AK) provide additional information concerning the effects of long-range transport from Asia on O₃ exposures. Oltmans et al. (2010) observed an O₃ episodic enhancement during April 2008 from biomass burning effluent from Eurasia that resulted in unusually high O₃ readings for this time of year in the western U.S. At Denali National Park in central Alaska, an hourly average of 79 ppb was recorded during an 8-h period in which the 8-h average was over 75 ppb, exceeding the O₃ ambient air quality standard threshold value in the U.S. The 8-h daily maximum at Yellowstone on 19 April (i.e., 69 ppb) suggests an enhancement during the period of suspected plume influence of 5-10 ppb above the other relatively high naturally caused O₃ values observed at the WY site. In Table 3-2, the nonparametric Mann-Kendall (M-K) test was used for testing for trends (see Lefohn et al., 2018 for additional information on the use of the M-K statistical method).

Table 3-2. Spring and summer trend patterns for the 4th highest daily maximum 8-h concentration exposure for 13 O₃ monitoring sites in the West for the March-April-May (MAM) and June-July-August (JJA) periods for 2000-2014.

Site	Site ID	Latitude	Longitude	Elev. (m)	MAM Trend	JJA Trend
Gothic	080519991	38.9564	-106.99	2926	No	Negative
Glacier NP	300298001	48.5103	-114	964	No	No
Yellowstone NP	560391011	44.5654	-110.4	2430	No	No
Pinedale	560359991	42.9288	-109.79	2388	No	Negative
Rainier NP	530530012	46.7841	-121.74	1615	No	Negative
Lassen Volcanic NP	060893003	40.54	-121.58	1755	No	Negative
Yosemite NP	060430003	37.7133	-119.71	1599	No	Negative
Great Basin NP	320330101	39.0051	-114.22	2058	No	No
Mesa Verde NP	080830101	37.1984	-108.49	2170	No	No
Denali NP	020680003	63.7232	-148.97	663	No	No
Chiricahua NM	040038001	32.0094	-109.39	1569	No	No
Grand Canyon NP	040058001	36.0586	-112.18	2070	No	Negative
Canyonlands NP	490370101	38.4583	-109.82	1794	No	Negative

NP=National Park

NM=National Monument

The ISA (EPA, 2020a) attributes increasing trend patterns observed until approximately 2010 at high-elevation western U.S. sites to long-range transport from Asia. However, as discussed above, this statement does not appear to be true for all high-elevation western U.S. sites. Additional emission reductions occurring in the future in Asia may have little influence on trends patterns at some western high-elevation O₃ monitoring sites.

3.2.7 What Do we Know about the Seasonal Pattern of Stratospheric-to-Tropospheric Transport to the Surface (STT-S) and Why is it Important?

The EPA's White Paper (EPA, 2015) stated the following:

Away from the earth's surface, O₃ can have an atmospheric lifetime on the order of weeks. As a result, background O₃, and to a lesser extent background O₃ precursors, can be transported long distances in the upper troposphere and be available to mix down to the surface when conditions are favorable. One of the largest natural sources of O₃ originates from production of O₃ in the stratosphere through interactions between ultraviolet light and molecular oxygen. O₃ exists in large quantities in the stratosphere and natural atmospheric exchange processes can transport stratospheric air into the troposphere. During certain meteorological conditions, discrete plumes of stratospheric air can be displaced far into the troposphere and impact ground-level O₃ concentrations. These events are called stratospheric intrusions and can result in relatively high USB levels of O₃ at the surface, especially at higher-elevation sites. Other natural sources of O₃ precursor

emissions include wildfires, lightning, and vegetation. Biogenic emissions of methane, which can be chemically converted to O₃ over relatively long time scales, can also contribute to USB O₃ levels. Finally, manmade precursor emissions from other countries can contribute to the global burden of O₃ in the troposphere and to increased USB O₃ levels.

In addition, page 1-25, the ISA states:

Deep stratospheric intrusions are common in the western U.S., impacting high elevation locations during the springtime. The incidence of tropopause folds is greatest in the early part (late winter and spring) of the year when synoptic-scale midlatitude cyclones are most active, occurring near upper level frontal zones where Rossby wave breaking is prevalent (Langford et al., 2017; Škerlak et al., 2015; EPA, 2013; Lin et al., 2012a).

Figs. 3-44, 3-45, 3-46, 3-47, and 3-48 illustrate as examples the daily maximum 8-h average concentration (MDA8) USB_{AB} estimates, observed daily MDA8 values, and the daily STT-S counts for Yellowstone National Park (WY), Jefferson County (CO), Rocky Mountain National Park (CO), Lassen Volcanic National Park (CA), and Sacramento (CA). For the Yellowstone National Park site, it appears that STT-S plays an important role during the April-October period. During the summer, when the STT-S counts are strongly reduced at the other three sites, USB_{AB} is slightly reduced at the high-elevation Jefferson and Rocky Mountain National Park sites and therefore, periods occur with “gaps” between observed values and USB_{AB}, likely attributable to anthropogenic sources. The amplitude of the “gap” varies strongly between the four sites. During rare events of STT-S > 0 in summer (e.g., Lassen, end of July and end of August), the two curves (i.e., total observed O₃ and USB_{AB}) approach one another, indicating that STT-S episodes can also occur in summer with the result there is a close agreement between observed values and USB_{AB}. For the Sacramento site, STT-S events occur during the spring and fall months. Gaps (i.e., the difference between the observed total O₃ and USB_{AB} concentrations) occur from mid-May through September, indicating the apparent influence of anthropogenic sources. STT-S events occur across the U.S. at all elevations with the result that USB_{AB} contributes in varying amounts (depending upon season and elevation of the site) to the observed O₃ concentrations across the U.S. USB_{AB}, while important in the high-elevation sites in the western U.S., is also important at low-elevation sites across the U.S. (Lefohn et al., 2011, 2012, 2014).

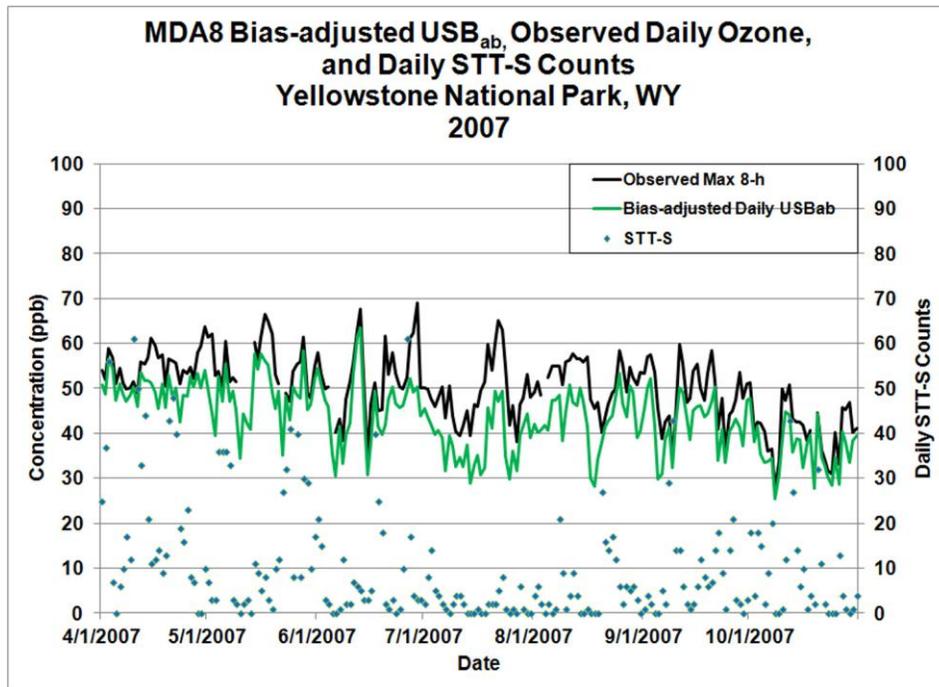


Figure 3-44. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for Yellowstone National Park (WY) (AQS ID 560391011) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

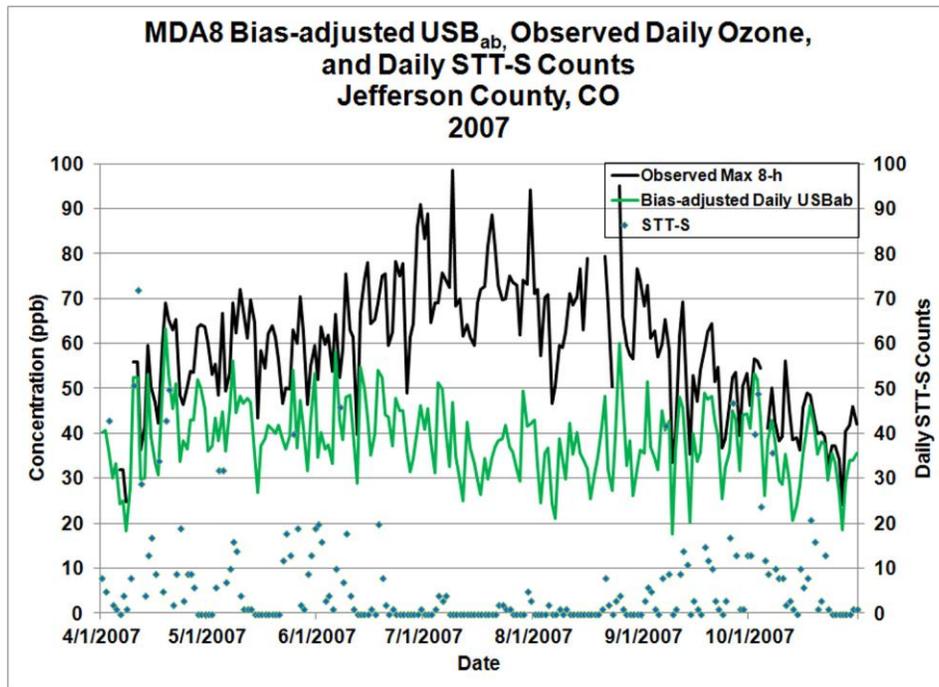


Figure 3-45. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for Jefferson County (CO) (AQS ID 080590006) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

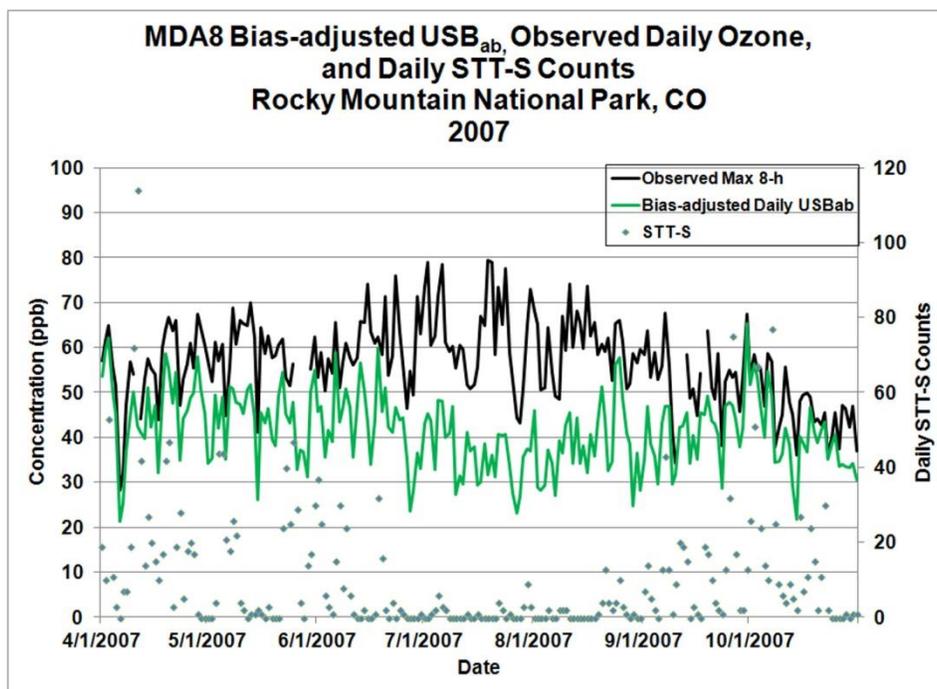


Figure 3-46. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for Rocky Mountain National Park (CO) (AQS ID 080690007) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

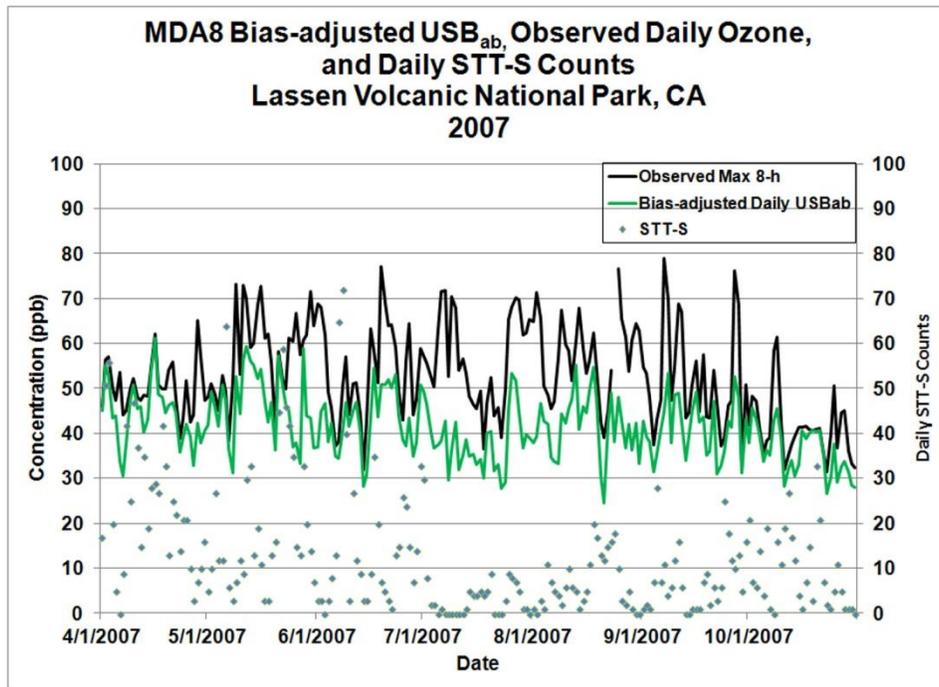


Figure 3-47. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for Lassen Volcanic National Park (CA) (AQS ID 060893003) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O_3 values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

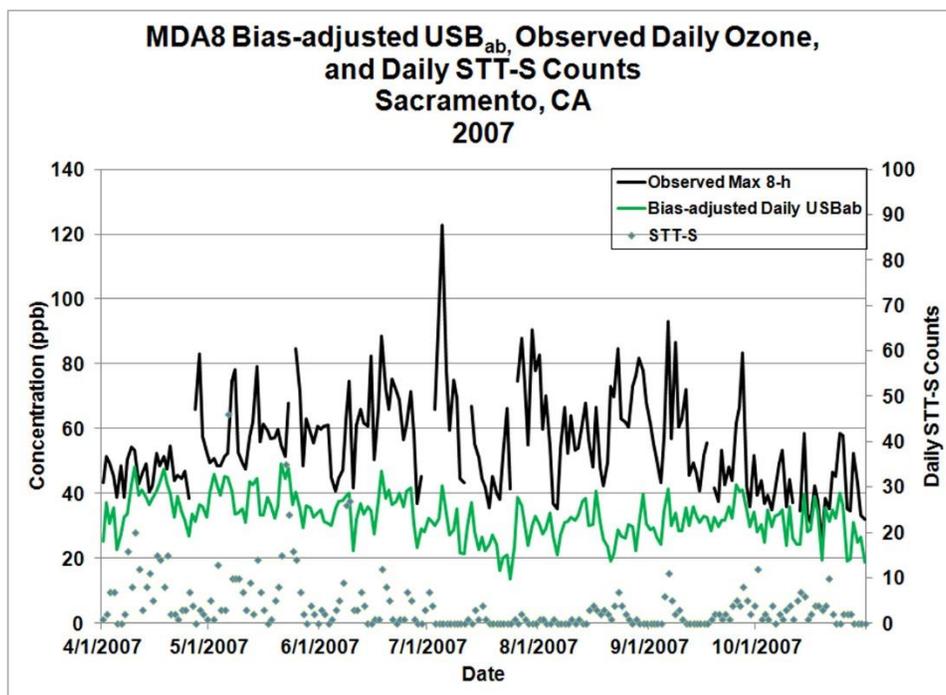


Figure 3-48. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for Sacramento (CA) (AQS ID 060670012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

3.2.8 Observed Ozone Exposure Patterns and Why the Patterns are Important

There appear to be substantial differences between (1) the most current background O₃ modeling results performed by the EPA and presented in the second draft PA (EPA, 2023) and (2) the EPA results performed in 2014 PA (EPA, 2014a), as well as other results published in the literature. On pages 2-30 and 2-31 of the 2014 PA (EPA, 2014a), the authors state the following:

For a variety of reasons, it is challenging to present a comprehensive summary of all the components and implications of background O₃. In many forums the term “background” is used generically and the lack of specificity can lead to confusion as to what sources are being considered. Additionally, it is well established that the impacts of background sources can vary greatly over space and time which makes it difficult to present a simple summary of background O₃ levels. Further, background O₃ can be generated by a variety of processes, each of which can lead to differential patterns in space and time, and which often have different regulatory ramifications. Finally, background O₃ is difficult to measure and thus, typically requires air quality modeling which has inherent uncertainties and

potential errors and biases. That said, EPA believes the following concise and three-step summary of the implications of background O₃ on the NAAQS review is appropriate, as based on previous modeling exercises and the more recent EPA analyses summarized herein. First, background O₃ exists and can comprise a considerable fraction of total seasonal mean MDA8 O₃ and W126 across the U.S. Air quality models can estimate the fractional contribution of background sources to total O₃ in an individual area. The largest absolute values of background (NB, NAB, USB, or apportionment-based USB) are modeled to occur at locations in the intermountain western U.S. and are maximized in the *spring and early summer seasons* (emphasis added). Second, the modeling indicates that U.S. anthropogenic emission sources are the dominant contributor to the majority of modeled O₃ exceedances of the NAAQS. Higher O₃ days generally have smaller fractional contributions from background. This finding indicates that the relative importance of background O₃ would increase were O₃ concentrations to decrease with a lower level of the O₃ NAAQS. Third and finally, while the majority of modeled O₃ exceedances have local and domestic regional emissions as their primary cause, there can be events where O₃ levels approach or exceed 60-75 ppb due to the influence of background sources. These events are relatively infrequent and EPA has policies that could allow for the exclusion of air quality monitoring data affected by these types of events from design value calculations.

To summarize the EPA's key observations in the 2014 PA (EPA, 2014a) about its modeling results stated

- First, background O₃ exists and can comprise a considerable fraction of total seasonal mean MDA8 O₃ and W126 across the U.S. Air quality models can estimate the fractional contribution of background sources to total O₃ in an individual area. The largest absolute values of background (NB, NAB, USB, or apportionment-based USB) are modeled to occur at locations in the intermountain western U.S. and are maximized in the spring and early summer seasons.
- Second, the modeling indicates that U.S. anthropogenic emission sources are the dominant contributor to the majority of modeled O₃ exceedances of the NAAQS. Higher O₃ days generally have smaller fractional contributions from background. This finding indicates that the relative importance of background O₃ would increase were O₃ concentrations to decrease with a lower level of the O₃ NAAQS.
- Third and finally, while the majority of modeled O₃ exceedances have local and domestic regional emissions as their primary cause, there can be events where O₃ levels approach or exceed 60-75 ppb due to the influence of background sources. These events are relatively infrequent and EPA has policies that could allow for the exclusion of air quality monitoring data affected by these types of events from design value calculations.

The seasonal pattern for background O₃ mentioned in the first bullet is supported by results summarized in the 2013 ISA (EPA, 2013). On page 2-17 of the 2014 PA (EPA, 2014a), the authors note

The ISA (EPA 2013, section 3.4) previously established that background concentrations vary spatially and temporally and that simulated mean background concentrations are highest at high-elevation sites within the western U.S. ***Background levels typically are greatest over the U.S. in the spring and early summer*** (emphasis added).

As noted in Section 3.2.4 of this review, the modeling results described in the second draft PA (EPA, 2023) indicate that

- The current analysis indicates that natural and USA O₃ contributions peak during the traditional O₃ season (May through September), while long-range intercontinental transport of international O₃ (i.e. contributions from China, India etc.) peaks in the spring (February through May). (page 2-64).
- The Natural component of USB exhibits the largest magnitude difference between the West and East. (page 2-66).
- The natural contribution has a single maximum in late summer in the West, whereas, in the East there is evidence of two peaks—the largest in late Spring and a second peak in early Fall. (page 2-48).

The EPA's USB modeling results summarized in the second draft PA (EPA, 2023) appear to show a different seasonal pattern for when background O₃ is highest when these results are compared with the previous conclusions in the 2014 PA (EPA, 2014a). As noted in the second draft PA (EPA, 2023), the USA contributions (i.e., anthropogenic) that drive exceedances generally peak in summer. As noted in the first bullet above, the Natural component in the model described in the second draft PA (EPA, 2023) also peaks during the traditional O₃ season. In contrast, as pointed out in the 2014 PA (EPA, 2014a), the ISA (EPA 2013, section 3.4) previously established that background concentrations are greatest over the U.S. in the spring and early summer (i.e., March – mid-June). As indicated in Section 3.2.4 of this review, the authors of the ISA (EPA, 2020a) did not resolve these conflicting conclusions identifying, based on background O₃ modeling, when seasonal mean background O₃ is greatest. Unfortunately, the second draft PA (EPA, 2023) did not adequately discuss these inconsistencies. However, a great majority of the peer-reviewed literature has identified spring and early summer as the period when background O₃ concentrations are the greatest across the U.S. In a review of the literature, Jaffe et al. (2018) concluded that model-calculated USB O₃ was greatest during the March through June period.

In the EPA's AQS database there is information that can provide guidance for better understanding the distribution of hourly average background O₃ concentrations, as well as the seasons that exhibit the highest background O₃ concentrations. In the database, information from some of the O₃ monitoring sites illustrate the compression described in the literature about the

distribution of hourly average O₃ concentrations, where the highest average values shift downward toward the mid-range concentrations and the lowest average concentrations shift upwards toward the mid values. In many cases for inland monitoring sites, the resulting distribution of hourly average concentrations resembles a bell-shaped-like curve.

The EPA, in cooperation with the U.S. Forest Service, established a network of air monitoring stations (referred to as National Air Pollution Background Network (NAPBN)), which was designed to measure levels of O₃ in remote areas within the contiguous 48 states (Evans et al., 1983). There were 8 monitoring sites (Green Mt. NF, VT; Kisatchie NF, LA; Custer NF, MT; Chequamegon NF, WI; Mark Twain NF, MO; Croatan NF, NC; Apache NF, AZ; Ochoco NF, OR) at various National Forests, which measured O₃, wind speed and direction, temperature, relative humidity, and solar radiation. The network was established to provide a reasonable long-term and continuous record of O₃ concentrations and patterns in areas well removed from anthropogenic sources of air pollution and to make these data available to the EPA and other interested researchers.

Of the 6 NAPBN stations operational for the entire year of 1979, all but the site at Custer NF in Montana recorded hourly average O₃ concentrations which were more than 0.08 ppm. Most days with elevated O₃ concentration, as defined by hourly average values >0.08 ppm, occurred in the spring and early summer months. Evans et al. (1983) hypothesized that the spring events, which occurred at the 7 NAPBN stations, may have been associated with natural sources (i.e., the stratosphere). The Custer NF site was located at Fort Howes in the eastern plains of Montana, near the Wyoming border. Fort Howes is located about 32 km south of Ashland, Montana. Inspecting the hourly average concentration data for the Custer NF site (1250 m, 45° 14' 00" N, 106° 15' 00" W), Fig. 3-49 illustrates that the frequency distribution appears to approach a Gaussian-like (i.e., bell shaped) distribution. Five hourly average concentrations of 75 ppb occurred on 25 April 1979. There were 16 hourly average concentrations at 70 ppb at the site, which occurred on 17 April (4 occurrences), 19 April (2 occurrences), 25 April (5 occurrences), and 26 April (5 occurrences). The two hourly instances on 19 April occurred at 0000 and 0100 in the early morning hours local standard time (LST). The 5 hourly instances of 70 ppb on 26 April occurred during the early morning hours of 0500-0900 LST. In other words, at the Montana site all the maximum hourly average O₃ concentrations occurred during the springtime and appear to have been related to possible stratospheric-to-tropospheric transport to the surface (STT-S). Without more detailed information concerning the meteorological conditions during the April 1979 periods, it is not possible to definitively associate the highest hourly exposures with natural stratospheric intrusions. However, it is a reasonable explanation for the cause of the elevated hourly O₃ concentrations and tends to support the hypothesis stated in Evans et al. (1983).

Lefohn et al. (1998) compared the Custer NF bell-shaped-like frequency distribution (Fig. 3-49) with the distribution of hourly average concentrations for an urban influenced site in Jefferson County, KY (AQS ID 211110027). The frequency distribution of the hourly average O₃ concentrations at the Kentucky site appeared to have a more log-normal-like shape (Fig. 3-50). In contrast to the Custer NF site, the urban-influenced site in Kentucky showed frequent high and low hourly average concentrations. Lefohn et al. (1998) noted that the Kentucky site

appeared to be influenced by NO titration of O₃ because of the occurrence of more frequent low hourly average concentrations.

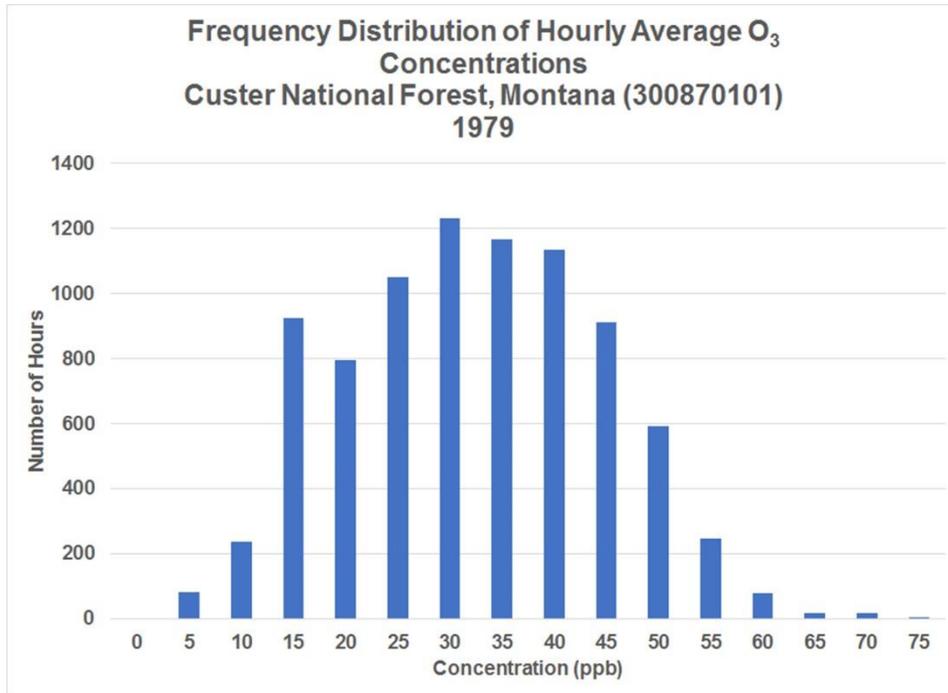


Figure 3-49. Frequency distribution of the hourly average O₃ concentrations in 1979 for Custer National Forest (MT) (300870101) monitoring station. Source of data is from the EPA's AQS database.

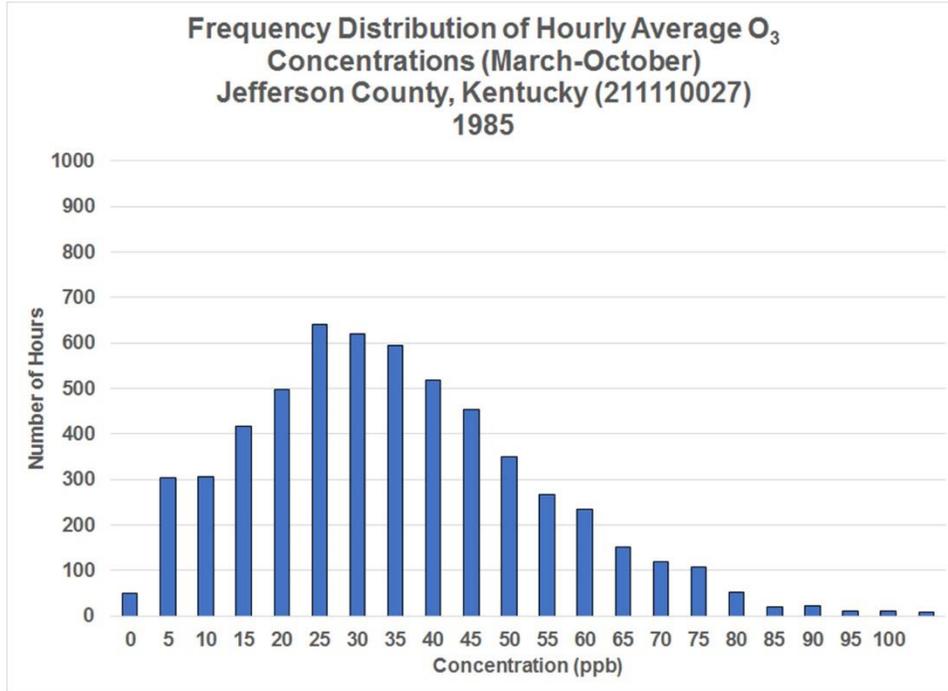


Figure 3-50. Frequency distribution of the hourly average O₃ concentrations in 1985 for an urban-influenced site in Jefferson County (KY) (211110027) monitoring station. Source of data is from the EPA’s AQS database.

Based on the comparison of the two sites, Lefohn et al. (1998) hypothesized that as adequate control strategies were implemented to meet the O₃ NAAQS that the distribution pattern of hourly average concentrations for inland monitoring sites might approach the distribution pattern observed at the Montana site, as well as other remote sites in the western U.S. The distribution of hourly average data for the same Kentucky site for 2017 is shown in Fig. 3-51. The distribution shape in 2017 is similar to the bell-shaped-like distribution observed for the Custer NF site in 1979. Although the site is still influenced by anthropogenic sources, the highest hourly average O₃ concentration has been reduced from 112 ppb (experienced in 1985) to 77 ppb (experienced in 2017). In addition, reviewing the two distribution figures for Jefferson County, Kentucky, illustrates the compression, where the highest hourly average concentrations moved downward toward the mid-range values and the lowest concentrations shifted upward toward the mid-range values.

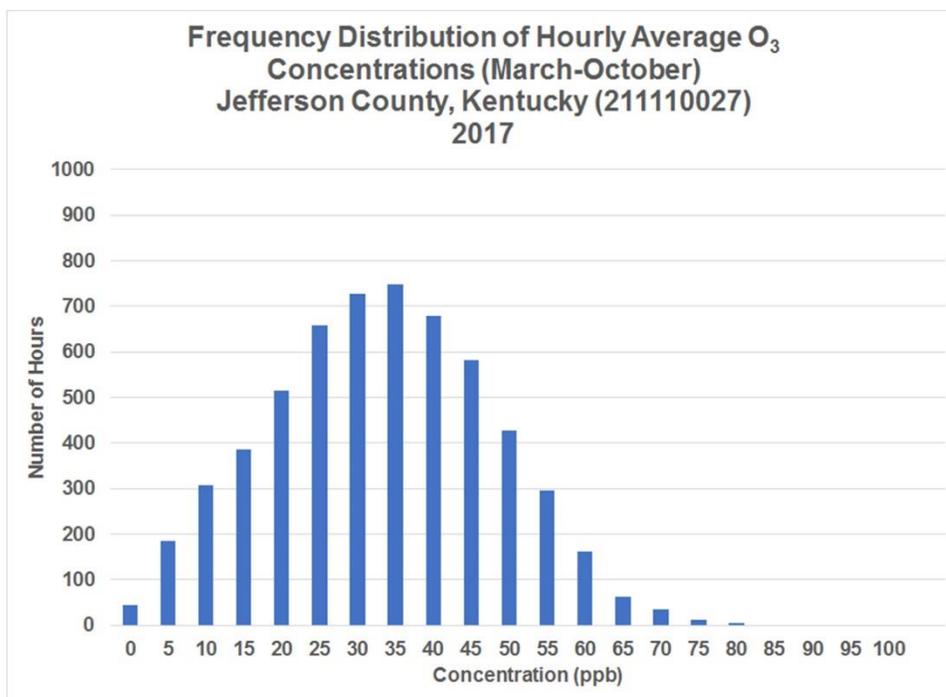


Figure 3-51. Frequency distribution of the hourly average O₃ concentrations in 2017 for an urban-influenced site in Jefferson County (KY) (211110027) monitoring station. Source of data is from the EPA’s AQS database.

Building upon the observations noted by Lefohn et al. (1998), EPA (2014a), and Simon et al. (2015), Lefohn et al. (2017) (authors: A. Lefohn, C. Malley, H. Simon, B. Wells, X. Xu, L. Zhang, and T. Wang) used data from 481 sites (276 in the EU, 196 in the U.S., and 9 in China) to investigate the response of 14 human health and vegetation O₃ exposure metrics to changes in hourly O₃ concentration distributions over time that resulted from changes in emissions. For the U.S. sites used in the analysis, the following selection criteria were used: (1) sites had to collect data with a minimum of 20 years (1995-2014); sites initiating monitoring as far back as 1980 were also used; (2) sites had to collect data year-round over the entire period; and (3) sites had to not experience large data gaps in the monitoring record with more than one large data gap of up to 1 year in length. The authors reported that at a majority of EU and U.S. sites, there was a reduction in the frequency of both relatively high and low hourly average O₃ concentrations. The patterns of changes in hourly average O₃ concentration distributions were separated into ten distinct ‘trend type’ categories. For each site, Lefohn et al. (2017) identified what portion of the distribution of hourly average concentrations had shifted. For characterizing patterns of change for the distributions, the trend types were described as follows (a yellow highlight below is provided to identify the interpretation of Trend Type 1):

- **Trend Type 0:** No trend.
- **Trend Type 1:** Both ends of the distribution shift toward the center. (Decreasing frequency of high and low concentrations).

- **Trend Type 2:** Low end shifts upward but high end does not change. (Decreasing frequency of low concentrations; increasing frequency of middle concentrations).
- **Trend Type 3:** High end shifts downwards but no change at lower end (Decreasing frequency of high concentrations; increasing frequency of middle concentrations).
- **Trend Type 4:** Entire distribution shifts downwards (Decreasing frequency of high concentrations, increasing frequency of low concentrations).
- **Trend Type 5:** The distribution shifts from the center toward both the high and the low ends of the distribution. (Increasing frequency of high and low concentrations).
- **Trend Type 6:** The middle of the distribution shifts downward but the high end does not change. (Increasing frequency of low concentrations, decreasing frequency of middle concentrations).
- **Trend Type 7:** The middle of the distribution shifts upward but the low end does not change. (Increasing frequency of high concentrations, decreasing frequency of middle concentrations).
- **Trend Type 8:** Entire distribution shifts upwards. (Increasing frequency of high concentrations, decreasing frequency of low concentrations).
- **Trend Type X:** Complex trends that do not fall into any of the categories listed above. It is not possible to categorize portions of the O₃ distribution into “low”, “middle”, and “high” for this trend type because the directions of the trends shift more than two times across the distribution.

Trend Type 1 (highlighted in yellow above), as discussed in Lefohn et al. (2017), identified the number of sites that exhibited the compression pattern described in Lefohn et al. (1998), EPA (2014a), and Simon et al. (2015). Lefohn et al. (2017) noted that because relative shifts of low and high hourly concentrations within the Trend Type 1 distribution can influence the median concentration, Trend Type 1 sites were further grouped in their analyses into three subcategories based upon trends in the median concentration: (1) “1a” sites had increasing trends in the median; (2) “1b” sites had no trend in the median; and (3) “1c” sites had decreasing trends in the median.

Table 3-3 (reproduced from Table 2 in Lefohn et al., 2017) below summarizes the trend type assignments for the 196 U.S. monitoring sites. Trend Type 1 (i.e., compression of the high- and low-end concentrations within the distribution, shifting more O₃ concentrations toward the center) was the most predominant trend pattern (84% of U.S. sites (165)). Most of the U.S. Trend Type 1 sites were classified as Trend Type 1a (i.e., increasing median); almost 30% were classified as Trend Type 1b (i.e., no trend in the median). Sixty-one percent of the U.S. sites analyzed by Lefohn et al. (2017) exhibited (1) compression of the high and low concentrations toward the middle of the distribution of hourly average O₃ concentrations and (2) increasing

trends in the median concentration. It would be anticipated that the observed increase in median concentrations would result in the average concentration also increasing for many of the same sites.

Table 3-3. Number of U.S. sites in each trend type category from the scatter dataset by degree of urbanization. Values in parentheses indicate the percent of rural, suburban, or urban sites that fall into each category. Source: Lefohn et al. (2017).

Trend Type	Rural	Suburban	Urban	Total
0 (No trend)	5 (6%)	0 (0%)	0 (0%)	5 (3%)
1a	31 (36%)	44 (76%)	44 (86%)	119 (61%)
1b	26 (30%)	11 (19%)	5 (10%)	42 (21%)
1c	4 (5%)	0 (0%)	0 (0%)	4 (2%)
2	4 (5%)	1 (2%)	2 (4%)	7 (4%)
3	7 (8%)	1 (2%)	0 (0%)	8 (4%)
4	9 (10%)	1 (2%)	0 (0%)	10 (5%)
7	1 (1%)	0 (0%)	0 (0%)	1 (<1%)
Total	87	58	51	196

The observations from Lefohn et al. (1998), Lefohn et al. (2010b), EPA (2014a), Simon et al. (2015), Lefohn et al. (2017), and Lefohn et al. (2018) indicate that emission reductions have resulted in some sites experiencing a compression of the distribution of hourly average O₃ concentrations and, in some cases, the compression of the high and low concentrations toward the middle results in a bell-shaped-liked distribution. *As emission reductions reduce the absolute value of the higher concentrations, background O₃ increases its role in the percent contribution to the total O₃ measured. Mathematically stated, in the limit, as U.S. anthropogenic contributions approach zero (which is not possible socially), what remains is a distribution of hourly average O₃ concentrations that represent background O₃ that, in many cases for inland sites, is described as a bell-shaped-like curve.*

Several examples exist today that illustrate the bell-shaped-like distribution described above. The amplitude and the width of the distribution vary from site to site. Fig. 3-52 illustrates the distribution of hourly average O₃ concentrations for 2018 for the Yellowstone National Park (WY) site. As indicated previously, the Yellowstone NP site exhibited no trend (using the nonparametric Mann-Kendall statistical test). For the year 2006, Lefohn et al. (2014) reported that background O₃ contributed a major portion of the total observed hourly average O₃ concentrations for all concentrations. Fig. 3-53 illustrates the distribution of hourly average O₃ concentrations for a site at Mesa Verde (CO) National Park. This site also exhibits the bell-shaped-like distribution pattern. Figs. 3-54 and 3-55 illustrate the distribution pattern for a site located in Garrett County, Maryland for 2005 and 2018, respectively. The site is identified in the AQS database as rural forested. In 2005, the Garrett County site experienced a maximum hourly average O₃ concentration of 100 ppb. In 2018, the site experienced a maximum hourly value of

75 ppb. Fig. 3-56 illustrates the distribution pattern for a site in Monroe County, Missouri. The site setting is listed as rural in the EPA’s AQS database. In 2000, the Monroe County site experienced a maximum hourly average O₃ concentration of 91 ppb. In 2018, the site exhibited a maximum hourly average O₃ concentration of 76 ppb.

The bell-shaped-like distribution curve provides us with the ability to gain insights about the distribution of background O₃ concentrations site by site. *While we cannot quantify the exact distribution of the hourly average background O₃ concentrations, we know that as the compression occurs as emission are reduced that background O₃ is encapsulated in the distribution.* For example, for the Yellowstone NP (WY) site in 2018 (Fig. 3-52), the highest frequency of hourly average O₃ concentrations is in the range of 40-45 ppb. The highest hourly average O₃ concentrations at this site are most likely associated with stratospheric contributions. The Mesa Verde (CO) site in 2018 (Fig. 3-53) shows the highest frequency of hourly average O₃ concentrations is in the range of 45-50 ppb. For the Garrett County (MD) site in 2018 (Fig. 3-55), the highest frequency of hourly average O₃ concentrations is 35 ppb. For the Monroe County (MO) site in 2018 (Fig. 3-56), the maximum frequency of O₃ concentrations is 30 ppb. In summary, for those sites still influenced by anthropogenic sources within the U.S./Canada/Mexico domain, as emissions continue to be reduced, the absolute values of the highest frequency in the distribution will continue to increase as background O₃ becomes even more dominant.

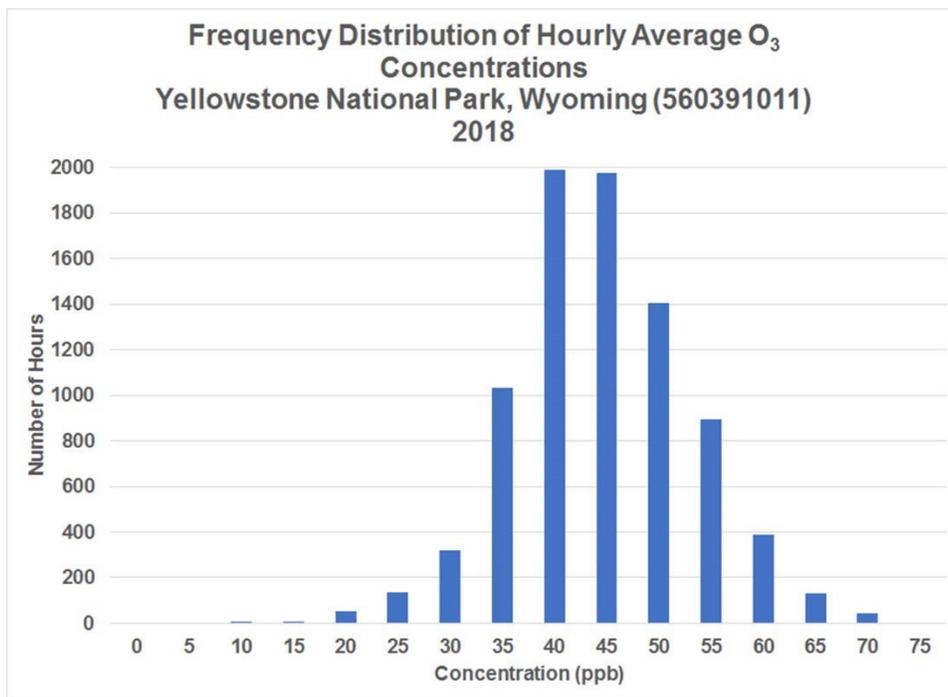


Figure 3-52. Frequency distribution of the hourly average O₃ concentrations for January-December 2018 for Yellowstone NP (WY) (560391011). Source of data is from the EPA’s AQS database.

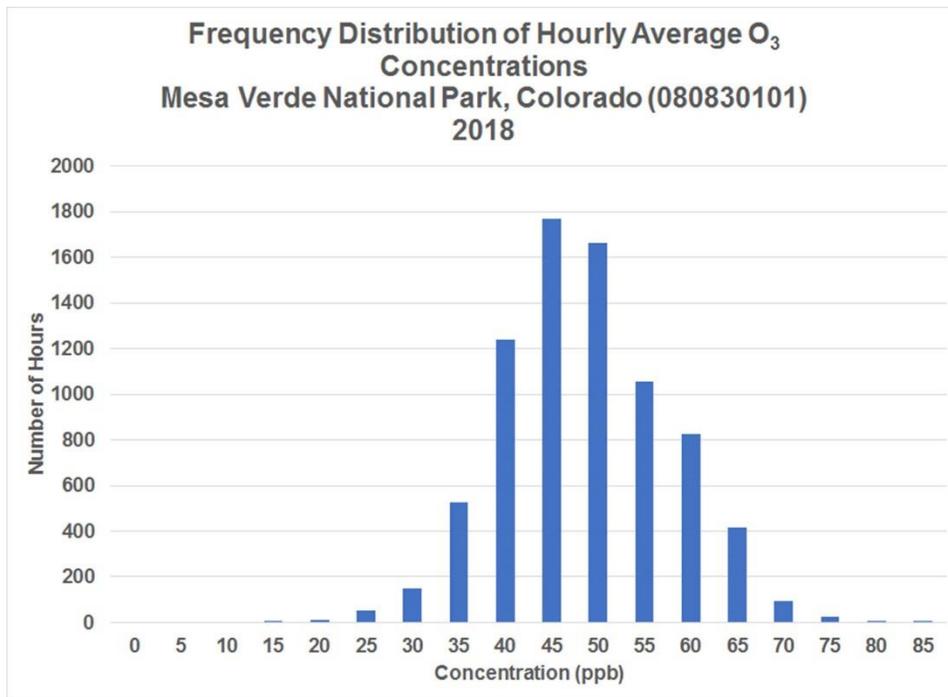


Figure 3-53. Frequency distribution of the hourly average O₃ concentrations for January-December 2018 for Mesa Verde National Park (CO) (080830101). Source of data is from the EPA’s AQS database.

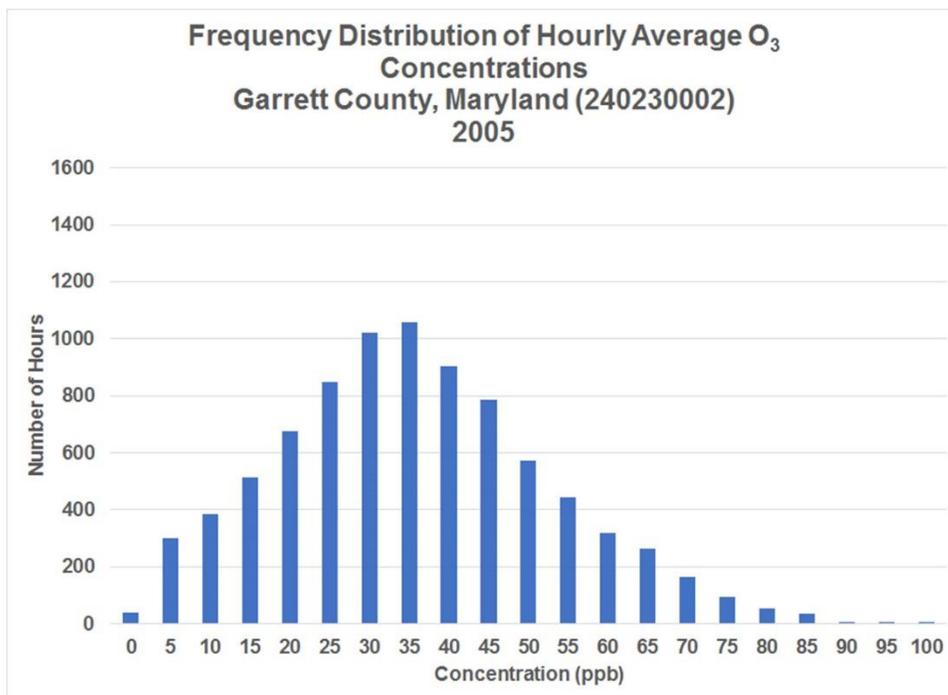


Figure 3-54. Frequency distribution of the hourly average O₃ concentrations for January-December 2005 for Garrett County (MD) (240230002). Source of data is from the EPA's AQS database.

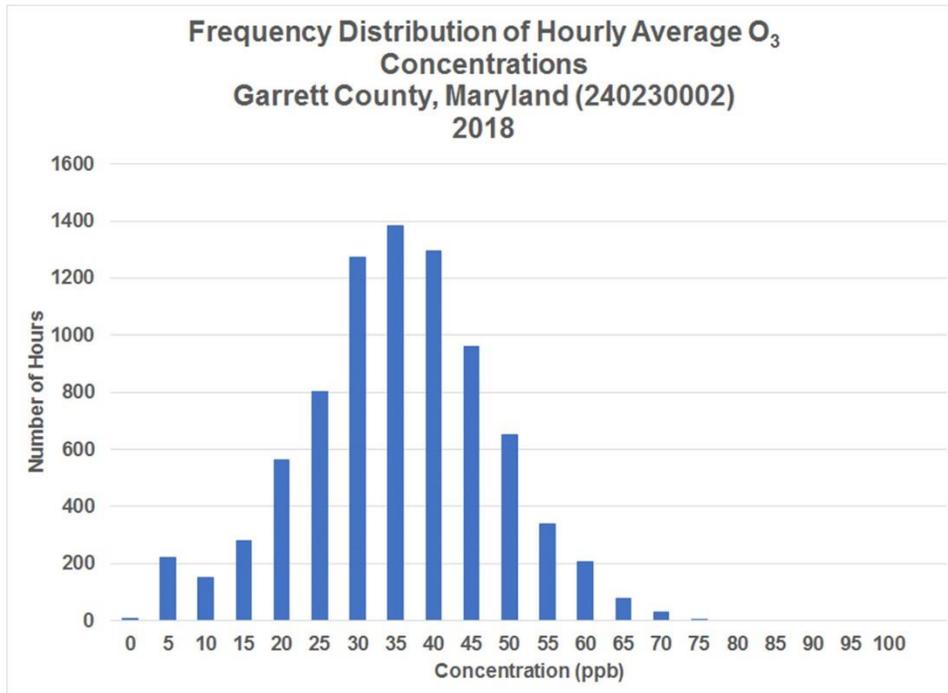


Figure 3-55. Frequency distribution of the hourly average O₃ concentrations for January-December 2018 for Garrett County (MD) (240230002). Source of data is from the EPA's AQS database.

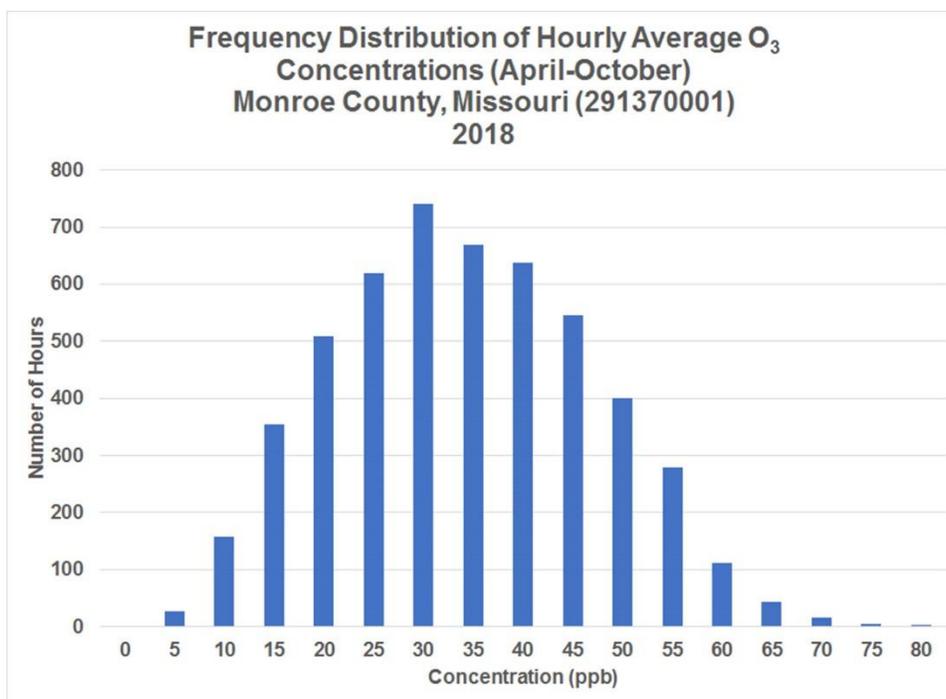


Figure 3-56. Frequency distribution of the hourly average O₃ concentrations for January-December 2018 for Monroe County (MO) (291370001). Source of data is from the EPA’s AQS database.

In addition to the observation of a compression of the distribution of hourly average O₃ concentrations (with the higher values shifting downward toward the mid-values and the lower values shifting upward toward the mid concentrations) as emission reductions occur, the period when the higher O₃ exposures occur has shifted from the July-August period toward the March-June months. The EPA in its Health Risk and Exposure Assessment analysis (EPA, 2014b) noted this shift in its risk assessment modeling results. In addition, EPA had concern that the O₃ monitoring season defined for each state and the District of Columbia was not adequately capturing the occurrence of daily maximum 8-h O₃ average concentrations equal to or above 0.060 ppm. In response to this concern, in the 2015 O₃ NAAQS rulemaking (Federal Register, 2015 – page 65416), the EPA determined that the lengthening of the O₃ monitoring seasons in 32 states and the District of Columbia was required. The Agency indicated that ambient O₃ concentrations in these areas could approach or exceed the level of the NAAQS, more frequently and during more months of the year compared with the length of the O₃ seasons prior to 2015. The EPA concluded that it was important to monitor for O₃ during the periods when ambient concentrations could approach the level of the NAAQS to ensure that the public was informed when exposure to O₃ could reach or has reached a level of concern. The EPA completed an analysis to address whether extensions of currently required monitoring seasons were appropriate (Rice, 2014). In the EPA analysis, all available data in the AQS were used, including data from monitors that collected O₃ data year-round during 2010-2013. More than half of the O₃ monitors were voluntarily operated on a year-round basis by monitoring agencies. The Agency determined the number of days where one or more monitors experienced a daily maximum 8-h O₃ average

equal to or above 0.060 ppm in the months outside each state's current O₃ monitoring season and the pattern of those days in the out-of-season months. The EPA believed that a threshold of 0.060 ppm, taking into consideration reasonable uncertainty, served as an appropriate indicator of ambient conditions that may be conducive to the formation of O₃ concentrations that approach or exceed the NAAQS. The Agency also considered regional consistency, particularly for those states with little available data. EPA noted that seasonal O₃ patterns varied year-to-year due primarily to highly variable meteorological conditions conducive to the formation of elevated O₃ concentrations early or late in the season in some years and not others. The EPA believed it was important that O₃ monitors operated during all periods when there was a reasonable possibility of ambient levels approaching the level of the NAAQS.

As a result of its analysis, modifications to the O₃ monitoring season involved adding earlier, as well as later months to the monitoring seasons that were used prior to 2015. Changes to the required O₃ monitoring seasons were finalized by the EPA (Federal Register, 2015 – page 65419) as follows for these states and the District of Columbia:

Colorado: Proposed addition of January, February, October, November, and December is finalized. The required season is revised to January – December.

Connecticut: Proposed addition of March is finalized, revising season to March – September.

Delaware: Proposed addition of March is finalized, revising season to March – October.

District of Columbia: Proposed addition of March is finalized, revising season to March – October.

Florida: Proposed addition of January, February, November, and December is finalized. The required season is revised to January – December.

Idaho: Proposed addition of April is finalized, revising season to April – September.

Illinois: Proposed addition of March is finalized, revising season to March – October.

Indiana: Proposed addition of March and October, revising season to March – October.

Iowa: Proposed addition of March is finalized, revising season to March – October.

Kansas: Proposed addition of March is finalized, revising season to March – October.

Maryland: Proposed addition of March is finalized, revising season to March – October.

Massachusetts: Proposed addition of March is finalized, revising season to March – September.

Michigan: Proposed addition of March and October is finalized, revising season to March – October.

Minnesota: Proposed addition of March is finalized, revising season to March – October.

Missouri: Proposed addition of March is finalized, revising season to March – October.

Montana: Proposed addition of April and May is finalized, revising season to April – September.

Nebraska: Proposed addition of March is finalized, revising season to March – October.

New Hampshire: Proposed addition of March is finalized, revising season to March – September.

New Jersey: Proposed addition of March is finalized, revising season to March – October.

New York: Proposed addition of March is finalized, revising season to March – October.

North Carolina: Proposed addition of March is finalized, revising season to March – October.

North Dakota: Proposed addition of March and April is finalized, revising season to March – September.

Ohio: Proposed addition of March is finalized, revising season to March – October.

Pennsylvania: Proposed addition of March is finalized, revising season to March – October.

Rhode Island: Proposed addition of March is finalized, revising season to March – September.

South Carolina: Proposed addition of March is finalized, revising season to March – October.

South Dakota: Proposed addition of March, April, May, and October is finalized, revising season to March – October.

Texas (Northern AQCR 022, 210, 211, 212, 215, 217, 218): Proposed addition of November is finalized, revising season to March – November.

Utah: Proposed addition of January, February, March, April, October, November, and December is finalized. The required season is revised to January – December.

Virginia: Proposed addition of March is finalized, revising season to March – October.

West Virginia: Proposed addition of March is finalized, revising season to March – October.

Wisconsin: Proposed addition of March and April 1 – 15 is finalized, revising season to March – October 15.

Wyoming: Proposed addition of January, February, March, and removal of October is finalized, revising season to January – September.

As discussed above, there is strong evidence supported in the literature that background O₃ across the U.S. is highest at many sites across the U.S. during the springtime (including into the month of June) and is an important contributor at many high-elevation sites throughout the year. As noted earlier (Section 3.1.2), actual O₃ monitoring data show that the highest O₃ exposures for the Park sites occur across the U.S. during the springtime and into early summer (i.e., March-June). As indicated earlier, the EPA in its 2014 PA (EPA, 2014c) (Welfare Appendix, page 7A-12) provided the highest 3-month W126 values and the timeframe corresponding to those W126 exposures for the Parks with O₃ monitors for the period 2006-2010. Using hourly average O₃ data from 57 National Parks, Table 7A-2 shows that several of the O₃ monitors in the Parks experienced their highest 3-month W126 exposures during the spring months and early summer.

One site not included in the table summarizing the analysis of O₃ monitoring data for the Parks in the EPA's 2014 PA (EPA, 2014c) (Welfare Appendix, page 7A-12) was the Look Rock site (TN) (470090101) in the Great Smoky Mountain NP (GRSM). Table 3-4 below illustrates the top-10 daily maximum 8-h average concentrations and the date/time associated with each occurrence. Beginning in 1989 and continuing to the present, the Park began monitoring at Look Rock (823 m), located on the Foothills Parkway on the TN side of the Park. As mentioned earlier, the Park has historically been subject to elevated O₃ levels (Neufeld et al., 2019). Neufeld et al. (2019) analyzed O₃ trends from 1989 to 2016 for six monitoring sites in and adjacent to GRSM and ranging in elevation from 564m to 2030m. Data from the Look Rock O₃ monitor were used in their analyses. The highest hourly average concentration in the Park was recorded at

the Look Rock site at 1600h on August 25, 1998. Note that in the early years (e.g., 1988), the top-10 8-h average concentrations occurred during the summer months. In 2018, all the top-10 8-h values occurred during the March-June period. The Look Rock (TN) site exhibited a shift of its top-10 8-h values from the summer to the March-June period. Neufeld et al. (2019) noted the shift in seasons following reduction of the peak concentrations may reflect a greater percentage of the remaining higher O₃ exposures being associated with background processes, such as stratospheric intrusions.

In addition to the Tennessee example, Blanchard et al. (2019) noted that the highest peak 8-h O₃ maxima typically occurred in summer throughout New York state prior to about 2010. Annual maxima now occur during spring at rural locations but continue to persist in summer in the New York City metropolitan area. Similarly, in the southeastern U.S., Blanchard and Hidy (2019) reported that the highest peak daily 8-h average O₃ concentrations tended to occur in summer, but the authors provide some evidence for a recent shift in the frequency of maxima to spring in some locations. Bowman et al. (2022) have discussed how changes in anthropogenic precursor emissions have driven shifts in the O₃ seasonal cycle throughout the northern midlatitude troposphere.

There are many sites within the EPA's AQS database that show similar shifting monthly patterns for the top-10 8-h values from the summer to the March-June period. For example, a site in Carter County, KY (210430500) illustrates the shifting pattern from the summer to the March-April months for the 1998-2018 period. The site is listed in the AQS database as rural residential. Table 3-5 (please see the table below) illustrates the top-10 daily maximum 8-h average concentrations and the date/time associated with each occurrence. In addition, there are many other sites in the AQS database that illustrate their highest 8-h daily maximum concentrations during the spring. The EPA performed an analysis in 2014, which addressed whether extensions of currently required monitoring seasons were appropriate (Rice, 2014). Performing an additional analysis and placing the results into the second draft Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards document (EPA, 2023) would add clarity to the situation and provide an update to the Agency's 2014 analysis.

In summary, in this section we identified several sites with bell-shaped-like distribution curves of the hourly average O₃ concentrations. We saw for the Yellowstone NP (WY) site, a location that does not exhibit statistically significant trends for the MDA8 metric using the Mann-Kendall nonparametric test, that the most frequent hourly average O₃ concentrations were in the 40-45 ppb range. Fig. 3-57 illustrates the average relative contributions of current hourly background (blue) and anthropogenic O₃ (red) in 2006 (Lefohn et al., 2014). The black line shows the distribution pattern overlaid onto the figure. Note that the distribution pattern observed (black line) in 2006 in Fig. 3-57 resembles the distribution pattern illustrated in Fig. 3-52 for 2018 and there appears to be little influence of anthropogenic sources. Fig. 3-57 described by Lefohn et al. (2014) illustrates that the percent background O₃ concentrations in the 40-45 ppb compared to total observed O₃ values consist mostly of estimated background O₃ concentrations. For those sites that are more influenced by anthropogenic sources than the Yellowstone NP site, as emission reductions occur, a compression of the distribution of hourly average O₃ concentrations (with the higher values shifting downward toward the mid-values and the lower values shifting upward toward the mid concentrations) occurs. As mentioned earlier in this

section, as emission reductions continue to decrease the absolute value of the higher concentrations, background O₃ increases its role as indicated by the percent contribution to the total O₃ measured. As U.S. anthropogenic contributions approach zero (which is not possible socially), what will remain is a distribution of hourly average O₃ concentrations that represent background O₃ that in many cases for inland sites is described as the bell-shaped-like curve.

Table 3-4. Top-10 daily maximum 8-h average concentrations and the date/time associated with each occurrence for Look Rock (TN) (470090101) O₃ monitoring site in Great Smoky Mountains National Park. All available data over the entire period of record were included in this analysis independent of the EPA-defined O₃ season.

Top-10	1998		1999		2005		2009		2012		2018	
1st	0.122	8/25/1998 14:00	0.110	7/23/1999 18:00	0.092	4/19/2005 14:00	0.084	6/25/2009 13:00	0.078	6/29/2012 12:00	0.073	6/7/2018 16:00
2nd	0.116	9/12/1998 16:00	0.110	8/18/1999 10:00	0.089	9/12/2005 15:00	0.070	6/24/2009 16:00	0.077	6/27/2012 18:00	0.069	6/5/2018 15:00
3rd	0.112	9/11/1998 14:00	0.107	9/4/1999 12:00	0.086	4/17/2005 16:00	0.069	4/18/2009 12:00	0.077	6/28/2012 13:00	0.068	6/15/2018 16:00
4th	0.110	9/2/1998 14:00	0.106	8/17/1999 19:00	0.086	7/25/2005 14:00	0.068	4/9/2009 15:00	0.075	7/1/2012 14:00	0.067	5/11/2018 13:00
5th	0.104	8/24/1998 20:00	0.105	9/1/1999 12:00	0.085	4/18/2005 12:00	0.068	4/17/2009 15:00	0.073	6/30/2012 12:00	0.066	3/16/2018 13:00
6th	0.103	9/4/1998 13:00	0.104	8/4/1999 17:00	0.085	6/22/2005 14:00	0.066	3/23/2009 12:00	0.071	6/24/2012 14:00	0.066	4/21/2018 13:00
7th	0.102	9/5/1998 16:00	0.104	8/6/1999 13:00	0.085	6/25/2005 12:00	0.066	6/26/2009 19:00	0.071	8/2/2012 15:00	0.065	4/12/2018 16:00
8th	0.099	9/17/1998 13:00	0.102	7/26/1999 13:00	0.084	5/18/2005 17:00	0.065	8/13/2009 14:00	0.070	6/22/2012 13:00	0.065	5/1/2018 15:00
9th	0.098	8/22/1998 15:00	0.102	9/2/1999 15:00	0.084	9/9/2005 13:00	0.064	4/8/2009 12:00	0.069	4/14/2012 17:00	0.064	4/13/2018 11:00
10th	0.098	8/29/1998 12:00	0.101	9/3/1999 14:00	0.084	9/20/2005 15:00	0.064	6/2/2009 10:00	0.069	6/23/2012 15:00	0.064	5/12/2018 17:00

Source: EPA AQS data run.

Table 3-5. Top-10 daily maximum 8-h average concentrations and the date/time associated with each occurrence for Carter County (KY) (210430500) O₃ monitoring site. All available data over the entire period of record were included in this analysis independent of the EPA-defined O₃ season.

Top-10	1998		2002		2008		2018	
1st	0.104	9/13/1998 11:00	0.094	8/3/2002 10:00	0.082	4/18/2008 10:00	0.064	4/13/2018 10:00
2nd	0.100	8/22/1998 11:00	0.093	9/9/2002 10:00	0.078	8/19/2008 11:00	0.064	4/27/2018 11:00
3rd	0.097	9/12/1998 11:00	0.088	6/22/2002 11:00	0.075	7/16/2008 11:00	0.063	5/1/2018 11:00
4th	0.096	8/23/1998 10:00	0.086	9/10/2002 10:00	0.072	5/1/2008 11:00	0.063	5/11/2018 11:00
5th	0.090	9/6/1998 11:00	0.083	7/5/2002 10:00	0.071	4/17/2008 11:00	0.061	4/21/2018 11:00
6th	0.090	9/7/1998 11:00	0.083	9/8/2002 10:00	0.070	5/6/2008 11:00	0.059	4/17/2018 12:00
7th	0.089	8/7/1998 10:00	0.080	8/9/2002 11:00	0.070	8/20/2008 11:00	0.058	5/12/2018 10:00
8th	0.088	8/20/1998 11:00	0.080	9/7/2002 10:00	0.069	5/7/2008 10:00	0.057	4/18/2018 13:00
9th	0.087	9/14/1998 10:00	0.079	5/24/2002 11:00	0.069	8/22/2008 11:00	0.057	5/9/2018 10:00
10th	0.086	5/15/1998 11:00	0.079	8/13/2002 10:00	0.067	5/29/2008 11:00	0.057	5/10/2018 11:00

Source: EPA AQS data run.

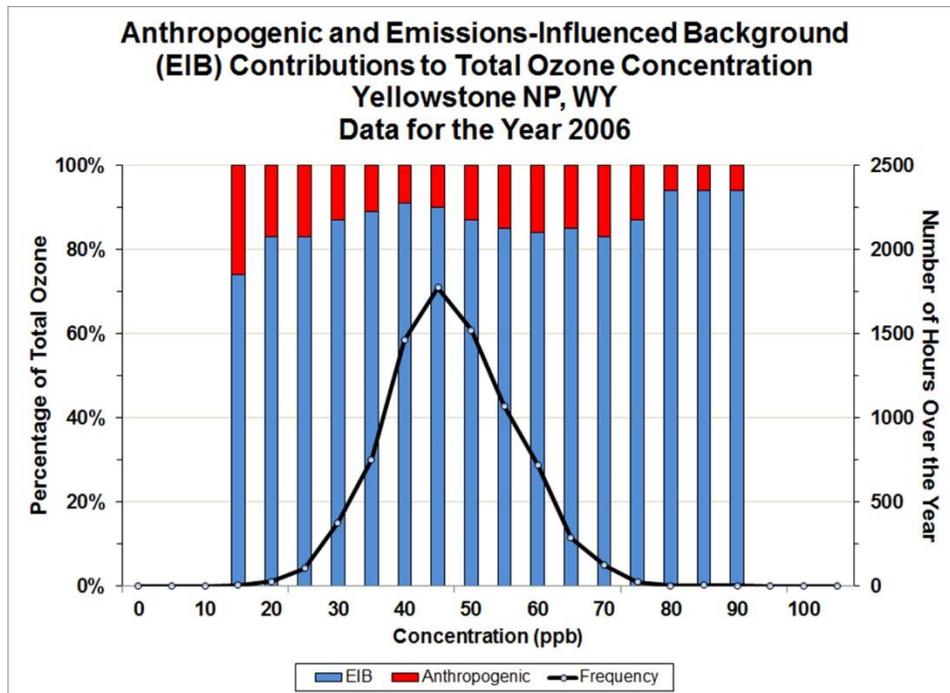


Figure 3-57. Average relative contributions of hourly background (blue) and anthropogenic O₃ (red) for Yellowstone NP (WY) (AQS ID 560391011) in 2006. The black line indicates the distribution of the hourly average O₃ concentrations. (Source: Lefohn et al., 2014).

While we cannot say at what point the bell-shaped-like curve will no longer change except for meteorological variability, if the curve remains stable from year-to-year except for minor variability, then we can gain insight about the distribution of background O₃.

Besides discussing the compression of the hourly average O₃ concentrations, we discussed a pattern where many sites within the EPA’s AQS database show shifting from the summer months to the March-June period for the higher daily maximum 8-h values. This pattern is important. As will be discussed in the next section, the seasonal change pattern provides us with the ability to overlay this information with the predictions indicated by background O₃ modeling.

3.2.9 Observed Ozone Exposure Patterns and Model Performance

The USB modeling results described in the second draft PA (EPA, 2023) indicate the following seasonal patterns:

- The current analysis indicates that natural and USA O₃ contributions peak during the traditional O₃ season (May through September), while long-range intercontinental transport of international O₃ (i.e., contributions from China, India etc.) peaks in the spring (February through May). (page 2-64).

- The natural contribution has a single maximum in late summer in the West, whereas, in the East there is evidence of two peaks— the largest in late Spring and a second peak in early Fall. (page 2-48).

As indicated in Section 3.2.4 and the previous section, the USB modeling results summarized in the first bullet above as indicated in the second draft PA (EPA, 2023) show a different seasonal pattern for when background O₃ is highest than the pattern reported in previous published results, including the EPA's own reports (EPA, 2013, 2014a). The bullets indicate that the Natural component (which is the largest contributor to background O₃) of the current EPA model is predicted to occur in the West in late summer. The previous conclusion in the 2014 PA (EPA, 2014a) and the 2013 ISA (EPA, 2013, in section 3.4) was that background O₃ was greatest over the U.S. during the spring and early summer (i.e., June).

Bias adjustment to estimated background O₃ estimates has been described in the literature. Dolwick et al. (2015) and Lefohn et al. (2014) used bias-adjusted estimates for USB_{AB} and EIB, respectively. Lefohn et al. (2014) concluded that, based on the tendency for their model to underestimate STT processes, the upward adjusted values of the hourly EIB concentration range were preferable to using an average or median value of the hourly range, especially during the spring at high-elevation sites. In their analyses, Lefohn et al. (2014) noted that model performance at low-elevation sites tended toward larger under prediction biases in cool months (i.e., November-April) and larger over prediction biases in warm months (June-October), particularly for sites in the southern and eastern U.S. The authors attributed site-specific monthly under and over predictions to their Global background O₃ (GBO₃) or anthropogenic O₃ modeling components. Global background O₃ (GBO₃) was defined as the sum of the global tropospheric and stratospheric components. For almost all high-elevation sites in their analyses, the model under predicted in the spring months when the above-median MDA8 GBO₃ coincided with diagnosed STT-S events as per the stratospheric trajectory analyses performed by Dr. Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich in Switzerland. Table 3-6 (Table 4 from Lefohn et al., 2014), summarizes the months when STT-S events coincided with months of highest MDA8 GBO₃ and the model tended to be under (u) or over (o) predicted. As indicated above, for almost all high-elevation sites, the model under predicted in the spring when the above-median MDA8 GBO₃ coincided with diagnosed STT-S events. Given the relatively small spring anthropogenic contributions at these sites, Lefohn et al. (2014) believed it was likely that the underestimates were associated with GBO₃. For some urban sites (i.e., Boston, Dallas, Detroit, New York, and Sacramento), the model also under predicted during the spring when higher GBO₃ coincided with diagnosed STT-S events; for others (i.e., Atlanta, Baltimore, Chicago, Cleveland, Georgia Station, Houston, Philadelphia, St. Louis, and Washington DC), spring over predictions occurred when higher GBO₃ coincided with STT-S events. For urban sites with higher spring anthropogenic influences, it was more likely that over predictions were associated with anthropogenic O₃.

The important role that the stratosphere played in episodic (i.e., short-term, high concentration events), as well as enhancements (subtle increases in O₃ concentration), to surface O₃ values was noted in Lefohn et al. (2012). The authors quantified the frequency of stratosphere-troposphere exchange (STE) events that result in O₃ concentration enhancements

(i.e., hourly average concentrations ≥ 50 ppb) observed at 39 high- and low-elevation monitoring sites in the U.S. during the years 2007-2009. The authors employed a forward trajectory-based approach to address the relationship between stratospheric intrusions and enhancements in hourly average O₃ concentrations. The authors results indicated that STT down to the surface (STT-S) frequently contributed to enhanced surface O₃ hourly averaged concentrations at sites across the U.S., with substantial year-to-year variability. The O₃ concentrations associated with the STT-S events appeared to be large enough to enhance the measured O₃ concentrations during specific months of the year. Months with a statistically significant coincidence between enhanced O₃ concentrations and STT-S occurred most frequently at the high-elevation sites in the Intermountain West, as well as at the high-elevation sites in the West and East. These sites exhibited a preference for coincidences during the springtime and in some cases, the summer, fall, and late winter. Besides the high-elevation monitoring sites, low-elevation monitoring sites across the entire U.S. experienced enhanced O₃ concentrations coincident with STT-S events. Tables 3-7 – 3-12 (Supplemental Tables S-1 – S-6 in Lefohn et al., 2012) provide a detailed description of the number of days in which the daily maximum hourly average O₃ concentration was ≥ 50 ppb and coincident with a direct STT-S event for each of the statistically significant STT-S months. The tables provide an indication of the variability of the coincidences at each site across years.

Table 3-6. Months when STT-S events coincided with months of highest MDA8 GBO₃ and the model tended to be under (u) or over (o) predicted. Source: Reproduced from Table 4 in Lefohn et al. (2014).

Site Name	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
High Elevation												
Yosemite NP, CA	o	o	u	u	u	u		u		u		
Denver, CO	u	u	u	u	u	u						
Gothic, CO	u	o	u	u	o	o						
Pinedale, WY		u	u	u	u							
Yellowstone NP, WY		u	u	u	u	u	o	o				
Shenandoah NP, VA		o	o	u								
Low Elevation												
Atlanta, GA			o	o								
Baltimore, MD			o									
Boston, MA		u										
Chicago, IL		o										
Cleveland, OH				o								
Dallas, TX			u									
Detroit, MI					u							
Georgia Station, GA				o								
Houston, TX			o									
Los Angeles, CA		o		o	u	u						
New York, NY				u								
Philadelphia, PA					o							
Sacramento, CA		o		u	u							
Seattle, WA												
St. Louis, MO				u	o					o		
Voyageurs NP, MN					u							
Washington DC			o									

On page 2-66 of the second draft PA (EPA, 2023), the authors note

For this analysis, we did not attempt to quantify the contributions from individual Natural sources (e.g., lightning, soil, fires, stratosphere) or to address exceptional events beyond basic screening to remove very large fire plumes.

On page 2-41 of the second draft PA (EPA, 2023), the authors note

Near the tropopause, there is a low bias in the model that is most pronounced in the spring. The low bias at the tropopause likely suggests an underestimate of stratospheric exchange. Mean bias drops to below 20% in the middle troposphere (600-300 hPa). The low-bias in the free troposphere may stem from underestimation of spring time stratospheric contribution in some regions.

On page 2-42 of the second draft PA (EPA, 2023), the authors also note

Dolwick et al., 2015) showed that multi-model estimates converged when applying bias correction, indicating that differences in USB estimates are correlated with model performance. No bias correction has been applied here, so in a limited manner bias in ambient predictions can help set expectations for bias in USB. **Based on hemispheric model evaluation, the stratospheric component in spring is likely underestimated leading to a USB low bias in spring** (emphasis added).

Not applying a bias correction to the estimated USB concentrations when biases in the model were observed may explain why the seasonal patterns exhibited in the model described in the second draft PA (EPA, 2023) did not match the patterns observed when one characterizes ambient data (see previous section). As noted above, the EPA model described in the second draft PA (EPA, 2023) exhibited biases in the springtime. As noted in Tables 3-7 – 3-12 (Supplemental Tables S-1 – S-6 in Lefohn et al., 2012), the spring months are when the stratosphere plays an important role at both low- and high-elevation O₃ surface sites across the U.S. Enhanced, as well as episodic, contributions to ambient O₃ levels occur from the stratosphere during this time. The modeled USB low bias in spring may explain some of the inconsistency observed in the seasonal patterns for USB observed in the EPA results described in the second draft PA (EPA, 2023).

Table 3-7. Months in which highest-elevation (> 2.3 km) monitoring sites in the Intermountain West exhibited a statistically significant coincidence value and there was a coincidence between the number of days with daily maximum hourly average O₃ concentrations ≥ 50 ppb and STT-S >0. The number of days during the specific month when the daily maximum hourly average concentration was ≥ 50 ppb and the STT-S was > 0 is in parenthesis (). Source: Table S-1 from Lefohn et al. (2012).

Site	Months
Yellowstone NP, WY	March 2007 (21), April 2007 (27), May 2007 (27), June 2007 (21), August 2007 (20), March 2008* (23), April 2008 (29), May 2008 (22), June 2008 (19), July 2008 (22), March 2009 (22), April 2009 (23), May 2009 (28), June 2009 (14)
Pinedale, WY	March 2007 (22), April 2007 (28), May 2007 (25), June 2007 (24), August 2007 (26), September 2007 (24), February 2008 (22), March 2008 (29), April 2008 (29), May 2008 (20), June 2008* (17), July 2008 (25), August 2008* (15), September 2008* (13), March 2009 (30), April 2009 (26), May 2009 (23), July 2009 (22)
Centennial, WY	April 2007 (25), May 2007 (20), June 2007 (25), August 2007 (20), September 2007 (24), February 2008 (21), March 2008 (30), April 2008 (30), May 2008 (28), June 2008 (25), July 2008 (24), August 2008 (25), September 2008 (20), March 2009 (29), April 2009 (27), May 2009* (9), June 2009* (17), July 2009 (19), August 2009 (23), September 2009 (24)
Gothic, CO	March 2007 (24), April 2007 (28), May 2007 (27), June 2007 (21), September 2007 (21), February 2008 (22), March 2008 (21), April 2008* (11), May 2008* (22), July 2008 (24), February 2009 (21), March 2009 (28), April 2009 (30), May 2009 (24), June 2009 (20), July 2009 (21), August 2009 (21)
Rocky Mountain NP, CO	March 2007 (23), April 2007 (23), May 2007 (23), June 2007 (20), September 2007 (25), February 2008 (20), March 2008 (29), April 2008 (30), May 2008 (26), June 2008 (27), July 2008 (25), August 2008 (20), September 2008 (19), March 2009 (29), April 2009 (25), May 2009 (24), June 2009 (23), July 2009 (19), August 2009 (20), September 2009 (21), November 2009 (18), December 2009 (23)

*Data capture less than 90% but statistically significant coincidences existed.

Table 3-8. Months in which higher-elevation (1.5 – 2.2 km) monitoring sites in the Intermountain West exhibited a statistically significant coincidence value and there was a coincidence between the number of days with daily maximum hourly average O₃ concentrations ≥ 50 ppb and STT-S >0. The number of days during the specific month when the daily maximum hourly average concentration was ≥ 50 ppb and the STT-S was > 0 is in parenthesis (). Source: Table S-2 from Lefohn et al. (2012).

Site	Months
Mesa Verde NP, CO	March 2007 (23), April 2007 (28), May 2007 (22), September 2007 (19), October 2007 (18), March 2008 (28), April 2008 (29), May 2008 (27), June 2008 (21), September 2008 (14), March 2009 (25), April 2009 (28), May 2009 (24), June 2009 (25), September 2009 (18)
Grand Canyon NP, AZ	February 2007 (13), March 2007 (23), April 2007 (26), May 2007 (27), September 2007 (25), October 2007 (18), February 2008 (20), March 2008 (29), April 2008 (29), May 2008 (28), June 2008 (20), September 2008 (18), March 2009 (21), April 2009 (26), May 2009 (24), June 2009 (26), September 2009 (16)
Canyonlands NP, UT	April 2007 (29), May 2007 (25), June (19), September 2007 (23), March 2008 (28), April 2008 (29), May 2008 (28), June 2008 (25), July 2008 (20), August 2008 (22), September 2008 (21), February 2009 (14), March 2009 (27), April 2009 (28), May 2009 (26), June 2009 (24), July 2009 (21), August 2009 (20), November 2009 (2)
Chiricahua NM, AZ	March 2007 (24), April 2007 (29), May 2007 (19), October 2007 (12), February 2008 (18), March 2008 (27), April 2008 (29), May 2008 (27), October 2008 (16), February 2009 (17), March 2009 (24), April 2009 (30), May 2009 (25)
Great Basin NP, NV	April 2007 (25), May 2007 (31), June 2007 (24), August 2007 (21), September 2007 (23), March 2008 (25), April 2008 (27), May 2008 (26), June 2008 (25), July 2008 (21), August 2008 (21), March 2009 (26), April 2009 (29), May 2009 (26), June 2009* (23), September 2009 (18)
Jefferson County, CO	March 2007 (21), April 2007 (16), May 2007 (21), September 2007 (22), March 2008 (27), April 2008 (29), May 2008 (25), June 2008 (24), July 2008 (20), March 2009 (24), April 2009* (19), May 2009 (20), June 2009 (20), September 2009 (18)

*Data capture less than 90% but statistically significant coincidences existed.

Table 3-9. Months in which high-elevation (> 1.3 km) monitoring sites in the West and East exhibited a statistically significant coincidence value and there was a coincidence between the number of days with daily maximum hourly average O₃ concentrations ≥ 50 ppb and STT-S >0. The number of days during the specific month when the daily maximum hourly average concentration was ≥ 50 ppb and the STT-S was > 0 is in parenthesis (). Source: Table S-3 from Lefohn et al. (2012).

Site	Months
Lassen Volcanic NP, CA	March 2007 (21), April 2007 (23), May 2007 (29), June 2007 (21), July 2007 (19), August 2007 (24), September 2007 (22), March 2008 (29), April 2008 (29), May 2008 (20), June 2008 (26), July 2008 (30), August 2008 (25), September 2008 (24), March 2009 (24), April 2009 (24), May 2009 (23), July 2009 (22), August 2009 (21), September 2009 (19)
Yosemite NP, CA (Turtleback Dome)	April 2007 (27), May 2007 (29), June 2007 (23), August 2007 (22), September 2007 (25), October 2007 (20), March 2008 (27), April 2008 (29), May 2008 (26), June 2008 (24), July 2008* (22), August 2008* (17), September 2008 (26), October 2008 (23), March 2009 (24), April 2009 (27), May 2009 (24), June 2009 (16), July 2009 (28), August 2009 (22), September 2009 (24)
Crestline, CA	March 2007 (26), April 2007 (24), May 2007 (26), October 2007 (20), March 2008 (21), April 2008 (28), May 2008 (29), June 2008 (21), September 2008 (24), October 28 (22), March 2009 (20), April 2009 (27), May 2009 (24), June 2009 (17), September 2009 (19)
Mount Washington, NH	March 2007 (17), May 2007 (20), September 2007 (15), March 2008 (19), April 2008 (23), May 2008 (23), June 2008 (16), March 2009 (25), April 2009 (21), May 2009 (24), September 2009 (15), November* 2009 (10)
Whiteface Mountain, NY	March 2007 (17), May 2007 (21), March 2008 (20), April 2008 (22), May 2008 (20), April 2009 (16), May 2009 (22), September 2009 (9)

*Data capture less than 90% but statistically significant coincidences existed.

Table 3-10. Months in which low-elevation (< 1.3 km) monitoring sites in the West and Intermountain West exhibited a statistically significant coincidence value and there was a coincidence between the number of days with daily maximum hourly average O₃ concentrations ≥ 50 ppb and STT-S >0. The number of days during the specific month when the daily maximum hourly average concentration was ≥ 50 ppb and the STT-S was > 0 is in parenthesis (). Source: Table S-4 from Lefohn et al. (2012).

Site	Months
Cheeka Peak, WA	May 2007 (7), August 2007 (1), June 2009 (3), September 2009 (1)
King County, WA	January 2007 (1), August 2008 (1), July 2009 (1)
Mount Rainier NP, WA	August 2008 (4), July 2009 (9)
Trinidad Head, CA	June 2008 (2)
El Dorado County, CA	May 2007* (11), September 2007 (18), March 2008 (23), April 2008 (28), May 2008 (23), June 2008 (21), October (15), April 2009 (22), June 2009 (17), September 2009 (20)
Glacier NP, MT	April 2008 (18), April 2009 (18)
Big Bend NP, TX	February 2007 (13), March 2007 (25), April 2007 (22), September 2007 (6), October 2007 (11), March 2008 (25), April 2008 (28), August 2008 (5), September 2008 (6), February 2009 (17), March 2009 (16), April 2009 (28), June 2009 (7)

*Data capture less than 90% but statistically significant coincidences existed.

Table 3-11. Months in which low-elevation (< 1.3 km) monitoring sites in the Midwest exhibited a statistically significant coincidence value and there was a coincidence between the number of days with daily maximum hourly average O₃ concentrations ≥ 50 ppb and STT-S >0. The number of days during the specific month when the daily maximum hourly average concentration was ≥ 50 ppb and the STT-S was > 0 is in parenthesis (). Source: Table S-5 from Lefohn et al. (2012).

Site	Months
Theodore Roosevelt NP, ND	April 2007 (16), May 2007 (15), April 2008 (22), May 2008 (20), May 2009 (17)
Voyageurs NP, MN	April 2007 (14), September 2007 (4), April 2008 (18), April 2009 (19), May 2009 (18)
Ann Arbor, MI	April 2007 (15), May 2007 (21), June 2007 (17), April 2008 (20), May 2008 (22), July 2008 (16), May 2009 (23), June 2009 (18)
Cook County, IL	June 2007 (14), April 2008 (15), June 2008 (15), April 2009 (14), May 2009 (17), July 2009 (13)
Alhambra, IL	April 2007 (22), June 2007 (16), July 2007 (17), April 2008 (19), May 2008 (22), June 2008 (17), April 2009 (14), May 2009 (16), September 2009 (8)
Stockton, IL	August 2009* (1)
Harris County, TX	March 2007 (6), May 2007 (9), June 2007 (1), April 2008 (12), May 2008 (10), June 2008 (1), July 2008 (3), September 2008* (4), October 2008 (16), March 2009 (11), April 2009 (13), May 2009 (5), June 2009 (4), July 2009 (1), August 2009 (6), September 2009 (4)

*Data capture less than 90% but statistically significant coincidences existed.

Table 3-12. Months in which low-elevation (< 1.3 km) monitoring sites in the East exhibited a statistically significant coincidence value and there was a coincidence between the number of days with daily maximum hourly average O₃ concentrations ≥ 50 ppb and STT-S > 0. The number of days during the specific month when the daily maximum hourly average concentration was ≥ 50 ppb and the STT-S was > 0 is in parenthesis (). Source: Table S-6 from Lefohn et al. (2012).

Site	Months
Georgia Station, GA	March 2007 (22), April 2007 (23), May 2007 (20), June 2007 (19), July 2007 (8), September 2007 (13), March 2008 (23), April 2008 (20), May 2008 (23), July 2008 (15), March 2009 (14), April 2009 (18), May 2009 (6)
Rockdale, GA	March 2007 (21), April 2007 (23), May 2007* (17), March 2008 (21), April 2008 (18), May 2008 (25), September 2008 (8), May 2009 (8), July 2009 (14)
Cuyahoga County, OH	April 2008 (19), May 2008 (20), May 2009 (16) July 2009 (15)
Bucks County, PA	May 2007 (19), June 2007 (17), July 2007* (13), May 2008 (16), April 2009 (15), May 2009 (16), June 2009 (10)
Shenandoah NP, VA	April 2007 (19), May 2007 (22), June 2007* (18), September 2007 (21), April 2008 (22), May 2008 (27), June 2008 (18), July 2008 (18), August 2008 (20), September 2008 (10), March 2009 (16), April 2009 (21), May 2009 (13), June 2009 (14), August 2009 (10)
Blackwater NWR, MD	September 2008 (10), May 2009 (10)
Abington, CT	July 2008 (14), April 2009 (14)
Fairfield, CT	April 2009 (15), June 2009 (11)
Chittenden County, VT	April 2008 (19), May 2008 (17), April 2009 (18), May 2009 (17)

*Data capture less than 90% but statistically significant coincidences existed.

Jaffe et al. (2018) discussed a site in Colorado, where simulations and contributions were compared for a monitor at Chatfield (AQS 08-035-0004) for May-September 2011. The authors described the site as a regulatory relevant suburban monitor southwest of Denver, Colorado. Fig. 6 in Jaffe et al. (2018) showed the observed and modeled daily MDA8 values using results from an EPA model. Monthly averaged biases at the Chatfield monitor were marginally negative in the EPA simulations. The authors noted that Fig. 6 in Jaffe et al. (2018) suggested four distinct segments of performance and simulated contributions at the Chatfield monitor that were related to contributions from noncontrollable O₃ sources. As pointed out by the authors, the simulations started in a USB O₃ dominated regime (May 1 to June 7), then went through a transition period (June 8 to July 15), and then ended with two periods dominated by local contributions (July 16 to August 22 and August 23 to September 29). During the USB O₃ dominated period, the EPA model experienced a mean bias of -2.1. During the transition period, the EPA simulations performed poorly and experienced a mean bias of -3.6. During the locally dominated period of July 16 to August 22, the simulation performed well with a mean bias of -0.9. During the locally dominated period of August 23 to September 29, the mean bias was -3.3. Jaffe et al. (2018) noted that the simulations performed better during periods of sustained contribution (USB O₃ or local). The authors noted that simulations performed even better when USB O₃ and local contribution were not anti-correlated, and simulations performed best when local contributions were dominant. The model performed well for average biases, but model correlation with observations was better when local contributions were dominant and when anti-correlation between local and USB O₃ contributions was weak.

Fig. 3-58 illustrates for the Chatfield site in 2011 the comparison of observed daily O₃ MDA8 concentrations and the STT-S counts (estimated for a site in the Denver area for 2011). The USB O₃ dominated regime identified in Jaffe et al. (2018) (May 1 to June 7) was a period of large numbers of STT-S counts, the transition period (June 8 to July 15) exhibited a declining number of STT-S counts, the period July 16 to August 22 experienced fewer STT-S counts, and the period August 23 to September 29 exhibited an increasing pattern of STT-S counts. The period from early July to early September was the time when the lowest number of STT-S counts occurred over the 2011 period. Based on the mean monthly bias observed by Jaffe et al. (2018) for the high-elevation site at Chatfield, the periods associated with greatest biases in the EPA model appeared to be associated with the period when the contribution of STT-S was most important. The lowest mean bias was -0.9, which occurred during July 16 to August 22, a period when the STT-S counts were the lowest.

The second draft PA (EPA, 2023, page 2-42) indicates, based on hemispheric model evaluation, that the stratospheric component in spring was likely underestimated leading to a USB low bias in spring. The authors noted that no bias correction had been applied to the EPA model used in the second draft PA (EPA, 2023). The second draft PA (EPA, 2023) observed that the Natural contribution has two peaks in the East. The largest occurred in late Spring and the second peak occurred in early Fall (EPA, 2023, page 2-48). It is interesting to note that the pattern described in the second draft PA (EPA, 2023) for the East, was observed in both the East and West by Lefohn et al. (2012). Lefohn et al. (2012) noted that for the 39 sites analyzed in their study, many of the sites during the spring, fall, and winter months experienced higher GBO₃, which was associated with more frequent stratosphere-to-troposphere transport to the surface (STT-S) enhancements according to their independent three-dimensional trajectories

based on global meteorological analyses. Patterns of higher spring Emissions Influenced Background (EIB) O₃ were followed by lower values during the summer, due to heightened chemical interaction with anthropogenic sources, which were then followed by rising EIB O₃ during the fall and winter months. For some high-elevation western U.S. sites, the seasonal pattern was less discernible due to relatively small anthropogenic contributions and the high EIB O₃ estimated throughout the year. EIB O₃ at all high-elevation sites contributed a significant proportion to total O₃ throughout the year and throughout the observed total O₃ frequency distribution, while EIB O₃ at most urban sites contributed a major portion to total O₃ during non-summer months and to the mid-range concentrations (30-50 ppb) of the frequency distribution. The different patterns in the West noted in EPA's USB model described in the second draft PA (EPA, 2023) and Lefohn et al. (2012) may be attributable to the lack of bias adjustment in the EPA model described in the second draft PA (EPA, 2023). One might hypothesize that if a bias adjustment were performed on the daily MDA8 predictions in the EPA model described in the second draft PA (EPA, 2023) that higher MDA8 values might have occurred during the spring and fall periods. If EPA had performed a bias adjustment, perhaps the EPA's USB modeling results might have agreed better with the seasonal USB patterns described in the Agency's 2014 PA (EPA, 2014a), as well as past publications in the peer-reviewed literature.

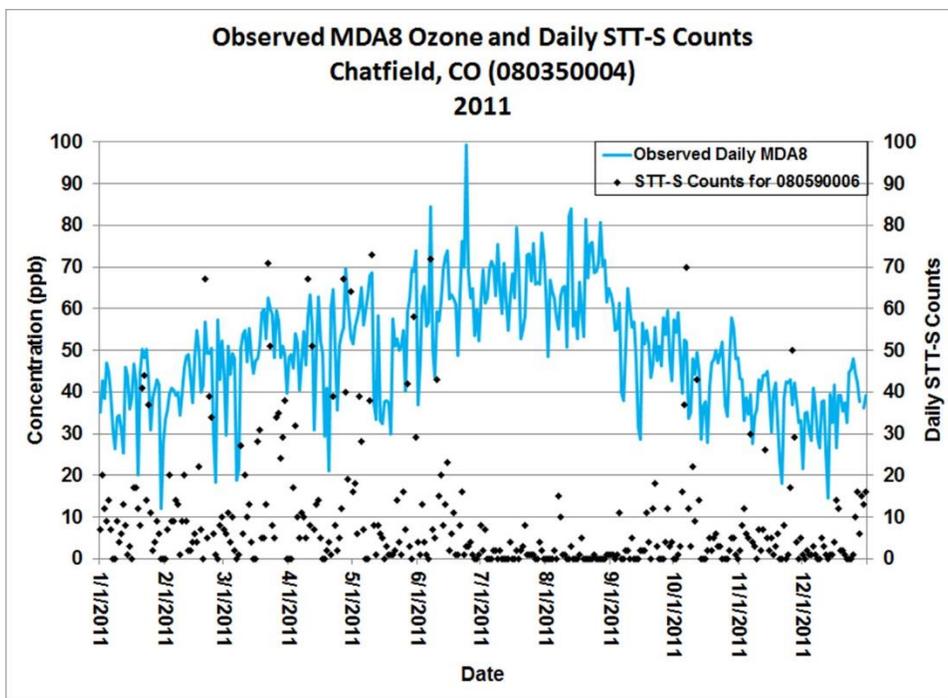


Figure 3-58. Observed MDA8 O₃ and daily STT-S counts for 2011 for the Chatfield (CO) (080350004) monitoring site. The STT-S counts, which were quantified for a site in Jefferson County (CO) (080590006), were superimposed over the observed data at the Chatfield site.

For the eight sites used in the risk assessment (Atlanta, Boston, Dallas, Detroit, Philadelphia, Phoenix, Sacramento, and St. Louis), the second draft PA (EPA, 2023) does not provide examples for specific sites that describe the observed and USB concentrations. In the

second draft PA (EPA, 2023), analyses are presented that estimate exposure and risk for simulated populations in eight study areas. The eight study areas represent a variety of circumstances about population exposure to short-term concentrations of O₃ in ambient air. The eight study areas range in total population size from approximately two to eight million and are distributed across the U.S. in seven different NOAA climate regions: The Northeast, Southeast, Central, East North Central, South, Southwest and West. In Figs. 3-59 through 3-65, total observed O₃ concentrations, USB_{AB} estimates (USB_{AB} data provided by the EPA), and STT-S counts are presented for 2007 for seven of the eight sites (i.e., Atlanta, Boston, Dallas, Detroit, Philadelphia, Sacramento, and St. Louis) used by the EPA in its risk analyses presented in the second draft PA (EPA, 2023).

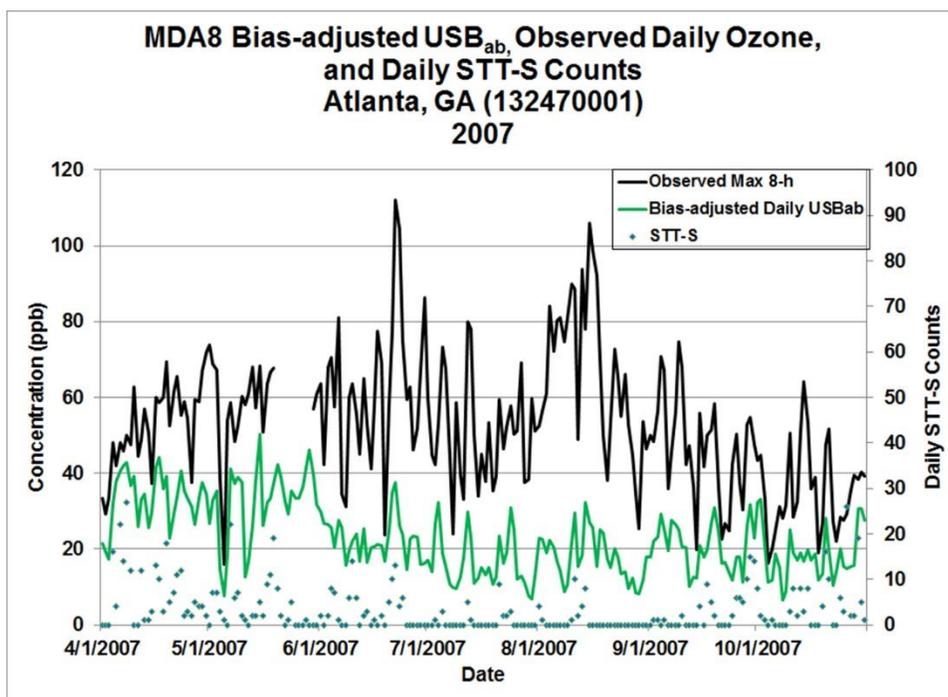


Figure 3-59. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Atlanta, Georgia (AQS ID 132470001) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

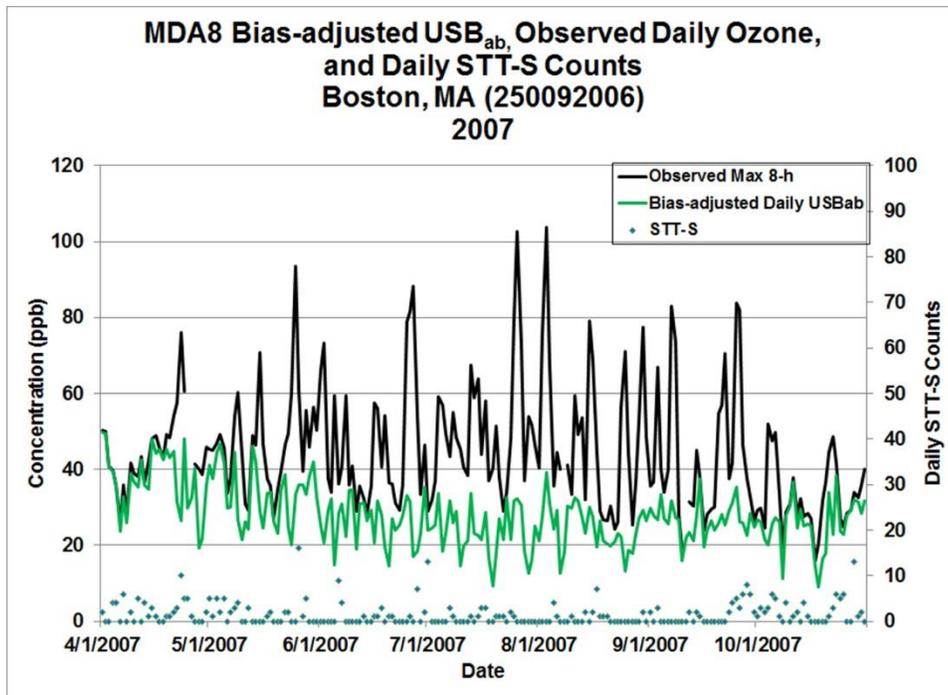


Figure 3-60. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Boston, Massachusetts (AQS ID 250092006) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

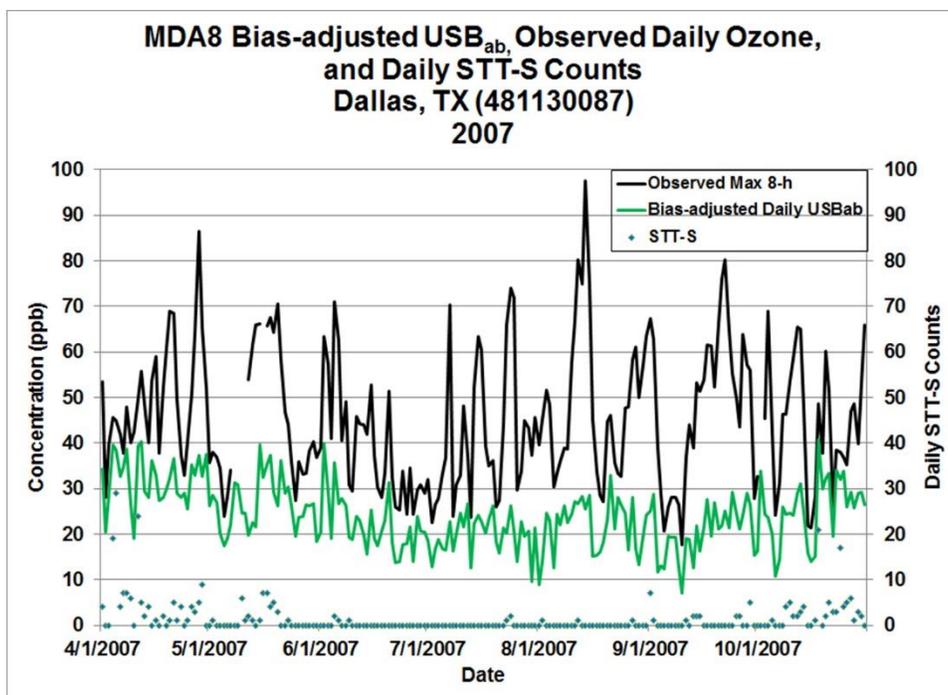


Figure 3-61. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Dallas, Texas (AQS ID 481130087) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

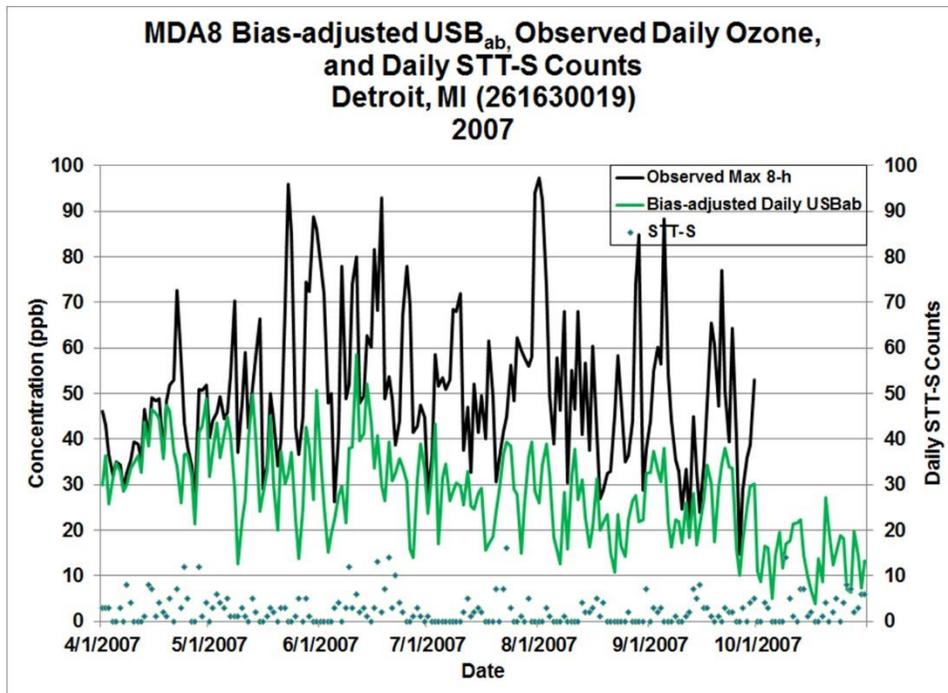


Figure 3-62. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Detroit, Michigan (AQS ID 261630019) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

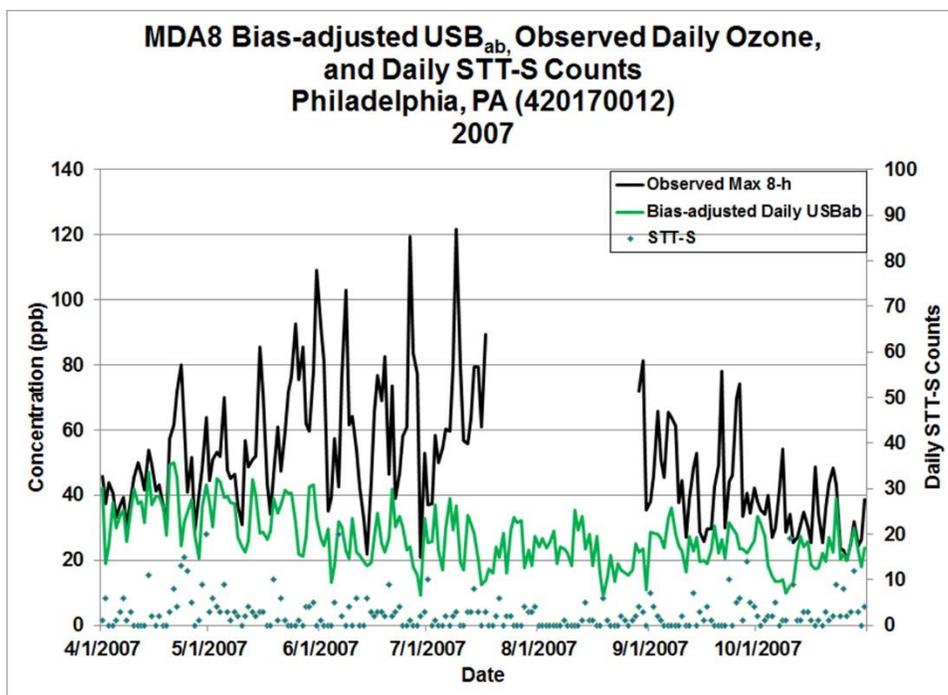


Figure 3-63. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Philadelphia, Pennsylvania (AQS ID 420170012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

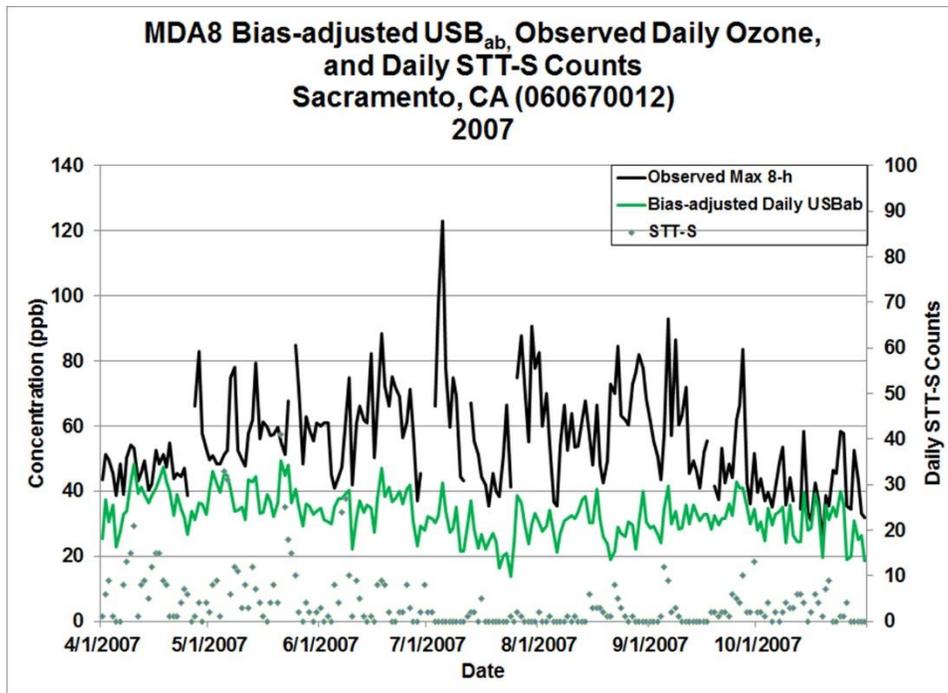


Figure 3-64. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in Sacramento, California (AQS ID 060670012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

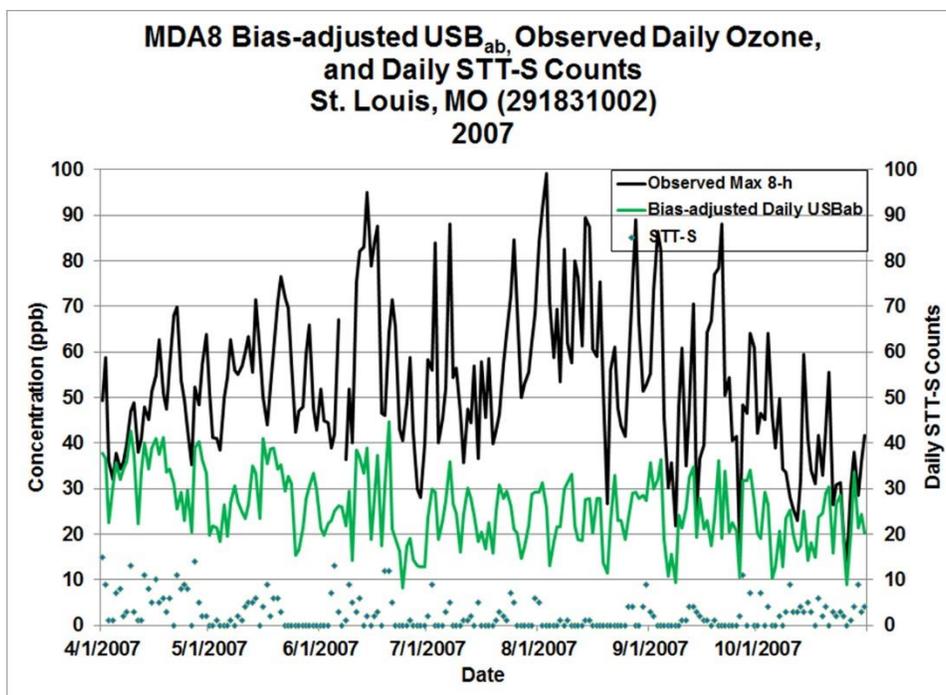


Figure 3-65. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background (USB_{AB}) 8-h daily maximum concentrations for a site in St. Louis, Missouri (AQS ID 291831002) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily O₃ values. Daily USB_{AB} 2007 values provided by the EPA. See Lefohn et al. (2011, 2012, 2014) for details of how the STT-S values are estimated.

For many of the sites (Figs. 3-59 - 3-65), higher spring USB_{AB} O₃ was followed by lower values during the summer, which was then followed by rising USB_{AB} O₃ during the fall months. Focusing on the difference between the observed and USB_{AB} concentrations (i.e., **the Gaps**), the pattern varies by site. The Gaps indicate the apparent influence of anthropogenic sources on each site. The pattern of the difference between the observed and USB_{AB} concentrations (i.e., gaps) indicate, for many of the seven sites, the variability of USB_{AB} during specific periods. Enhanced O₃ levels associated with stratospheric contribution occur across the seven sites with the result that USB_{AB} contributes to varying amounts (i.e., depending upon season and location of the site) to the total observed O₃ concentrations across the U.S.

3.2.10 Model Performance USB versus USB_{AB}

In a perfect world when models perform well, the estimates of USB background O₃ should be higher than USB_{AB}. This is because the USB estimates are defined as the O₃ concentration that would occur if all U.S. anthropogenic O₃ precursor emissions were removed (ISA, 2020a, Page ES-3). While USB is an estimate of O₃ concentrations that could be achieved if all U.S. anthropogenic sources were eliminated, USB_{AB} is an estimate of how much O₃ can be

attributed to background sources when those anthropogenic sources are still present. As noted in the second draft PA (EPA, 2023, page 2-38)

This analysis follows the zero-out approach for simplicity of interpretation and consistency with previous EPA analyses. In urban areas, this approach will estimate higher natural and USB contributions than total O₃ when NO_x titration is present. The estimate, therefore, is an estimate of what concentrations could be without U.S. anthropogenic emissions and not the fraction of observed O₃ that is USB.

Background O₃ is titrated by the NO associated with NO_x sources. Thus, it would be anticipated that USB estimated concentrations would generally be higher than the estimated USB_{AB} values, which represent titrated background O₃. Fig. 3-66 illustrates the difference between the estimates of USB and USB_{AB}, before and after a bias adjustment is made (Dolwick et al., 2015). Prior to the application of the bias-adjustment, there was a clear tendency for the CMAQ zero-out modeling to estimate higher levels of USB MDA8 O₃ concentrations compared to source apportionment USB_{AB} over most sites in the western U.S. As noted by the authors, constraining the USB and USB_{AB} estimates for model bias brings the estimates across the two methodologies closer together.

Fig. 3-67 illustrates the bias-adjusted model estimates for USB and USB_{AB} from the CMAQ and CAMx models, respectively (Dolwick et al., 2015). The spatial patterns for the April-October mean bias-adjusted MDA8 values between the two sets of estimates are similar. However, differences exist as one compares the site estimates using the two models. The differences in the estimates described in Dolwick et al. (2015) were based on 7-month mean bias-adjusted MDA8 values. If one examined the daily time series for USB and USB_{AB} MDA8 values, the differences in the estimates for USB and USB_{AB} would be greater. These daily differences are not quantified here because a comparison of daily USB and USB_{AB} values was not available to assess.

Dolwick et al. (2015) noted that because the two distinct model approaches estimated similar background impacts over the rural portions of the western U.S., the authors believed greater confidence could be placed on the combined results. However, the authors noted that while the CAMx and CMAQ model simulations provided consistent estimates in their study of rural USB O₃ levels in the western U.S., the CAMx source-apportionment approach (i.e., USB_{AB}) predicted lower background contributions in the urban areas than USB, as anticipated, because anthropogenic emissions reacted with and destroyed some fraction of the O₃ in the CAMx tracer species used to track the background O₃ contribution.

As noted in earlier sections, the USB modeling results described in the second draft PA (EPA, 2023) indicate the following seasonal patterns: (1) natural and USA O₃ contributions peak during the traditional O₃ season (May through September), while long-range intercontinental transport of international O₃ (i.e. contributions from China, India etc.) peaks in the spring (February through May) (page 2-64); (2) the natural contribution has a single maximum in late summer in the West, whereas, in the East there is evidence of two peaks—the largest in late Spring and a second peak in early Fall (page 2-48). The previous conclusion in the 2014 PA

(EPA, 2014a) and the 2013 ISA (EPA, 2013, in section 3.4) was that background O₃ was greatest over the U.S. during the spring and early summer (i.e., March-June period). While the estimates of the absolute values for background O₃ might differ depending upon the choice of whether USB or USB_{AB} was used in the modeling effort, I do not believe that the choice of USB or USB_{AB} explains the differences in the seasonal patterns associated with the background O₃ modeling results in the second draft PA (EPA, 2023) from previous reported patterns for background O₃.

As noted earlier, Dolwick et al. (2015) and Lefohn et al. (2014) used bias-adjusted estimates for USB_{AB} and Emissions Influenced Background (EIB), respectively. The different pattern in the West noted in the second draft PA (EPA, 2023) compared to the pattern reported by Dolwick et al. (2015) and Lefohn et al. (2014) may be attributable to the lack of bias adjustment in the current EPA USB model described in the second draft PA (EPA, 2023). The second draft PA (EPA, 2023) notes that bias adjustment was not performed in the modeling effort. In their analyses, Lefohn et al. (2014) noted that model performance at low-elevation sites tended toward larger under prediction biases in the cool months (i.e., November-April) and larger over prediction biases in warm months (June-October), particularly for sites in the southern and eastern U.S. For almost all high-elevation sites in their analyses, the model under predicted in the spring months when the above-median MDA8 Global Background O₃ coincided with diagnosed STT-S events as per the stratospheric trajectory analyses performed by Dr. Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich in Switzerland. For urban sites with higher spring anthropogenic influences, it was more likely that over predictions were associated with anthropogenic O₃. There continues to be strong evidence, as supported in the literature, as well as EPA's own analyses (e.g., EPA, 2014a), that background O₃ across the U.S. is highest at many sites during the springtime (including into the month of June) and background O₃ is an important contributor at many high-elevation sites throughout the year.

In summary, empirical data indicate that as emission reductions occur across the U.S., the higher MDA8 concentrations shift at many O₃ monitoring sites from the summer toward the March-June months. In addition, as emission reductions occur, the distribution of hourly average concentrations shift from the higher values toward the middle values and the lower values shift upward toward the middle values. There is a compression of the distribution. Further, as emission reductions occur, background O₃ concentrations increase their percentage in the observed total O₃ concentration with the result that the compressed distribution of hourly average concentrations begins to resemble the distribution of background O₃. Hopefully, if one were to apply a bias adjustment to the estimated modeled USB estimates described in the second draft PA (EPA, 2023), patterns would result that resemble the background O₃ patterns published previously in the literature, as well as the patterns observed in the empirical data as emission reductions have occurred. If bias adjustments to the model do not change the seasonal patterns described for the current EPA USB model described in the second draft PA (EPA, 2023), then it is recommended that further model sensitivity analyses be undertaken. The patterns described earlier derived from empirical data (i.e., the compressed distributions and the seasonal shift from the summer months to the March-June period) provide those of us who have applied models to estimate USB, USB_{AB}, or Emissions Influenced Background (EIB) with the opportunity to assess the adequacy of our results. As mentioned earlier, background O₃ plays an important role in the Agency's risk analyses. In turn, the risk analyses play an important role in the Administrator's

margin of safety determination for the human health O₃ standard. The margin of safety determinations play an important role in the Administrator’s final decision on the O₃ NAAQS.

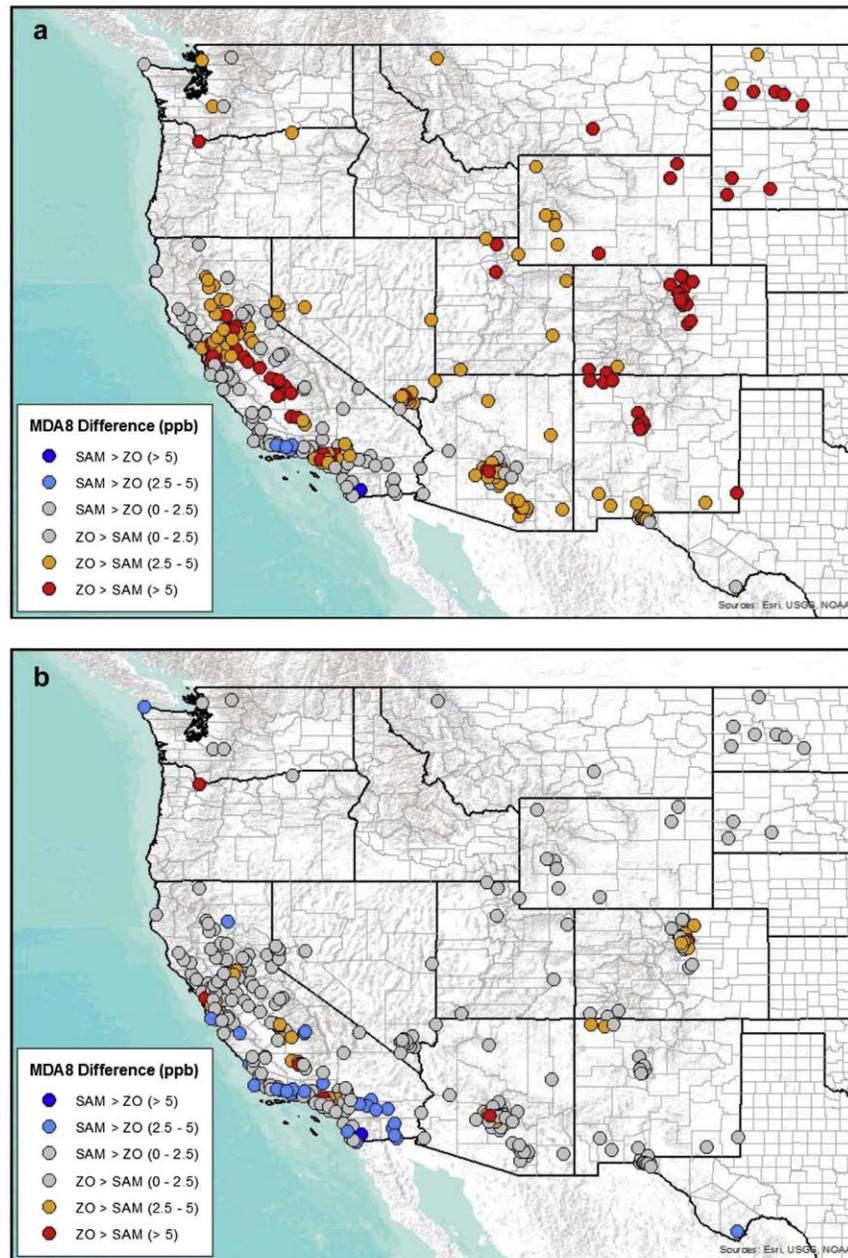


Fig. 5. Difference (ppb) in unadjusted April–October mean MDA8 USB ozone vs mean MDA8 USB_{AB} ozone at monitoring locations across the western U.S. Brighter colors indicate sites where zero out (ZO) estimates of USB exceed source apportionment (SAM) estimates of USB_{AB}. b. Difference (ppb) in bias-adjusted April–October mean MDA8 USB ozone vs. bias-adjusted mean MDA8 USB_{AB} ozone at monitoring locations across the western U.S. Brighter colors indicate sites where zero out (ZO) estimates of USB exceed source apportionment (SAM) estimates of USB_{AB}. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Figure 3-66. Difference between the estimates of USB and USB_{AB}, before and after a bias adjustment is made. Source: Dolwick et al. (2015).

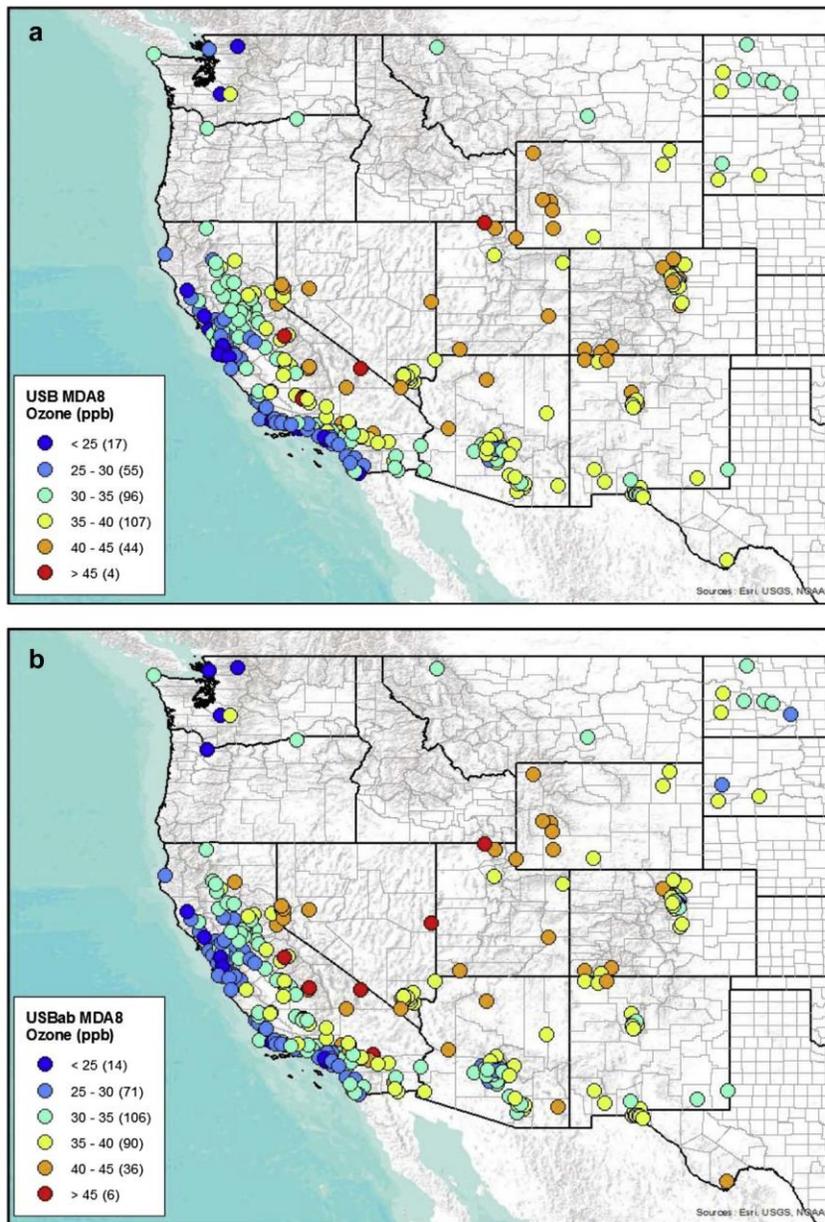


Fig. 6. April–October mean bias-adjusted USB MDA8 ozone (ppb) at monitoring locations across the western U.S., as estimated by a 2007 CMAQ zero out simulation. b. April–October mean bias-adjusted USB_{AB} MDA8 ozone (ppb) at monitoring locations across the western U.S., as estimated by a 2007 CAMx source apportionment simulation.

Figure 3-67. April-October mean bias-adjusted USB MDA8 O₃ (ppb) at monitoring locations across the western U.S., as estimated by a 2007 CMAQ zero-out simulation. b. April-October mean bias-adjusted USB_{AB} MDA8 O₃ (ppb) at monitoring locations across the western U.S., as estimated by a 2007 CAMx source-apportionment simulation. Source: Dolwick et al. (2015).

3.2.11 Background O₃ and the W126 Exposure Metric

The Lapina et al. (2014) analysis is summarized in the second draft PA (EPA, 2023, pages 2-32, 2-34, 2-35, 2-66) for discussing the contribution for background O₃ to W126 cumulative exposures. Lapina et al. (2014), using three regional or global chemical transport models, quantified the W126 exposure index in the U.S. in the absence of North American anthropogenic emissions (North American background or “NAB”). The investigators noted that the season with the highest observed O₃ concentrations depends on a specific location. According to the authors, this made modeling the maximum 3-month W126 cumulative value in the continental U.S. computationally expensive. To avoid this expense, Lapina et al. (2014) focused on a fixed 3-month period, May–July 2010 for their analyses. However, as noted in previous sections, the maximum 3-month W126 exposures occur during the March-June period in many vegetation areas across the U.S. To better quantify the importance of background O₃ in influencing the W126 cumulative exposures, the estimates performed in the 2014 PA (EPA, 2014a) may provide more insight on the importance of background O₃ influencing the W126 values than the analyses performed by Lapina et al. (2014).

In the 2014 PA (EPA, 2014a), the EPA used the 2007 zero-out modeling to assess NB (i.e., natural background), NAB (North American Background), and USB influences at four sample locations: Atlanta GA, Denver CO, Farmington NM, and Riverside CA. Each of the four analyses locations had relatively high observed values of W126 in 2007, as averaged over all sites in the area: Atlanta (25.1 ppm-hrs), Denver (19.6 ppm-hrs), Farmington (20.2 ppm-hrs), and Riverside (36.0 ppm-hrs). EPA considered the fractional influence of background O₃ on annual W126 levels in four locations. Fig. 3-68 (originally Fig.2-16 in the 2014 PA) shows the results. Based on the fractional influence methodology, natural background sources were estimated to contribute 29-50% of the total modeled W126, with the highest relative influence in the intermountain western U.S. (i.e., Farmington, NM) and the lowest relative influence in the eastern U.S. (i.e., Atlanta). U.S. background (USB) was estimated to contribute 37-65% of the total modeled W126. As noted in the 2014 PA (EPA, 2014a), the proportional impacts of background were slightly less for the W126 metric than for seasonal mean MDA8 (discussed in section 2.4.2 of the 2014 PA), because of the sigmoidal weighting function that places more emphasis on higher O₃ days when background fractions were generally lower. The key conclusion from the EPA’s cursory analysis summarized in the 2014 PA (EPA, 2014a) was that background O₃ could comprise a non-negligible portion of current W126 levels across the U.S. These fractional influences were greatest in the intermountain western U.S. and were slightly smaller than when the seasonal mean MDA8 metric was used in the analysis.

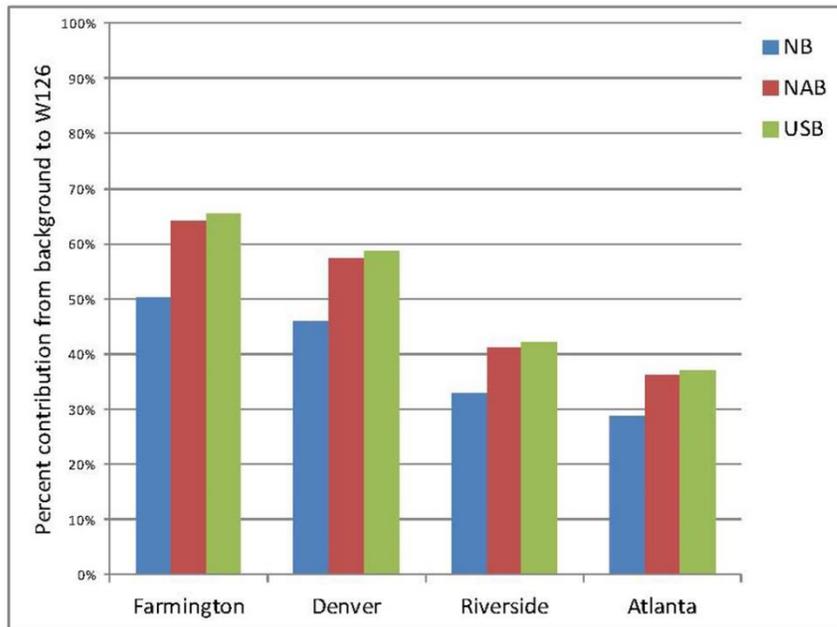


Figure 2-16. Fractional influence of background sources to W126 levels in four sample locations. Model estimates based on 2007 CMAQ zero-out modeling.

Figure 3-68. Fractional influence of background sources to W126 levels in four sample locations. Model estimates based on 2007 CMAQ zero-out modeling. Source: EPA (2014a).

3.2.12 Empirical Evidence Supporting the Relationship between Reductions of Ozone Precursors and Changes in High and Low Ozone Concentrations – COVID-19 Worldwide Lockdown Emission Reductions

Earlier, it was indicated that as emissions are reduced, the highest hourly average O₃ concentrations are *reduced*, and at many sites the lowest concentrations are *increased* (due to less NO titration of O₃ as NO_x emissions are reduced). Simon et al. (2015) and Sicard et al. (2016) reported more ambient increases in low O₃ concentrations during winter, the time of year where less NO titration of O₃ is most likely to be favored, than during the summer months when O₃ production efficiency is high (Lefohn et al., 2017). This phenomenon has been observed at many locations across the U.S. (Simon et al., 2015; Lefohn et al., 2017, 2018; EPA, 2020b). The same phenomenon has also been observed at sites outside of the U.S. (Lefohn et al., 2018).

Because of the unfortunate COVID-19 pandemic, many countries around the world during the spring (Northern Hemisphere) and fall (Southern Hemisphere) of 2020 instituted immediate lockdown orders. As a result of these orders, anthropogenic emissions were severely reduced. Recognizing the opportunity to investigate how severe emission reductions influenced air pollution concentrations, researchers documented these changes. One important result of the observations documented over the short period in 2020, as emissions were reduced, was additional confirmation about the changes that occurred in the high concentration part of the distribution, as well as the low part of the distribution. These observations are important because

they provide insight concerning changes in O₃ distribution patterns as emissions are reduced to attain current and future NAAQS standards to protect human health and welfare.

In earlier sections, the two fundamental principles were described. The first fundamental principle (i.e., **Higher Hourly Average O₃ Concentrations Should be Weighted More than Middle and Lower Values when Assessing Human Health and Environmental Effects**) focuses on the higher hourly average concentrations within the distribution. The second fundamental principle (i.e., **Daily Maximum Hourly Averaged O₃ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide**) focuses on the changes in the entire distribution of hourly average concentrations as emissions are reduced. During the 2020 COVID-19 lockdown, scientists characterized changes in the O₃ levels using combinations of exposure metrics. Some of the investigators used metrics that focused on the lower end of the distribution, while other researchers applied metrics that focused on the upper end. In some cases, the investigators reported *increasing* O₃ during lockdown, while other researchers reported *decreasing* O₃. As will be discussed below, there are possible reasons responsible for explaining the increases and decreases in O₃ concentrations during lockdown.

Sicard et al. (2020) reported on the effect of lockdown due to the COVID-19 pandemic on air pollution in four Southern European cities (Nice, Rome, Valencia, and Turin) and Wuhan, China. The focus of their study was on O₃. Compared to the same period in 2017-2019, the daily O₃ mean concentrations increased at urban stations by 24% in Nice, 14% in Rome, 27% in Turin, 2.4% in Valencia, and 36% in Wuhan during the lockdown in 2020. According to Sicard et al. (2020), the increase in O₃ concentrations was mainly explained by an unprecedented reduction in NO_x emissions leading to a lower O₃ titration by NO.

Huang et al. (2020) reported the results of imposed nationwide lockdown restrictions in China after the Chinese New Year in January of 2020. The authors reported that despite large decreases in primary pollution, there were several periods of heavy haze pollution in East China during the COVID-19 lockdown, raising questions about the well-established relationship between human activities and air quality. The authors calculated differences in averaged concentrations of NO₂, PM_{2.5}, O₃, and the PM_{2.5}/CO ratio prior to lockdown (January 2 – January 23, 2020) and during lockdown (January 26 – February 17, 2020). Huang et al. (2020) reported that large decreases in NO_x emissions occurred during lockdown from transportation with increases in average O₃ concentrations and nighttime NO₃ radical formation, and these increases in atmospheric oxidizing capacity in turn facilitated the formation of secondary inorganic and organic particulate matter.

Le et al. (2020) reported for eastern China that up to 90% reduction of certain emissions during the city-lockdown period were identified from satellite and ground-based observations. The primary focus period during the COVID-19 lockdown in China was from January 23 to February 13, 2020. This period encompassed a 7-day national holiday traditionally celebrating the Lunar New Year, during which previous studies have noted the reduction in anthropogenic emissions. Unexpectedly, extreme particulate matter levels simultaneously occurred in northern China. The author's synergistic observation analyses and model simulations showed that anomalously high humidity promoted aerosol heterogeneous chemistry, along with stagnant

airflow and uninterrupted emissions from power plants and petrochemical facilities, contributing to severe haze formation. In addition, Le et al. (2020) noted that because of non-linear production chemistry and titration of O₃ in winter, reduced nitrogen oxides resulted in O₃ enhancement of 3-week average concentrations in urban areas, further increasing the atmospheric oxidizing capacity and facilitating secondary aerosol formation.

Mahato et al. (2020), described the results of a nationwide lockdown in Delhi, India initially from March 24 to April 14, 2020 and extended up to May 3, 2020. With the aid of air quality data of seven pollutant parameters (PM₁₀, PM_{2.5}, SO₂, NO₂, CO, O₃, and NH₃) for 34 monitoring stations spread over the megacity, the authors evaluated the spatial pattern of air quality in pre and during-lockdown phases. The results demonstrated that during lockdown air quality was significantly improved. Among the selected pollutants, concentrations of PM₁₀ and PM_{2.5} were reduced (>50%) in comparison to the pre-lockdown phase. In comparison to the previous year (i.e., 2019) during the same time the reduction of PM₁₀ and PM_{2.5} was as high as about 60% and 39%, respectively. Among other pollutants, NO₂ (-52.68%) and CO (-30.35%) levels were also reduced during the lockdown phase. About 40% to 50% improvement in air quality was identified just after four days of commencing lockdown. Using the average of the daily 8-h average concentration across sites, the authors noted that the concentration of O₃ increased in the industrial and transport dominated locations (>10% increase). Mahato et al. (2020) noted that the cause for the increase in O₃ concentration, especially in the industrial and transport dominated areas, was the decrease of NO, which led to the lowering of the O₃ consumption (titration, NO + O₃ = NO₂ + O₂) and caused an increase in O₃ concentrations.

Dantas et al. (2020) described the impact of COVID-19 partial lockdown during the fall period on the air quality of the city of Rio de Janeiro, Brazil. On March 16, 2020, the state's governor declared a public health emergency in the city of Rio de Janeiro and partial lockdown measures came into force a week later. The pre-lockdown period was March 2 – 22, 2020 and lockdown was from March 23 – April 16. The authors compared the particulate matter, carbon monoxide, nitrogen dioxide, and O₃ concentrations determined during the partial lockdown with values obtained in the same period of 2019 and with the weeks prior to the virus outbreak. For the comparison of the results obtained in different days, median concentrations of hourly average values were used in their analyses. The concentrations varied with substantial differences among pollutants and among the three studied monitoring stations. According to the authors, CO levels showed the most significant reductions (30.3–48.5%) since they were related to light-duty vehicular emissions. The authors noted that NO₂ showed reductions, while PM₁₀ levels were only reduced during the first lockdown week. Dantas et al. (2020) reported that in April, an increase in vehicular flux and movement of people was observed mainly because of the lack of consensus about the importance and need of social distancing and lockdown. The authors noted that O₃ concentrations increased probably due to the decrease in the nitrogen oxides level. When compared to the same period of 2019, NO₂ and CO median values were 24.1–32.9 and 37.0–43.6% lower. The authors cautioned that meteorological interferences, mainly the transport of pollutants from the industrial areas, might have influenced the results.

Patel et al. (2020) presented a case study from Auckland, New Zealand, an isolated Southern Hemisphere city, which is largely unaffected by long-range pollution transport or industrial sources of air pollution. In this city, traffic flows reduced by 60–80% because of a

government-led initiative to contain the virus by limiting all transport to only essential services. The authors characterized changes in ambient pollutant concentrations of NO₂, O₃, BC, PM_{2.5}, and PM₁₀ between the lockdown period (March 27 – April 17, 2020), with data from this period compared with data for a similar time of year (February to April) extending back to 2015 for assessing consideration of the impacts of the local meteorology on air pollution levels at the same time of year. Using 24-hour average air pollution concentrations, statistically significant changes were observed before and after lockdown at two sites for all pollutants evaluated, based on t-tests. Patel et al. (2020) showed that the source emission reductions had significant but non-linear impacts on air quality. While emission inventories and receptor modelling approaches confirm the dominance of traffic sources for NO_x (86%), and BC (72%) across the city, observations suggest a consequent reduction in NO₂ of only 34–57% and a reduction in BC of 55–75%. While these two pollutants were reduced, O₃ concentrations increased. The authors noted that a lesser increase in O₃ would be anticipated due to the already low background levels of O₃ observed in the Southern Hemisphere. The observed reductions in PM_{2.5} (still likely to be dominated by traffic emissions), and PM₁₀ (dominated by sea salt, traffic emissions to a lesser extent, and affected by seasonality) were found to be significantly less (8–17% for PM_{2.5} and 7–20% for PM₁₀).

In the U.S. during the lockdown period, substantial anthropogenic emissions were reduced. To better understand the spatial extent of reductions of nitrogen oxides across the U.S., an overview summary figure is presented. Goldberg et al. (2020) used TROPOMI satellite data to illustrate substantial drops in NO₂ during COVID-19 physical distancing between 2020 and 2019. The authors used three different methods. In Method 1, they compared an average of 15 March – 30 April 2020 to the same timeframe of 2019 and account for impact of changes due to solar zenith angle. In Method 2, the authors develop a strategy to account for varying weather patterns without the use of a chemical transport model. In this method, they normalize each day's NO₂ observation to a day with “standard” meteorology—like standard temperature and pressure (STP) conditions in a laboratory setting. They accounted for four different day-varying effects (solar zenith angle, wind speed, wind direction, and day-of-week). In Method 3, the authors inferred a TROPOMI NO₂ column amount assuming no COVID-19 precautions using the GEM-MACH regional chemical transport model, which was operationally run in forecast mode. The authors then compared the actual TROPOMI columns to the theoretical columns. Methods 2 and 3 both accounted for year-varying meteorology, while Method 1 did not.

The authors reported that meteorological patterns were especially favorable for low NO₂ in much of the U.S. during spring 2020, complicating comparisons with spring 2019. After accounting for solar angle and meteorological considerations, the authors calculated that NO₂ declines ranged between 9.2% and 43.4% among 20 cities in North America, with a median of 21.6%. Of the studied cities, largest NO₂ drops (>30%) were in San Jose, Los Angeles, and Toronto, and smallest declines (<12%) were in Miami, Minneapolis, and Dallas. The authors pointed out that normalized NO₂ changes could be used to highlight locations with greater activity changes and better understand the sources contributing to adverse air quality in each city. For illustrative purposes, Fig. 3-69 is presented using Method 1. Note that the largest decreases in NO₂ were near major cities in North America. The authors reported regional decreases in eastern North America. On the other hand, Central and Northwestern U.S. appeared to have seen little change between years. If one wishes to assess the potential for long-range transport to the

U.S., it is important to note that the lockdown period for China was approximately from January 23 to February 13, 2020 (Le et al., 2020). The lockdown in India was imposed initially for three weeks from March 24 to April 14, 2020 and extended to May 3, 2020 (Mahato et al. (2020). Goldberg et al. (2020) defined the “post-COVID-19” period as the timeframe when COVID-19 precautions were instituted, which was defined to be initiated on March 15. According to the authors, the period was plus or minus a few days in various U.S. cities. Fig. 3-69 illustrates column NO₂; in rural areas, column NO₂ changes may be de-coupled to near-surface NO₂ changes.

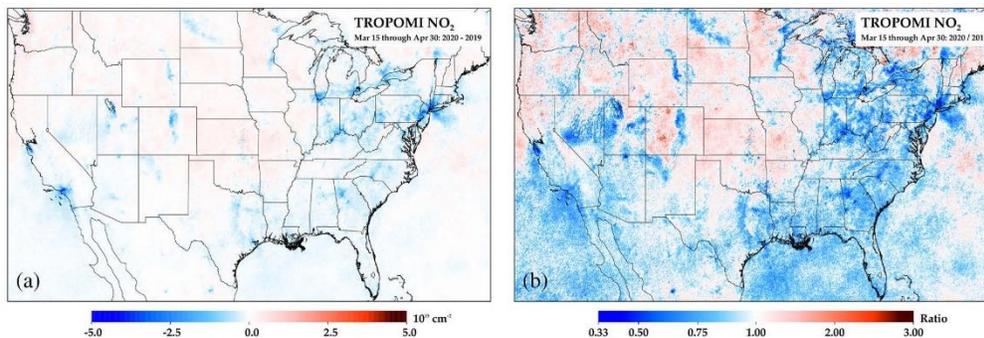


Figure 2. TROPOMI NO₂ differences between 2019 and 2020, using 15 March to 30 April 2020 as the post-COVID-19 period. Plots are showing (a) the absolute difference and (b) the ratio between years.

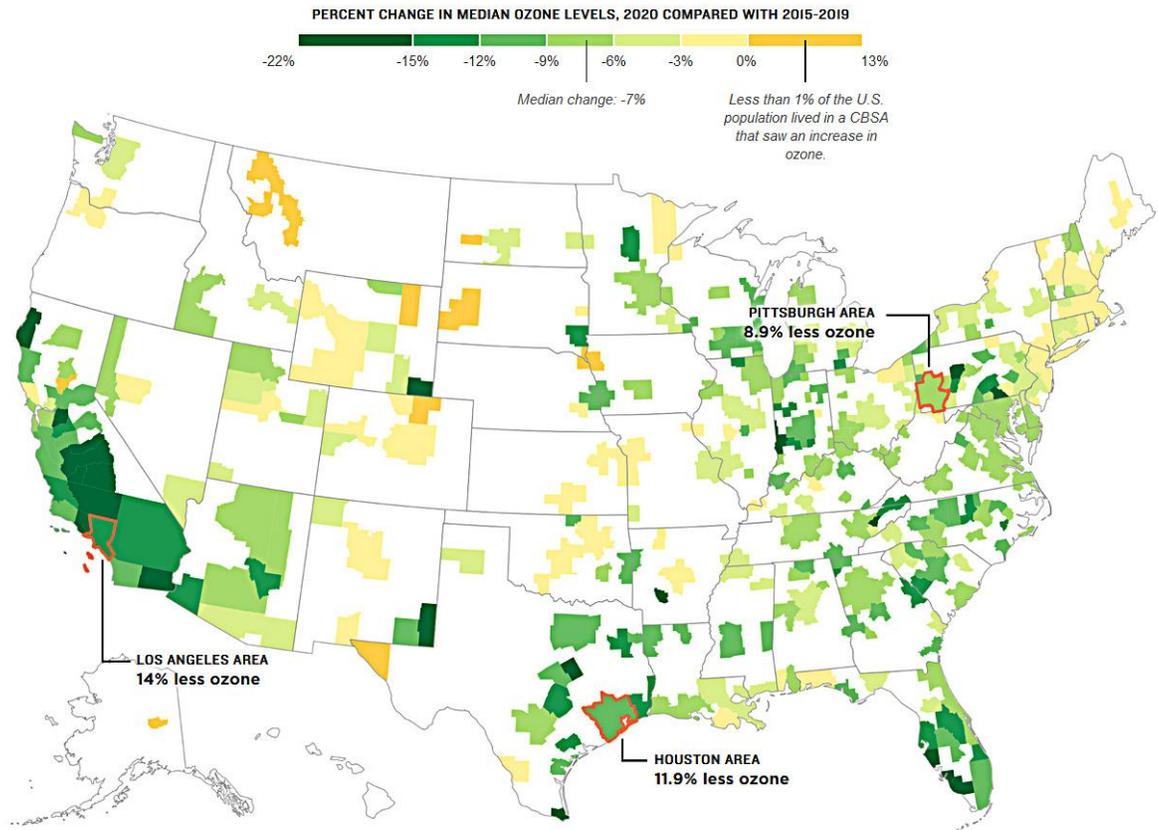
Figure 3-69. TROPOMI NO₂ differences between 2019 and 2020, using the 15 March to 30 April 2020 as the post-COVID-19 period using Method 1 as defined in the study. Plots are showing (a) the absolute difference and (b) the ratio between years. Source: Goldberg et al. (2020).

Chen et al. (2020) described the impacts of COVID-19 lockdown on air quality over the U.S. Many state governments, but not all, in the U.S. issued lockdown or business restrictions amid the COVID-19 pandemic in March 2020. Using air quality data for 28 sites, their analyses revealed widespread but nonuniform reductions of NO₂ and CO during the first phase of lockdown (March 15–April 25, 2020) relative to a pre-lockdown baseline periods (January 25 – March 7, 2020, as well as the same period for 2017–2019). The authors applied a ratio of the mean concentrations at each site during two time periods (P1 defined over the period March 15 – April 25 and P0 defined for the period January 25 – March 7, 2020). The ratio was then compared to similar ratios derived for the baseline period for 2017, 2018, and 2019. The authors reported that the reductions, up to 49% for NO₂ and 37% for CO, were statistically significant at two thirds of the sites and tended to increase with local population density. According to the authors, significant reductions of particulate matter (PM_{2.5} and PM₁₀) only occurred in the Northeast and California/Nevada metropolises, where NO₂ declined the most, while the changes in O₃ concentrations were mixed and relatively minor.

Using data from the EPA databases, Sommer et al. (2020) reported O₃ decreases at many U.S. locations. Sommer et al. (2020), in a National Public Radio (NPR) article, noted that O₃ concentrations decreased during the spring lockdown of 2020 at many locations compared to

spring levels for the years 2015, 2016, 2017, 2018, and 2019. The authors compared the median value for the daily maximum 8-h average concentrations detected during March 15 – April 30, 2020, with levels reported during the comparable period over the previous five years (2015-2019). The authors pooled all the data for the qualifying monitors within a given CBSA together and compared the 2020 median values with the previous 5 years. Their analysis revealed that, in most locations, the median O₃ concentrations decreased by 15% or less. Fig. 3-70 below summarizes the results of their analysis. For the areas investigated, O₃ reductions occurred because of lockdowns in parts of the West, Midwest, and East. According to the authors, during the lockdown period, a series of rainstorms swept through southern California in March, which helped remove pollutants from the air. The Seattle area showed modest declines in O₃ levels, while the Portland area experienced small reductions in O₃ concentrations. The Northeastern U.S. experienced little change. The Mid-Atlantic states experienced modest declines in O₃ levels. In comparing Fig. 3-69, illustrating that the NO₂ changes from Goldberg et al. (2020), with Fig. 3-70, it appears that the spatial patterns for NO₂ reductions were similar to the spatial patterns of change observed for O₃ by Sommer et al. (2020).

Car traffic across the country is down about 40%, but ozone pollution has barely decreased compared with levels over the past five years. Factories, refineries, power plants and diesel trucks are still dominant sources of pollution in many places.



Notes

The areas shown in color represent 335 out of 929 core-based statistical areas (CBSA) in the United States, which are one or more counties, anchored by an urban center of 10,000 people or more. Areas in white are either CBSAs with no available data, too little data or areas outside of CBSAs. Places where ozone increased in 2020 had low ozone levels between 2015 and 2019.

Source: Environmental Protection Agency Air Quality System and AirNow

Credit: Daniel Wood/NPR

Figure 3-70. Percent change in median ozone levels of daily maximum 8-h average concentrations, with 2020 spring levels compared with 2015-2019. Source: Sommers et al. (2020). National Public Radio article (May 19, 2020). Traffic Is Way Down Because of Lockdown, But Air Pollution? Not So Much.
[https://www.npr.org/sections/health-shots/2020/05/19/854760999/traffic-is-way-down-due-to-lockdowns-but-air-pollution-not-so-much.](https://www.npr.org/sections/health-shots/2020/05/19/854760999/traffic-is-way-down-due-to-lockdowns-but-air-pollution-not-so-much)

Using the same O₃ metric (the median value of the daily maximum 8-h average concentrations) applied by Sommer et al. (2020), daily maximum 8-h average concentrations from the EPA AQS and AirNow databases were downloaded for 52 U.S. O₃ monitoring sites for the period March 15 – April 25, 2020. For most of the 52 monitoring sites, the EPA-defined O₃ season contained the months of March and April. The median values characterized by site for the March and April period in 2020 were compared with the same period for 2017, 2018, and 2019. Changes between 5% and -5% were subjectively defined as NO CHANGE because of possible

year-to-year variability of meteorological factors, as well as year-to-year variability of stratospheric-tropospheric transport to the surface (STT-S). For this analysis, it was not possible to take into consideration these, as well as other, variables. Table 3-13 summarizes for 28 O₃ monitoring sites in the West the changes that occurred during lockdown.

Table 3-13. Percent change in median ozone levels of daily maximum 8-h average concentrations in the West for 28 monitoring sites, with 2020 March 15 – April 25 levels compared with 2017-2019.

City	State	AQS ID	Percent Change	Direction of Change
Denali NP	AK	020680003	-4	NO CHANGE
Seattle	WA	530330080	-6	DECLINE
Mount Rainier NP	WA	530530012	-7	DECLINE
Portland	OR	410510080	-2	NO CHANGE
Los Angeles	CA	060371103	-21	DECLINE
Joshua Tree NP	CA	060719002	-12	DECLINE
San Jose	CA	060850005	-7	DECLINE
Fresno	CA	060190011	-23	DECLINE
Sequoia & Kings Cany NP	CA	061070009	-16	DECLINE
Lassen Volcanic NP	CA	060893003	-10	DECLINE
Redding	CA	060890004	-6	DECLINE
Las Vegas	NV	320030540	-8	DECLINE
Great Basin NP	NV	320330101	0	NO CHANGE
Phoenix	AZ	040139997	-16	DECLINE
Chiricahua NM	AZ	040038001	-7	DECLINE
Grand Canyon NP	AZ	040058001	-10	DECLINE
Albuquerque	NM	350010023	-9	DECLINE
Glacier NP	MT	300298001	-5	NO CHANGE
Helena	MT	300490004	-1	NO CHANGE
Cheyenne	WY	560210100	-19	DECLINE
Yellowstone NP	WY	560391011	-3	NO CHANGE
Denver	CO	080590006	-2	NO CHANGE
Denver	CO	080350004	-3	NO CHANGE
Mesa Verde NP	CO	080830101	-6	DECLINE
Rocky Mountain NP	CO	080690007	-4	NO CHANGE
Salt Lake City	UT	490353006	-3	NO CHANGE
Canyonlands NP	UT	490370101	-8	DECLINE
Dinosaur NM	UT	490471002	-5	NO CHANGE

Sites, such as Denali National Park (AK), Glacier National Park (MT), and Yellowstone National Park (WY), which experience little locally generated pollution, experienced no change in O₃ concentrations during lockdown. Similar to the results reported by Sommer et al. (2020), the Seattle area showed modest declines in O₃ levels, while the Portland area experienced small reductions in O₃ concentrations. Table 3-14 summarizes for 10 O₃ monitoring sites in the Midwest, the changes that occurred during lockdown when compared to the same period for 2017 – 2019. Similar patterns of change occurred in the Midwest in this analysis compared to the Sommer et al. (2020) results. Table 3-15 summarizes for 14 O₃ monitoring sites in the East the changes that occurred during lockdown when compared to the same period for 2017 – 2019. The Mid-Atlantic states experienced modest declines in O₃ levels. This analysis showed little changes

in O₃ concentrations occurring in the Northeastern U.S., which was similar to those noted by Sommer et al. (2020).

Table 3-14. Percent change in median ozone levels of daily maximum 8-h average concentrations in the Midwest for 10 monitoring sites, with 2020 March 15 – April 25 levels compared with 2017-2019.

City	State	AQS ID	Percent Change	Direction of Change
Bismarck	ND	380150003	-11	DECLINE
Minneapolis	MN	270031002	-9	DECLINE
Chicago	IL	170310001	-9	DECLINE
Indianapolis	IN	180970078	-10	DECLINE
Kansas City	KS	202090021	-1	NO CHANGE
St. Louis	MO	295100085	-9	DECLINE
Tulsa	OK	401431127	-12	DECLINE
Houston	TX	482010024	-23	DECLINE
Big Bend NP	TX	480430101	-12	DECLINE
Chamizal Nat. Memorial.	TX	481410044	1	NO CHANGE

Table 3-15. Percent change in median ozone levels of daily maximum 8-h average concentrations in the East for 14 monitoring sites, with 2020 March 15 – April 25 levels compared with 2017-2019.

City	State	AQS ID	Percent Change	Direction of Change
Cincinnati	OH	390610040	-12	DECLINE
Cleveland	OH	390350060	-11	DECLINE
Pittsburgh	PA	420030067	-6	DECLINE
Boston	MA	250250042	-3	NO CHANGE
Cape Cod Nat. Seashore	MA	250010002	-2	NO CHANGE
Providence	RI	440071010	-7	DECLINE
New York	NY	360810124	0	NO CHANGE
Bellevue State Park	DE	100031013	-6	DECLINE
Grantville	MD	240230002	-13	DECLINE
District of Columbia		110010043	-2	NO CHANGE
Richmond	VA	510870014	-6	DECLINE
Raleigh	NC	371830014	-12	DECLINE
Great Smoky Mountain NP	TN	470090101	-7	DECLINE
Atlanta	GA	130890002	-13	DECLINE

As noted above, the cause for the increase in O₃ concentration at some locations during lockdown, especially in the industrial and transport dominated areas, was associated with the decrease of NO, which led to the lowering of the O₃ consumption (titration, NO + O₃ = NO₂ + O₂). As emissions are reduced, Simon et al. (2015) note that in the U.S., increasing O₃ trends generally occur in the winter months (defined by the authors as October-April) in more urbanized areas and at the lower end of the O₃ distribution. As indicated in this section, the response of O₃ concentrations to lockdown varied at many locations around the world. Some of

the variation could be associated with the selection of the specific exposure metric used to investigate changes in O₃ concentrations. Some of the investigators selected 24-h average concentrations, while others selected longer-term averaged hourly concentrations, long-term average of daily maximum 8-h average O₃ concentrations, median concentration hourly average values, the ratio of mean concentrations, and the median O₃ levels of the daily maximum 8-h average concentrations. Some of these metrics focus on the lower O₃ concentrations, while other metrics focus on the higher O₃ concentrations. Using the same hourly average O₃ concentration data, Lefohn et al. (2017, 2018) have discussed how the selection of different exposure metrics results in different trend outcomes.

As discussed earlier, as emission reductions occur, the different parts of the distribution of hourly average O₃ concentrations shift differently. Both ends of the distribution of hourly average O₃ concentrations shift toward the middle of the distribution. The lower concentrations shift upward, while the higher concentrations shift downward. Based on the metric selected to characterize changes in O₃ exposures during lockdown compared to previous years, different results would be anticipated. A relevant question to ask is: “Which O₃ exposure metric is most appropriate when comparing differences between exposures during lockdown and other periods?” The answer is that it depends upon the question being asked. For example, if one were interested in exploring the answer to the question: “What is the benefit to public health by lowering emissions associated with O₃ concentrations,” one would focus on a metric associated with the higher part of the O₃ distribution of hourly average concentrations. This is because as noted in earlier discussions, results from controlled human health exposure studies showed that the higher hourly average O₃ concentrations were associated with greater effects. Alternatively, if one were interested in the chemical phenomena associated with how O₃ concentrations change as NO_x concentrations were reduced during lockdown, one might ask the question: “Do the lower hourly average O₃ concentrations increase or decrease due to NO_x reductions?” The first question deals with public health, while the second focuses on how chemical changes in the atmosphere influence the changes in the lower hourly average concentrations. Thus, if one were interested in how emission reductions during lockdown affected public health, one should focus on O₃ exposure indices that highlight changes in the higher concentrations. One such index is the daily maximum of the 8-h average concentration. For health and vegetation concerns, focus should be on the change in the frequency of the upper part of the distribution of concentrations at specific monitoring sites.

The “natural experiment”, a term applied by Patel et al. (2020), showed that large source emission reductions had significant impacts on air quality across the world. The reported worldwide results illustrated that during lockdown as emissions were reduced the lower O₃ concentrations shifted upward. The shifting of the lower concentrations upward as extreme emission reductions occurred during lockdown provides additional evidence that support the second fundamental principle that **Daily Maximum Hourly Averaged O₃ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide.**

4. The Use of the Current Form of the Secondary Ozone Standard versus an Alternative W126/N100 Secondary Ozone Standard to Protect Vegetation

4.1 Introduction

As noted in Section 2, both vegetation effects research and controlled laboratory studies of human volunteers indicate that higher O₃ hourly average concentrations elicit a greater effect on hour-by-hour physiologic response than lower hourly average values. The nonlinear response to O₃ for both human health and vegetation means that the higher values need to be weighted more than the mid and lower hourly average O₃ concentrations. Information provided in Section 3 indicate that both modeling and empirical data show that as emission reductions occur, the distribution of hourly O₃ average concentrations shift from the higher values toward the middle values and the lower values shift upward toward the middle values. There is a compression of the distribution. In addition, as emission reductions occur across the U.S., the time when the highest MDA8 concentrations occur shift at many O₃ monitoring sites from the summer toward the March-June months. Like the seasonal shifting patterns observed for the highest MDA8 concentrations, Neufeld et al. (2019) reported that as emissions were reduced, at most of the six sites analyzed in the Great Smoky Mountains National Park, the maximum 3-month W126 exposures shifted from mid-summer to the April–June period. Neufeld et al. (2019) noted that for the sites analyzed in their analysis, the shift in seasons following reduction of the peak concentrations may reflect a greater percentage of the remaining higher O₃ exposures being associated with natural processes that affect background O₃, such as stratospheric intrusions. The documented information described in Sections 2 and 3 establishes an important scientific foundation from which one can critically evaluate the second draft PA’s (EPA, 2023) recommendation that the current form and level of the secondary O₃ NAAQS is adequate to protect vegetation.

In summarizing the available evidence considering the secondary O₃ NAAQS, the second draft PA (EPA, 2023, page 4-131) concludes

Thus, consideration of the available information, as discussed above, leads us to conclude that the combined consideration of the body of evidence and the quantitative air quality and exposure analyses, including associated uncertainties, do not call into question the adequacy of the protection provided by the current secondary standard. Rather, this information provides support for the current standard, and thus supports consideration of retaining the current standard, without revision.

More specifically, we first conclude that the evidence continues to support O₃ as the indicator for the secondary NAAQS for photochemical oxidants. Second, we recognize that the evidence supports a standard that protects against cumulative growth impacts, such as those characterized by the W126 index, as well as against high, peak hourly concentrations that are associated with visible foliar injury, as well as reduced tree growth.

The second draft PA (EPA, 2023, pages 4-132 – 4-133) concludes

In reaching these conclusions, we have also considered the continued attention given by the CASAC to the W126 index as a potentially suitable metric for a secondary standard. As discussed in this and other sections above, there are aspects of O₃ air quality not controlled by a cumulative index such as the W126 index, and that contribute risk of vegetation effects. These air quality aspects include occurrences of elevated hourly O₃ concentrations which have demonstrated a role in effects such as reduced growth and visible foliar injury. We note the inability of the W126 index to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. This is illustrated, for example, by the fact that some of the experimental exposures on which the RBL E-R functions are based reflect W126 index values that may currently occur in areas of the U.S. that meet the current standard and yet the experimental exposures include tens of hours at/above 100 ppb (which have been found to contribute to greater RBL for the same W126 index, as discussed in sections 4.3.3.1.2 and 4.3.4.1) while more than a handful of such hours are rare in areas meeting the standard. Thus, we conclude that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. In order to achieve such control, in addition to control of cumulative exposures, to protect against vegetation effects, we conclude that if a secondary standard in the form of the W126 index were to be considered by the Administrator, it should also be accompanied by an additional standard more focused on control of high hourly concentrations.

Thus, based on its own analyses in the second draft of the PA (EPA, 2023), the EPA appears to reach the following conclusions for the secondary O₃ NAAQS:

1. The EPA believes that the evidence supports a standard that protects against (1) cumulative growth impacts, such as those characterized by the W126 index, as well as against (2) high, peak hourly concentrations that are associated with visible foliar injury, as well as reduced growth.
2. The EPA believes that there are aspects of O₃ air quality not controlled by a cumulative index, such as the W126 index, that contribute to the risk of vegetation effects. These air quality aspects include occurrences of elevated hourly O₃ concentrations, which have demonstrated a role in effects such as reduced growth and visible foliar injury.
3. The EPA believes that the W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. In its analysis, the EPA notes that some of the experimental exposures on which the RBL E-R functions are based upon W126 index values which are experienced in areas of the U.S. that meet the current standard. However, some of the experimental exposures include tens of hours at/above 100 ppb, which are not exhibited in current areas meeting the current 8-h standard.

4. The EPA believes that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. To achieve such control, in addition to control of cumulative exposures to protect against vegetation effects, the Agency concludes that if a secondary standard in the form of the W126 index were to be considered by the Administrator, the secondary standard should also be accompanied by an additional standard more focused on control of high hourly concentrations.
5. The EPA believes that the body of evidence does not call into question the adequacy of the protection for vegetation provided by the current secondary standard.

Each of the above five items will be addressed in this section (Section 4). As discussed in these comments, the EPA's conclusion that the current form and level of the secondary O₃ NAAQS is adequate to protect vegetation is not as certain as the Agency believes.

On page 4-129, the second draft of the PA (EPA, 2023) notes

We next consider the quantitative analyses available in this review with regard to the control of air quality conditions that might pose risks to the public welfare by the current standard. In so doing, we also consider the W126 index values associated with a cross-species median tree seedling RBL below 6%. As noted above, at the time the current standard was set in 2015, the W126 index value corresponding to a median RBL of 6% was 19 ppm-hrs (based on the E-R functions for 11 species recognized in multiple past reviews). In this reconsideration, we have additionally considered, at the recommendation of the CASAC, a newly available study, released since the 2020 ISA. The new study derives E-R functions for 16 species, including six additional to the prior study, using an approach different than that employed by the previously available study. The more recent analyses incorporate different methodology and considerations from the prior study, with neither approach clearly indicated to provide estimates more appropriate for representing RBL in the context here. Accordingly, as summarized in section 4.5.1.2 above, consideration of RBL estimates from each of the two studies and from a combined consideration that provides estimates for all 17 species yields median estimates associated with 6.0% to range from 19 ppm-hrs up to somewhat above 25 ppm-hrs. Thus, air quality summarized by a W126 index below 19 ppm-hrs, and even somewhat higher, may, based on the additional study, be concluded to yield a cross-species median RBL below 6.0%.

The EPA believes that when considering the (1) RBL estimates from each of the two studies (i.e., Study 1 based on the E-R functions for 11 species recognized in multiple past reviews and Study 2 based on the E-R functions for 16 species as analyzed by Lee et al. (2022) and (2) the RBL estimates derived from a combined consideration of the two studies), the median estimates, associated with 6.0%, biomass loss range from 19 ppm-hrs up to somewhat above 25 ppm-hrs. Thus, the EPA concludes in the second draft PA (EPA, 2023) that a yield a cross-species median RBL below 6.0% is predicted for a W126 index below 19 ppm-hrs.

One of the decisions by the Administrator is whether he believes the Agency can protect the most vulnerable vegetation at a W126 level of 19 ppm-hrs. Based on combining the results from the two studies, the second draft of the PA (EPA, 2023) notes that a W126 index below 19 ppm-hrs, and even somewhat higher will adequately protect vegetation based on the median RBL methodology applied by the Agency. However, Lee et al. (2022) identifying the most sensitive tree species (i.e., black cherry, ponderosa pine, quaking aspen, red alder, American sycamore, tulip poplar and winged sumac) predict that an O₃ cumulative exposure, using a three-month 12-h W126, will produce a 5% biomass loss in the range 2.5–9.2 ppm-hrs. Lee et al. (2022) note that these seven sensitive tree seedling species are ecologically important and occur widely across U.S.

Lee et al. (2022) note that

The most sensitive species in our analysis had a biomass loss of 5% at a W126 of 2.5–9.2 ppm-hrs and N100 ranged from 0 to 7 at those exposures. However, there were few experimental exposures between 2.5 and 9.2 ppm-hrs. In that same 2.5–9.2 ppm-hrs range, U.S. O₃ monitoring data from 2016 to 2018 indicate that N100 ranged from 0 to 10, with an N100 of 0 at most monitors (US EPA, 2020b). While the episodic peaks have been declining across the U.S. in the past few decades, there are still areas with N100 peaks where the sensitive species occur. ***Therefore, we believe the exposure response relationships reported in this study remain relevant to current O₃ exposures, especially for the sensitive species*** (emphasis added).

In their publication, Lee et al. (2022) refer to the N100 metric, which is the number of hours \geq 100 ppb.

For developing functions, Lee et al. (2022) used a different approach than Lee and Hogsett (1996). As noted in the second draft of the PA (EPA, 2023, page 4A-31)

Lee et al. (2022) derived species-specific E-R functions for RBL (termed “PRBL” [predicted RBL] in Lee et al al., 2022), by a different approach than that of Lee and Hogsett (1996). The different approaches yield different RBL functions for species common to both studies even when based on the same experiments (see Table 4A-2 and Table 4A-9). While the Lee and Hogsett (1996) analysis explicitly derived functions describing the response for each experimental dataset and then derived a composite function in terms of a 92-day W126 from the experiment-specific functions for each species, Lee et al. (2022) applied a statistical model to all exposure and response datasets for each species. Before application of the model, the authors scaled the W126 index for single-year exposures to 92 days (through application of a factor of 92 divided by the experiment duration). For 2-year exposures, the authors assigned the scaled 92-day W126 index for the second year of exposure to the 2-year responses.

The analyses by Lee et al. (2022) derived parameters for functions to estimate RBL from W126 index based on linear or Weibull models parameterized to

describe tree seedling biomass (as log total dry weight) for each species at the chamber mean level as a function of 92-day W126 index. The Weibull model is then used to derive the function for RBL (termed “PRBL” [predicted RBL] in Lee et al., 2022).¹⁰ The three-parameter Weibull model was fit using a mixed-effects model to estimate the 3 parameters. This allowed the inclusion of additional explanatory variables that are correlated with the response, but not related to O₃ exposure. In these analyses, this was primarily related to initial plant size, which varied from chamber to chamber. Four species assessed response after both one and two years of exposure. For three of those species the response was not significantly different for the 2-yr exposure (than for the first year of exposure), however there was a significant difference in response between the two for ponderosa pine, so two sets of parameters were included, one for the single-year exposure and one for the 2-year exposure (Table 4A-9 and section 4A.3.2below).

As noted in the second draft of the PA (EPA, 2023, page 4-129), the EPA believes that while the Lee et al. (2022) analyses incorporated different methodology and considerations from that used in the Lee and Hogsett (1996) study, neither approach, according to the Agency, clearly indicated estimates more appropriate for representing RBL. On page 4A-33 of the second draft of the PA (EPA, 2023), the different E-R functions for the RBL and the W126 index for seedlings of 10 tree species from Lee and Hogsett (1996) and Lee et al. (2022) are illustrated. The figure is presented below in Fig. 4-1.

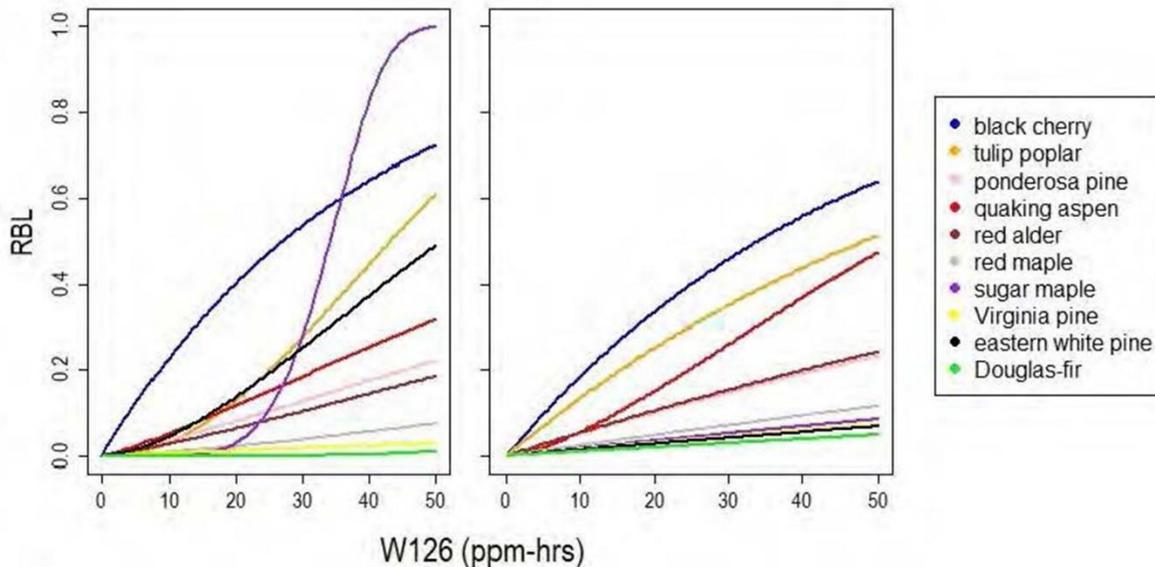


Figure 4A-16. E-R functions for RBL and W126 index for seedlings of 10 tree species from Lee and Hogsett (1996) (left) and Lee et al. (2022) (right).

Figure 4-1. E-R functions for RBL and W126 index for seedlings of 10 tree species from Lee and Hogsett (1996) (left) and Lee et al. (2022) (right). Source: EPA (2023, page 4A-33).

An important decision by the Administrator is whether to focus on the results associated with (1) the most recent analyses performed by Lee et al. (2022, (2) the analyses summarized by

Lee and Hogsett (1996) or (3) a combined consideration of the two analyses. There are clear differences between the predicted E-R functions associated with the different methodologies applied by Lee and Hogsett (1996) and the most recent analysis by Lee et al. (2022). The Administrator's choice of which of the three analyses to use will affect decisions about whether the current form of the secondary O₃ NAAQS is appropriate for protecting vegetation. Applying an average of the Lee and Hogsett (1996) and the Lee et al. (2022) analyses can mathematically obscure the differences associated with each of the two analyses. While Lee et al. (2022) described the two models, the authors did not state in their paper whether one model was preferred more than the other. However, one might assume that the lead author of both papers (i.e., Dr. Henry Lee) prefers the model selected in the most current analyses as described in Lee et al. (2022) rather than the model applied in the Lee and Hogsett (1996) analyses. As indicated above, my assumption is that the lead author (i.e., Dr. Lee) modified the model applied in Lee and Hogsett (1996) because he and his coauthors believed that the most current model which was used in Lee et al. (2022) represented an improvement over the model applied in Lee and Hogsett (1996). At this point, without a detailed critique provided in the second draft PA (EPA, 2023) justifying the use of the average of the two analyses, it is suggested that the focus be on the results associated with the Lee et al. (2022) analyses.

An additional decision by the Administrator is how much weight to give to the major concerns expressed in the second draft of the PA (EPA, 2023) that some of the experimental treatments contained in the NCLAN and tree seedling experiments experienced numerous occurrences of high hourly average O₃ concentrations (e.g., N100 values). Footnote 74 in the second draft PA (EPA, 2023, page 4-55) notes that even for the experimental treatments with W126 index levels of a magnitude commonly experienced at U.S. sites that meet the current 8-hour standard, the values for N100 extend up above 10 to more than 40 in one instance (black cherry and aspen). The second draft PA (EPA, 2023) notes that across the full set of treatments, values for N100 extend into the hundreds and up above 500 over 121 days. As will be discussed in this section, some of the experimental exposure protocols applied in both the NCLAN and tree seedling experiments were designed to explore the different growth effects on vegetation associated with episodic and daily peak exposures (see for example Hogsett et al., 1985). The daily peak experimental exposure profiles were **not** designed to mimic ambient exposures experienced in the U.S. In many instances, the focus of some of the experiments was on comparing vegetation growth effects associated with (1) episodic treatments that contained infrequent high hourly average O₃ concentrations and (2) daily peak treatments that contained more frequent high (but less magnitude than the concentrations used in the episodic exposures). For example, Lee et al. (2022) report that the total plant biomass response to the daily peak O₃ treatment for the 1989 quaking aspen study (Study 11) was significantly less than that for the episodic treatment with both treatments experiencing similar W126 values. To summarize, the daily peak experimental exposure profiles used in some of the NCLAN and tree seedling experiments were not designed to mimic ambient exposures experienced in the U.S.

However, besides the daily peak treatments which were not designed to mimic ambient exposures, there were treatments in the vegetation experiments where N100 values occurred. As noted in the second draft of the PA (EPA, 2023), Lefohn and Foley (1992) and Lefohn et al. (1997) reported that some of the O₃ treatments in the NCLAN and tree seedling experiments experienced hourly average O₃ concentrations ≥ 100 ppb. Please note that Lefohn and Foley

(1992), which is cited in the second draft PA (EPA, 2023), is currently not listed in any of the reference sections within the document. However, as noted earlier, Lee et al. (2022) report that the most sensitive species in their analysis experienced a biomass loss of 5% at a W126 of 2.5–9.2 ppm-hrs with N100 ranging from 0 to 7 at those exposures. In that same 2.5–9.2 ppm-hrs range, Lee et al. (2022) report that U.S. O₃ monitoring data from 2016 to 2018 indicated that N100 ranged from 0 to 10, with an N100 of 0 at most monitors. The authors noted that while the episodic peaks have been declining across the U.S. in the past few decades, there are still areas with N100 peaks where the sensitive species identified by Lee et al. (2022) occur.

The second draft of the PA (EPA, 2023) favors the use of the current form and level of the primary standard (the 3-year average of the 4th highest daily maximum 8-h average concentration) to protect vegetation. However, the EPA has concluded that if a secondary standard in the form of the W126 index were to be considered by the Administrator, it should also be accompanied by an additional standard focused on the control of the high hourly concentrations. As noted later in Section 4 of these comments, the Forest County Potawatomi Community (FCPC) currently couples the W126 exposure index with the number of hours \geq 100 ppb (N100) for its O₃ Air Quality Related Values (AQRVs) and related Threshold Effect Levels (TELS) to protect vegetation. FCPC believes that the W126 and N100 indicators are most suitable for the protection of its Class I area. As will be discussed in this section, it is recommended, based on the decisions made by the Forest County Potawatomi Community, that only a single secondary O₃ standard would be required (versus EPA's suggestion of two secondary standards – one secondary standard based on a cumulative form (i.e., W126) and another secondary standard based on the control of high hourly average concentrations (i.e., N100). A single secondary O₃ standard would be based on the coupling of the W126 with the N100 to protect vegetation. The decision to change the form and level of the current secondary NAAQS (i.e., the 3-year average of the 4th highest daily maximum 8-h average concentration) to a form and level associated with the W126 cumulative metric coupled with the N100 should be based on whether the current form and level of the secondary NAAQS can control for the W126 level and N100 values that the EPA Administrator deems critical for the protection of vegetation.

It is important to review the evidence that led to the EPA's decision in the second draft PA (EPA, 2023) to recommend to the Administrator that he retain the current form of the welfare O₃ standard without revision. The pages that follow review the evidence and provide detailed comments on the Agency's recommendation.

The second draft PA (EPA, 2023, page 4-2) notes that

As a result of the O₃ NAAQS review completed in 2015, the level of the secondary standard was revised to 0.070 ppm, in conjunction with retaining the indicator (O₃), averaging time (8 hours) and form (fourth-highest annual daily maximum 8-hour average concentration, averaged across three years). The establishment of this standard in 2015, and its retention in 2020, is based primarily on consideration of the extensive welfare effects evidence base compiled from more than fifty years of extensive research on the phytotoxic effects of O₃, conducted both in and outside of the U.S., that documents the impacts of O₃ on plants and their associated ecosystems (U.S. EPA, 1978, 1986,

1996, 2006, 2013). Key considerations in the 2015 decision were the scientific evidence and technical analyses available at that time, as well as the Administrator's judgments regarding the available welfare effects evidence, the appropriate degree of public welfare protection for the revised standard, and available air quality information on seasonal cumulative exposures (in terms of the W126-based exposure index¹) that may be allowed by such a standard (80 FR 65292, October 26, 2015).

The second draft PA (EPA, 2023, page 4-2) describes the definition of the W126 index in footnote 1 as

¹ The W126 index is a cumulative seasonal metric described as the sigmoidally weighted sum of all hourly O₃ concentrations during a specified daily and seasonal time window, with each hourly O₃ concentration given a weight that increases from zero to one with increasing concentration (80 FR 65373-74, October 26, 2015). The units for W126 index values are ppm-hours (ppm-hrs). More detail is provided in section 4.3.3.1.1 below.

The second draft PA (EPA, 2023, page 4-4) notes that

For quantifying effects on tree seedling growth as a surrogate or proxy for a broader array of vegetation-related effects using the RBL metric, in 2015 and 2020 the evidence base provided established E-R functions for seedlings of 11 tree species (80 FR 65391-92, October 26, 2015; 2014 PA, Appendix 5C; 85 FR 87307-9, 87313-4, December 31, 2020; 2020 PA, Appendix 4A). Cumulative O₃ exposure was evaluated in terms of the W126 cumulative seasonal exposure index, an index supported by the evidence in the 2013 and 2020 ISAs for this purpose and that was consistent with advice from the CASAC in both reviews (2013 ISA, section 9.5.3, p. 9-99; 80 FR 65375, October 26, 2015; 2020 ISA, section 8.13; 85 FR 87307-8, December 31, 2020). In judgments regarding effects that are adverse to the public welfare, the decision setting the standard in 2015, and that retaining it in 2020, both utilized the RBL as a quantitative tool within a larger framework of considerations pertaining to the public welfare significance of O₃ effects (80 FR 65389, October 26, 2015; 73 FR 16496, March 27, 2008; 85 FR 87339-41, December 31, 2020).

The second draft PA (EPA, 2023, page 4-5) notes that

In using the RBL estimates as a proxy, the Administrator in 2015 focused her attention on a revised standard that would generally limit cumulative exposures to those for which the median RBL estimate for seedlings of the 11 species with established E-R functions would be somewhat below 6% (80 FR 65406-07, October 26, 2015).⁴ She noted that the median RBL estimate was 6% for a cumulative seasonal W126 exposure index of 19 ppm-hrs (80 FR 65391-92, Table 4, October 26, 2015). Given the information on median RBL at different W126 exposure levels, using a 3-year cumulative exposure index for assessing

vegetation effects,⁵ the potential for single-season effects of concern, and CASAC comments on the appropriateness of a lower value for a 3-year average W126 index, the Administrator judged it appropriate to identify a standard that would restrict cumulative seasonal exposures to 17 ppm-hrs or lower, in terms of a 3-year W126 index, in nearly all instances (80 FR 65407, October 26, 2015). Based on such information, available at that time, to inform consideration of vegetation effects and their potential adversity to public welfare, the Administrator additionally judged that the RBL estimates associated with marginally higher exposures in isolated, rare instances were not indicative of effects that would be adverse to the public welfare, particularly in light of variability in the array of environmental factors that can influence O₃ effects in different systems and uncertainties associated with estimates of effects associated with this magnitude of cumulative exposure in the natural environment (80 FR 65407, October 26, 2015).

The second draft PA (EPA, 2023, page 4-6) continues

Using these objectives, the 2015 decision regarding a standard revised from the then existing (2008) standard was based on extensive air quality analyses that included the most recently available data as well as air monitoring data that extended back more than a decade (80 FR 65408, October 26, 2015; Wells, 2015). These analyses evaluated the cumulative seasonal exposure levels in locations meeting different alternative levels for a standard of the existing form and averaging time. These analyses supported the Administrator's judgment that a standard with a revised level in combination with the existing form and averaging time could achieve the desired level of public welfare protection, considered in terms of cumulative exposure, quantified as the W126 index (80 FR 65408, October 26, 2015). Based on the extensive air quality analyses and consideration of the W126 index value associated with a median RBL of 6%, and the W126 index values at monitoring sites that met different levels for a revised standard of the existing form and averaging time, the Administrator additionally judged that a standard level of 70 ppb would provide the requisite protection. The Administrator noted that such a standard would be expected to limit cumulative exposures, in terms of a 3-year average W126 exposure index, to values at or below 17 ppm-hrs, in nearly all instances, and accordingly, to eliminate or virtually eliminate cumulative exposures associated with a median RBL of 6% or greater (80 FR 65409, October 26, 2015).

The second draft PA (EPA, 2023, page 4-7) notes that

In 2020, as in 2015, the Administrator considered the available information regarding the appropriate O₃ exposure metric to employ in assessing adequacy of air quality control in protecting against RBL. In addition to finding it appropriate to continue to consider the seasonal W126 index averaged over a 3-year period to estimate median RBL (as was concluded in 2015), the Administrator in 2020 also judged it appropriate to also consider other metrics including peak hourly

concentrations⁸ (85 FR 87344, December 2020). With regard to these conclusions, his considerations included the extent of conceptual similarities of the 3-year average W126 index to some aspects of the derivation approach for the established E-R functions, the context of RBL as a proxy (as recognized above), and limitations associated with a reliance solely on W126 index as a metric to control exposures that might be termed “unusually damaging”⁹ (85 FR 877339-40, December 31, 2020).

The second draft PA (EPA, 2023, page 4-8) pointed out that with regard to the derivation and application of the established E-R functions, the 2020 review recognized several factors that contributed to the uncertainty and some resulting imprecision or inexactitude to RBL estimated from single-year seasonal W126 index values. Footnote 10 in the second draft PA (EPA, 2023, page 4-8) notes that

¹⁰ The E-R functions were derived mathematically from studies of different exposure durations (varying from shorter than one to multiple growing seasons) by applying adjustments so that they would yield estimates normalized to the same period of time (season). Accordingly, the estimates may represent average impact for a season, and have compatibility with W126 index averaged over multiple growing seasons or years (85 FR 87326, December 31, 2020; 2020 PA, section 4.5.1.2, Appendix 4A, Attachment 1). The available information also indicated that the patterns of hourly concentrations (and frequency of peak concentrations, e.g., at/above 100 ppb) in O₃ treatments on which the E-R functions are based differ from the patterns in ambient air meeting the current standard across the U.S. today (85 FR 87327, December 31, 2020). Additionally noted was the year-to-year variability of factors other than O₃ exposures that affect tree growth in the natural environment (e.g., related to variability in soil moisture, meteorological, plant-related and other factors), that have the potential to affect O₃ ER relationships (ISA, Appendix 8, section 3.12; 2013 ISA section 9.4.8.3; PA, sections 4.3 and 4.5). All of these considerations contributed to the finding of a consistency of the use of W126 index averaged over multiple years with the approach used in deriving the E-R function, and with other factors that may affect growth in the natural environment (85 FR 87340, December 31, 2020).

The second draft PA (EPA, 2023), pages 4-9 and 4-10) notes that

With regard to the EPA’s use of a 3-year average W126 index to assess protection from RBL, the 2020 decision additionally took into account the 2019 court remand on this issue, including the remand’s reference to protection against “unusually damaging years.” (85 FR 87325-87328, December 31, 2020). Accordingly, the EPA considered air quality analyses of peak hourly concentrations in the context of considering protection against “unusually damaging years.” With regard to this caution, and in the context of controlling exposure circumstances of concern (e.g., for growth effects, among others), the EPA considered air quality analyses that investigated the annual occurrence of elevated hourly O₃ concentrations which may contribute to vegetation exposures of concern (2020 PA, Appendix 2A, section 2A.2; Wells, 2020). These air quality

analyses illustrate limitations of the W126 index (whether in terms of a 3-year average or a single year) for the purpose of controlling peak concentrations,¹¹ and also the strengths of the current standard in this regard. The air quality analyses show that the form and averaging time of the existing standard, in addition to controlling cumulative exposures in terms of W126 (as found in the 2015 review), is much more effective than the W126 index in limiting peak concentrations (e.g., hourly O₃ concentrations at or above 100 ppb)¹² and in limiting number of days with any such hours (Wells, 2020, e.g., Figures 4, 5, 8, 9 compared to Figures 6, 7, 10 and 11).¹³ Thus, the W126 index, by its very definition, and as illustrated by the air quality data analyses, does not provide specificity with regard to year-to-year variability in elevated hourly O₃ concentrations with the potential to contribute to the “unusually damaging years” that the CASAC had identified for increased concern in the 2015 review. As a result, the 2020 decision found that a standard based on a W126 index (either a 3-year or a single-year index) would not be expected to provide effective control of the peak concentrations that may contribute to “unusually damaging years” for vegetation.¹⁴ Based on all of the above, the 2020 decision concluded that control of such years is a characteristic of the existing standard (the effectiveness of which is demonstrated by the air quality analyses), and that that use of a seasonal W126 averaged over a 3-year period, which is the design value period for the current standard, to estimate median RBL using the established E-R functions, in combination with a broader consideration of air quality patterns, such as peak hourly concentrations, is appropriate for considering the public welfare protection provided by the standard (85 FR 87340-87341, December 31, 2020).

The second draft PA (EPA, 2023, page 4-9) in footnote 11 notes that

¹¹ The W126 index cannot, by virtue of its definition, always differentiate between air quality patterns with high peak concentrations and those without such concentrations.

The second draft PA (EPA, 2023, page 4-10) notes in footnote 12 that

¹² As described in section 4.3.3 below, the occurrence of high concentrations (including those at or above 100 ppb [e.g., Smith, 2012; Smith et al., 2012]), as well as cumulative exposures influence the effects of O₃ on plants.

The second draft PA (EPA, 2023, page 4-10) notes in footnote 14 that

¹⁴ From these analyses, the Administrator concluded that the form and averaging time of the current standard is effective in controlling peak hourly concentrations and that a W126 index based standard would be much less effective in providing the needed protection against years with such elevated and potentially damaging hourly concentrations.

Based on the stated goal by the EPA to control cumulative O₃ exposures, as well as peak hourly average concentrations (Federal Register, 2020b; EPA, 2023), to protect vegetation (i.e., damage and visible foliar injury), it is important to review the historical development of the cumulative W126 exposure index and the coupling of this metric with the frequency of hourly average O₃ concentrations ≥ 100 ppb (N100). This historical review will provide a firm foundation for further discussion in Section 4 of these comments for the support of a unique secondary O₃ NAAQS, different in form and level from the current 8-hour average, for the protection of vegetation in the United States.

4.2 The Development and Application of the W126 Exposure Index

In 1985, based on the experimental findings associated with vegetation effects that the higher hourly average O₃ concentrations were more important than the mid- and low-level values, the W126 exposure index was created by the author of these comments. The weighting of the W126 cumulative exposure index was based on physiological considerations. The weighting is continuous, does not require the identification of a threshold concentration, and deals with plateau considerations at the high end of the distribution of exposures. The W126 exposure index is a non-threshold metric that is the sigmoidally weighted sum of all hourly O₃ values observed during a specified daily and seasonal time window, where each hourly O₃ concentration is assigned a weight that increases from zero to one with increasing value. Lefohn and Runeckles (1987) proposed the use of the sigmoidally weighted index for assessing vegetation based on evidence indicating the greater relative importance of higher concentrations in affecting vegetation in comparison to the mid and lower values (Musselman et al., 1983; Hogsett et al., 1985). Lefohn et al. (1988) mathematically described and applied the W126 exposure index to develop vegetation exposure-response relationships. The W126 exposure index has the form: $W126 = \sum w_i \times C_i$ with weight $w_i = 1/[1 + M \times \exp(-A \times C_i/1000)]$, where $M = 4403$, $A = 126$, and where C_i is the hourly average O₃ concentration in units of ppb. The M and A constants were derived based on the desire to weight (1) the hourly average concentration levels at a value of one at ≥ 100 ppb and (2) the hourly average concentrations at extremely low values below 40 ppb. The weighting value of “one” for all hourly average O₃ concentrations ≥ 100 ppb was based on the recommendation of Professor O.C. Taylor, a senior vegetation researcher at the University of California Riverside and a participant in the EPA’s National Crop Loss Assessment Network (NCLAN) program. The weighting levels below 40 ppb were assigned low values because it was assumed that background O₃ concentrations were mostly associated with these lower levels (EPA, 2006). As is recognized today, hourly average concentrations associated with background O₃ can, at limited times and locations, be significantly higher than 40 ppb because of stratospheric-tropospheric transport to the surface (Lefohn et al., 2011, 2012, 2014; Emery et al., 2012; Lin et al., 2012; Federal Register, 2015; EPA, 2020b). The weightings applied to the hourly average O₃ concentrations for the calculation of the W126 exposure index are illustrated in Fig. 4-2. The form of the W126 exposure metric applies the experimental evidence that the higher O₃ hourly average concentrations are more important than the mid and low values for assessing vegetation impact. The use of the index implies that there exists a relative vegetation effects relationship between the higher part of a distribution of hourly average O₃ concentrations compared to the middle and lower part of the distribution. That is, the assumption, based on the experimental evidence in the field, as well as in the laboratory, is that a reduction of the higher

hourly average O₃ concentrations in a distribution of ambient hourly average O₃ concentrations will result in a reduction in vegetation damage and visible foliar injury. This assumption was validated when reductions in ambient O₃ exposures in the Los Angeles Basin resulted in reductions in ponderosa pine growth impacts in the San Bernardino National Forest (EPA, 2013, 2023). There is nothing in the formulation of the W126 metric that requires the occurrence of “high peak” concentrations. The formulation of the metric simply implies that there exists a relative relationship among the various hourly average O₃ concentrations within the distribution of concentrations and some hourly average distributions may be more important than other distributions in eliciting an adverse effect on vegetation. In some locations in the U.S., the distribution of hourly average O₃ concentrations can contain “peak” hourly average concentrations, but in other locations “peak” concentrations are absent. For this discussion, the term “peak” hourly average concentrations refers to hourly average values ≥ 100 ppb as discussed by Lefohn and Foley (1992, 1993) and Lefohn et al. (1997). ***Based on the discussion in the second draft PA (EPA, 2023), if it is found experimentally that a specific O₃ exposure regime, for example one that does not experience numerous numbers of “high peak concentrations” or no “high peak concentrations” is responsible for eliciting an important adverse effect on vegetation, it is assumed that the EPA (as a policy decision implemented by the Administrator) would employ a secondary O₃ NAAQS that is protective of such exposure regimes if the affected vegetation were found to be ecologically and/or economically important.***

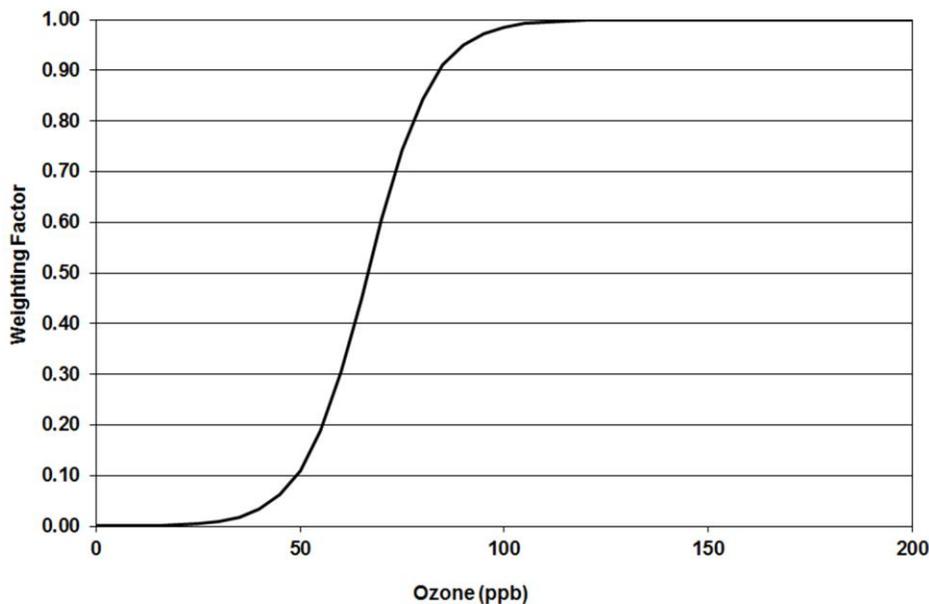


Figure 4-2. The weighting applied to hourly average ozone values for the calculation of the W126 exposure index (see Lefohn et al., 1988).

In the second draft PA (EPA, 2023, page 4-91), the EPA states

We additionally note that while in its approach to emphasizing higher concentrations, the W126 index assigns greater weights to higher hourly concentrations, it cannot, given its definition as an index that sums three months

of weighted hourly concentrations into one, single value, always differentiate between air quality patterns with frequent high peak concentrations and those without such concentrations.⁹⁵ While the metric describes the pattern of varying growth response observed across the broad range of cumulative exposures examined in the tree seedling E-R studies (see Appendix 4A), given the way it is calculated the W126 index can conceal peak concentrations that can be of concern. More specifically, one season or location could have few, or even no, hourly concentrations above 100 ppb⁹⁶ and the second could have many such concentrations; yet (due to greater prevalence of more mid-range concentrations, e.g., contributing to a generally higher average hourly concentration in the second) each of the two seasons or locations could have the identical W126 index (e.g., equal to 25 or 15 or 10 ppm-hrs, or some other value), as discussed in section 4.4.1 above.

Footnote 95 in the second draft PA (EPA, 2023, page 4-91 states

⁹⁵ This is illustrated by the following two hypothetical examples. In the first example, two air quality monitors have a similar pattern of generally lower average hourly concentrations but differ in the occurrence of higher concentrations (e.g., hourly concentrations at or above 100 ppb). The W126 index describing these two monitors would differ. In the second example, one monitor has appreciably more hourly concentrations above 100 ppb compared to a second monitor; but the second monitor has higher average hourly concentrations than the first. In the second example, the two monitors may have the same W126 index, even though the air quality patterns observed at those monitors are quite different, particularly with regard to the higher concentrations, which have been recognized to be important in eliciting responses (as noted above).

The second draft PA (EPA, 2023, page 4-92) notes

Accordingly, in our consideration of the potential for vegetation-related effects to occur under air quality conditions associated with the current standard, while we continue to focus on the W126 index as an appropriate metric for assessing potential risk, given the availability of ER functions for growth impacts as a function of W126 index, we also take note of the important role of the frequency of particularly high concentrations. As indicated by the findings of Lee et al. (2022), W126 index alone does not completely describe aspects of O₃ exposure that elicit effects on growth, and as discussed in section 4.3.3.2 above, this may also be the case for other effects, particularly, visible foliar injury. As discussed in section 4.3.3.2 above, the evidence indicates an important role for peak concentrations (e.g., that might be characterized by metrics such as N100) in influencing the occurrence and severity of visible foliar injury. Thus, while we continue to recognize the W126 index as an appropriate and biologically relevant focus for assessing air quality conditions with regard to potential effects on vegetation growth and related effects, we recognize the need to also account for the influence of the pattern and magnitude of peak concentrations.

While the second draft of the PA (EPA, 2023) appears to be highly focused on the occurrence of hourly average O₃ concentrations ≥ 100 ppb, as noted earlier, there is nothing in the formulation of the W126 exposure index that requires the occurrence of “high peak” concentrations. The formulation of the metric simply implies that there exists a relative relationship among the various hourly average O₃ concentrations within the distribution of concentrations and some distributions may be more important than other distributions in eliciting an adverse effect on vegetation. The distribution of hourly average O₃ concentrations can contain or not contain “peak” hourly average concentrations. To repeat an earlier statement in this discussion, if it is found experimentally either in the field or laboratory that a specific O₃ exposure regime, for example one that does not experience numerous occurrences of “high peak” concentrations or, in fact, contains no “high peak” concentrations is responsible for eliciting an important adverse effect on vegetation, it is assumed that the EPA (as a policy decision implemented by the Administrator) would employ a secondary O₃ NAAQS that is protective of such cumulative exposure regimes if the affected vegetation were found to be ecologically and/or economically important. The form of the secondary O₃ NAAQS would not necessarily be the current 8-hour form of the secondary NAAQS if it were found that the 8-hour form could not adequately prevent those O₃ exposure regimes responsible for vegetation damage or visible foliar injury to occur.

As noted above, the W126 exposure index is a cumulative seasonal metric described as the sigmoidally weighted sum of all hourly O₃ concentrations during a specified daily and seasonal time window, with each hourly O₃ concentration given a weight that increases from zero to one with increasing concentration. The magnitude of the W126 index is influenced by the distribution of the hourly average O₃ concentrations during a specified daily and seasonal time window. Although the weighting scheme for the W126 metric (Fig. 4-2) allows for all hourly average O₃ concentrations, as noted previously, it is not a necessary requirement that hourly average O₃ concentrations ≥ 100 ppb are present for vegetation damage to occur. The weighting scheme indicates that within any distribution of hourly average O₃ concentrations that occur during a specified daily and seasonal time window, the higher hourly average O₃ concentrations during this defined period are weighted more than the middle and lower values. Throughout EPA’s discussion in the second draft PA (EPA, 2023), the EPA has emphasized its concern about the occurrence of hourly average O₃ concentrations ≥ 100 ppb (referred to as the N100 or D100 values) in eliciting adverse vegetation effects (e.g., visible foliar injury or damage). As noted above, the Agency refers at times to these hourly average O₃ values as “high peak” concentrations. There is no requirement that hourly average O₃ concentrations ≥ 100 ppb must occur to elicit adverse vegetation effects. ***What is most important is the identification of those O₃ exposure regimes responsible for causing adverse vegetation effects.*** In some cases, those O₃ exposure regimes responsible for eliciting adverse effects may be associated with an extremely low number of occurrences of hourly average O₃ concentrations ≥ 100 ppb or perhaps even no occurrences. In the second draft PA (EPA, 2023), the EPA appears to be overly focused on the numerous occurrences of hourly average O₃ concentrations experienced in some of the experimental vegetation regimes. It is suggested that the second draft of the PA (EPA, 2023) focus on capturing those O₃ exposure regimes responsible for eliciting adverse effects on the tree and crop species of interest. Lee et al. (2022) reported that for the most sensitive species in their analysis, these species experienced a biomass loss of 5% at a W126 of 2.5–9.2 ppm-hrs with

N100 values ranging from 0 to 7 at those W126 exposures. Specifically, based on the discussion in the second draft PA (EPA, 2023), if it is found experimentally that a specific O₃ exposure regime, for example one that does not experience numerous numbers of “high peak” concentrations or no “high peak” concentrations is responsible for eliciting an important adverse effect on vegetation, it is assumed that the EPA (as a policy decision implemented by the Administrator) would employ a secondary O₃ NAAQS that is protective of such exposure regimes if the affected vegetation were found to be ecologically and/or economically important.

In its discussion of the secondary standard, the second draft PA (EPA, 2023) refers to the work of Lefohn and Foley (1992) and Lefohn et al. (1997). Lefohn and Foley (1992) noted that the NCLAN experimental protocol applied (1) incremental and (2) proportional additions that resulted in some of the treatments experiencing elevated O₃ concentrations, whose presence affected the form of the exposure-response models developed by the various investigators. At many of the treatment levels, the magnitude of the various cumulative indices appeared to be influenced by the elevated concentrations that resulted from the NCLAN artificial exposure protocols. Some, but not all, of the exposure treatments using the NCLAN protocols attempted to mimic O₃ exposures experienced under ambient conditions. Lefohn and Foley (1992) noted that when high hourly average concentrations were present in an ambient exposure regime, single-parameter cumulative indices appeared to relate O₃ exposures with vegetation growth reductions. However, as noted by the authors, if high hourly average concentrations were not necessarily present in the ambient exposures, then the application of a single-parameter exposure for protecting vegetation might provide inconsistent results if the exposure-response relationships were based on experiments that contained frequent occurrences of hourly average O₃ concentrations ≥ 0.10 ppm in the experimental exposure regimes. Lefohn and Foley (1992) recommended that an exposure index, such as the W126, should be combined with other parameters to accurately quantify the presence of the high hourly average concentrations occurring in the NCLAN experiments. The authors noted that because of the importance of describing the presence or absence of the higher hourly average concentrations, the combination of exposure parameters used to describe those regimes responsible for vegetation effects should focus on characterizing the upper tail of the hourly average O₃ concentration distribution curve. A similar recommendation was made by Lefohn and Foley (1993).

As noted by Lefohn and Foley (1992), it is important to identify the specific O₃ regimes responsible for the adverse vegetation effects (e.g., biomass or yield loss). Lefohn et al., (1997) described an approach for combining experimental exposure-response effects data for deciduous and coniferous seedlings and/or trees with characterized O₃ ambient exposure data for a local area. The authors identified both the magnitude of the cumulative exposure indices used in their study and the number of hourly average values ≥ 0.10 ppm. Similar to Lefohn and Foley (1992), the authors were concerned if ambient air quality data were used to predict growth losses using exposure-response relationships derived from experimental data that contained numerous occurrences of hourly average O₃ concentrations ≥ 0.10 ppm, biomass loss might possibly be overestimated. Fig. 4-3, a reproduction of Table 3 in Lefohn et al. (1997), shows the 24-h 92-day estimated biomass losses at the 10% level for experimental data available for some tree species in the Southern Appalachian region of the United States. The number of estimated exposure hours ≥ 0.10 ppm over the period is also provided.

Table 3. Ozone exposure levels as a function of tree response category

Tree response category	W126 (ppm h)		Exposure hours \geq 0.10 ppm
Minimal	\geq 0	and	\geq 0
Level 1 (only high sensitive species affected) (e.g. black cherry)	\geq 5.9	and	\geq 6
Level 2 (moderately sensitive species affected) (e.g. yellow-poplar)	\geq 23.8	and	\geq 51
Level 3 (resistant species affected) (e.g. red oak)	\geq 66.6	and	\geq 135

Figure 4-3. The ozone W126 and N100 24-h exposure levels as a function of tree response category. (Lefohn et al., 1997).

The use of the two-parameter (i.e., W126 and N100) exposure index discussed by Lefohn et al. (1997) was applied by the Forest County Potawatomi Community (FCPC). Recognizing the importance of applying the W126 exposure index in establishing its air quality related values, combined with an index that included the number of hours \geq 100 ppb to protect vegetation, the FCPC (<https://lnr.fcpotawatomi.com/current-vegetation-w126-n100/>) noted that

The US EPA considers the W126 the most biologically relevant cumulative, seasonal form appropriate to consider as a secondary ozone standard in the context of the Agency's 2008 and 2015 ozone rulemaking (U.S. EPA, 2008; US EPA, 2010; US EPA, 2013; US EPA, 2014a; US EPA, 2014b; US Federal Register, 2015) to protect sensitive natural vegetation and ecosystems in specially designated areas. The Agency used the W126 index to determine the secondary standard for ozone in its 2015 revision to the national ozone standard, concluding that vegetation would be protected by a standard that generally limits cumulative seasonal exposures to a 3-year average W126 value of 17 ppm-hours (ppm-hrs) or lower. But rather than setting the secondary standard using a W126 value, EPA determined that an 8-hour standard level of 0.070 ppm (the primary standard) was equivalent and would provide the same protective levels as the W126 value of 17 ppm-hrs.

FCPC believes that the W126 index is most suitable to its Class I area. Based on the review of the available scientific literature, FCPC chose to use the W126 exposure index accumulated over a 24-h period for a 3-month period as one of two indices to protect vegetation. FCPC determined that using the 24-hour W126 index, rather than the 12-hour W126 suggested by EPA, provides greater protection of FCPC's vegetation resources.

Due to the importance of peak concentrations affecting the results in the experiments used to develop exposure-response relationships for assessing vegetation, FCPC decided to use the N100 metric (i.e., number of hourly average concentrations equal to or greater than 100 ppb [0.10 ppm] for the same 3-month period) as the second index to be used in combination with the W126 exposure index for assessing vegetation impact.

As noted in its Air Quality Related Values (AQRVs) Implementation Frameworks and Assessment and Modeling Protocols document (https://lnr.fcpotawatomi.com/wp-content/uploads/2015/01/FCPC-AQRV-Implementation_modeling-protocols-combined-doc.pdf),

Air quality related values (AQRVs) are resources sensitive to air quality and include a wide array of vegetation, soils, water, fish and wildlife, and visibility. While Mandatory Federal Class I areas are required to identify AQRVs, non-Federal Class I areas are not, however, FCPC did so in fulfilling its obligation under the terms of the 1999 Class I Final Agreement between FCPC and the State of Wisconsin. In May, 2015, FCPC finalized the adoption of selected Air Quality Related Values (AQRVs) and related Threshold Effect Levels (TELs) (see Table 1 FCPC AQRV Threshold Effects Levels).

The Forest County Potawatomi Community established the O₃ thresholds for the 3-year average of the 3-month (June, July, and August) 24-h cumulative W126 value at 7.0 ppm-hrs and the 3-year average of the 3-month (June, July, and August) number of hours ≥ 100 ppb (N100) at 4 for vegetation (<https://lnr.fcpotawatomi.com/wp-content/uploads/2015/01/FCPC-AQRV-Threshold-Effects-Levels-Webpage-version1.pdf>).

An important consideration of the Forest County Potawatomi Community was the time of day of accumulation of the W126 and N100 exposures. A large number of species have varying degrees of nocturnal stomatal conductance (Musselman and Minnick, 2000). Although nocturnal stomatal conductance is much lower compared to daytime conductance, stomatal conductance coupled with enhanced O₃ exposures can possibly affect vegetation injury and growth if these two are matched with low nighttime detoxification potential (Heath et al., 2009). The implication is that the additional evidence of O₃ uptake at night may interfere with recovery and the FCPC believed that this evidence should be considered in establishing an appropriate time for accumulation. Vegetation growing in remote, high-elevation, and rural areas is more likely to experience some conductance, enhanced O₃ concentrations, and low defense capability during the nighttime and early morning hours. Thus, the FCPC focused on the 24-h period because it was believed that accumulating exposure over daylight hours (i.e., 12-hour periods) may not be as appropriate as accumulating over a 24-h period for assessing vegetation effects.

Another important consideration associated with the Forest County Potawatomi Community's decision to restricting the determination to the summer months (June, July, and August) was the concern that elevated 24-h, W126 exposures during the springtime might be associated with stratospheric-tropospheric exchange events. The FCPC did not desire that these occurrences be considered when assessing possible vegetation impacts associated with anthropogenic emissions. The literature has discussed the potential importance of stratospheric O₃ enhancing surface O₃ concentrations with the result that vegetation may be affected with acute injuries (see for example Skelly, 2000). The FCPC noted that it might not be possible to prevent vegetation injury resulting from exposure to O₃ in the spring associated with stratospheric O₃ in the FCPC Class I area or other areas in the region. However, by minimizing increases in O₃ exposures from anthropogenic sources, the FCPC believed that vegetation foliar injury and damage could be kept at a minimum.

The second draft PA (EPA, 2023, page 4-46) discussed results from Davis and Orendovici (2006). During 1993-1996 and 2001-2003, the investigators evaluated the percentage of plants (incidence) exhibiting O₃-induced foliar symptoms on vegetation within a National Wildlife Refuge located along the Atlantic Ocean coast of New Jersey. Incidence varied among plant species and years. The greatest incidence of O₃ symptoms, across all plant species, occurred in 1996, followed by 2001 > 1995 > 1994 > 1993 > 2003 > 2002. As noted in the second draft PA (EPA, 2023, pages 4-46 and 4-47), Davis and Orendovici (2006) reported that among the statistical models investigated, the model with the best fit to visible foliar injury incidence data was found to be one that included N100 and W126 indices, as well as a drought index. The authors concluded that their study indicated that of the various measures of O₃ currently employed at the time in the U.S., the W126 exposure index, coupled with the N100 metric, was the most strongly related to incidence of O₃ symptoms in the field.

It is important to note that the second draft PA (EPA, 2023, page 4-122) concludes for visible foliar injury the following:

In considering the available information that might inform the Administrator's judgments regarding visible foliar injury, we note a paucity of established approaches to inform the Administrator's judgment of a magnitude, severity or extent of visible foliar injury related effects appropriately concluded to be known or anticipated to cause adverse effects to the public welfare. However, some general conclusions or observations may be supported. For example, based on the available evidence and associated quantitative analyses, we have less confidence and greater uncertainty in the existence of adverse public welfare effects with lower O₃ exposures. More specifically, as discussed in the prior sections, the available information suggests that O₃ air quality associated with W126 index values below 25 ppm-hrs (in a single year), particularly when in combination with infrequent occurrences of hourly concentrations at or above 100 ppb, is not likely to pose a risk of visible foliar injury in natural areas of an extent and severity that might reasonably be considered to be of public welfare significance.

The statement indicates that extremely high O₃ exposures can result in more frequent occurrences of visible foliar injury than when more moderate exposures occur, and that the EPA has less confidence and greater uncertainty in the existence of a relationship between visual foliar injury and O₃ at lower exposures. Developing a reliable relationship between visual foliar injury and O₃ exposure has been a difficult challenge for vegetation researchers. Some O₃ exposures result in visible foliar injury to the plant without growth reduction; other exposures result in growth reduction and decrease in productivity without visible foliar injury, whereas some exposures result in both. Hildebrand et al. (1996) noted that soil moisture content plays an important role in the sensitivity of vegetation to O₃ exposure. The second draft PA (EPA, 2023, page 4-89) notes that

With regard to visible foliar injury, as in the 2015 review, we lack established E-R relationships that would quantitatively describe relationships between visible foliar injury (occurrence and incidence, as well as injury severity) and O₃

exposure, as well as factors influential in those relationships, such as soil moisture conditions.

In summary, as noted in the second draft PA (EPA, 2023), Lefohn and Foley (1992) and Lefohn et al. (1997) documented the frequency of the occurrence of elevated O₃ concentrations in some of the NCLAN and tree seedling experiments. Since the reporting of this phenomenon in the literature, many have discussed the use of the W126 exposure index, coupled with the N100 metric, so that the exposure-response models developed in the crop and tree seedling experiments are useful for predicting vegetation effects using current ambient O₃ exposures. As noted earlier in this subsection, *it is important to identify, using research results appropriate for establishing the O₃ secondary NAAQS, those exposure regimes responsible for causing adverse vegetation effects*. In some cases, those O₃ exposure regimes responsible for eliciting adverse effects may be associated with an extremely low number of occurrences of hourly average O₃ concentrations ≥ 100 ppb or perhaps, in some cases, even no occurrences ≥ 100 ppb. For the case of the Forest County Potawatomi Community, the tribe finalized the adoption of its selected O₃ Air Quality Related Values (AQRVs) and related Threshold Effect Levels (TELs) using the 3-year average of the 3-month (June, July, and August) 24-h cumulative W126 value at 7.0 ppm-hrs, coupled with the 3-year average of the 3-month (June, July, and August) number of hours ≥ 100 ppb at 4. Depending upon the specified vegetation at risk (e.g., most sensitive) and the experimental O₃ regimes used for developing exposure-response relationships for this vegetation, it is possible that lower W126 values than those applied by the Forest County Potawatomi Community, coupled with lower number of occurrences of hourly average O₃ concentrations ≥ 100 ppb than applied by the Forest County Potawatomi Community, could be established as a secondary O₃ NAAQS.

4.3 Identifying W126 Coupled with N100 Exposures Estimated to Result in Biomass or Yield Reduction Under Current Ozone Exposures.

As noted in the second draft PA (EPA, 2023), the number of hourly averaged O₃ concentrations ≥ 100 ppb has been an additional metric applied in some studies that focused on O₃ effects on vegetation as an indicator of peak hourly O₃ concentrations (e.g., Lefohn and Foley, 1992; Lefohn et al., 1997, Smith, 2012; Davis and Orendovici, 2006; Kohut, 2007). As discussed in Subsection 4.2, although the weighting scheme for the W126 metric (Fig. 4-2) includes all hourly average O₃ concentrations, including hourly average O₃ concentrations ≥ 100 ppb, it is not a necessary requirement that hourly average O₃ concentrations ≥ 100 ppb must be present for vegetation damage to occur. For example, Lee et al. (2022) reported that for the most sensitive species in their analysis, these species experienced a biomass loss of 5% at a W126 of 2.5–9.2 ppm-hrs with N100 values ranging from 0 to 7 at those W126 exposures. Fig. 4-4 (Fig. 4-10 in the second draft PA (EPA, 2023), page 4-67) illustrates the 3-year average of the number of hourly average O₃ concentrations ≥ 100 ppb (N100) values at O₃ monitoring sites with valid design values for 2018-2020. Of the 1105 monitoring sites, almost 74% do not experience N100 values for the 2018-2020 period. Fig. 4-5 (Fig. 4F-13, page 4F-22) in the second draft PA (EPA, 2023) illustrates the Trend in N100 values from 2000 to 2020 based on data from 808 U.S. O₃ monitoring sites.

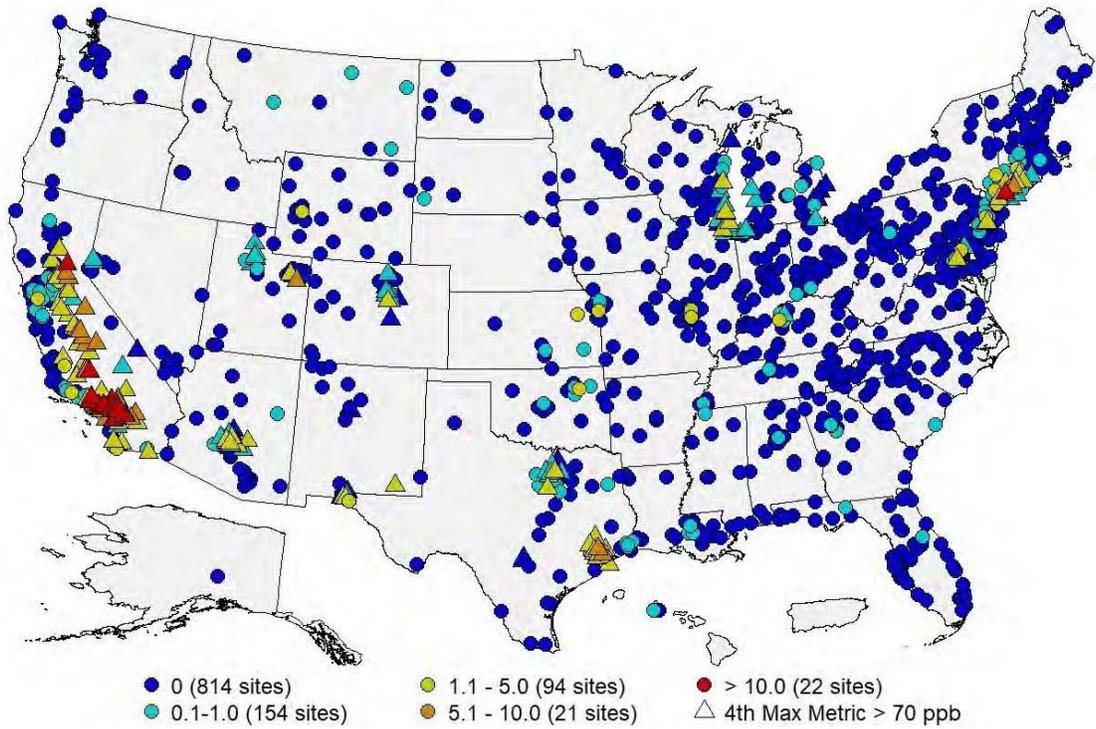


Figure 4-10. N100 values (2018-2020 average) at monitoring sites with valid design values.

Figure 4-4. N100 values at monitoring sites with valid design values (2018-2020 average). (EPA, 2023, page 4-67).

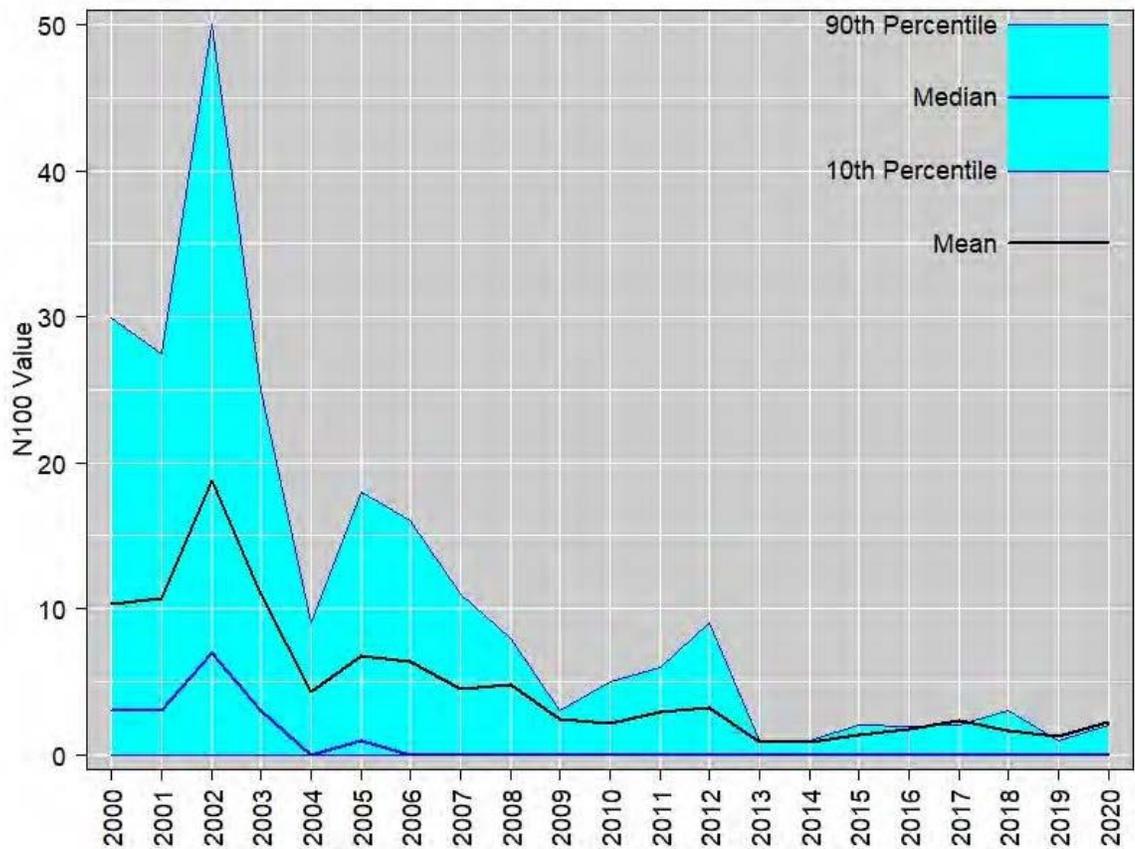


Figure 4F-13. Trend in N100 values from 2000 to 2020 based on data from 808 U.S. O₃ monitoring sites

Figure 4-5. Trend in N100 values from 2000 to 2020 based on data from 808 U.S. O₃ monitoring sites. (EPA, 2023, page 4F-22).

As the second draft PA (EPA, 2023, page 4F-21 notes, the median N100 values in the U.S. have been zero since 2006, meaning over half of the monitoring sites have experienced N100 values of zero. The mean N100 value has decreased from more than ten in 2000-2002 to less than two in recent years, a decline of more than 80%. The 90th percentile value has decreased at an even faster rate. Thus, in recent years, a little less than 75% of the monitors included in the second draft PA (EPA, 2023) analysis experience no N100 values.

The second draft PA (EPA, 2023, page 4F-3) notes that the W126 metric values were calculated using the hourly O₃ concentration data in parts per million. For daytime hours (defined as the 12-hour period from 8:00 AM to 8:00 PM Local Standard Time each day), the hourly concentration values at each O₃ monitoring site were weighted using the following equation:

$$\text{Weighted O}_3 = \text{O}_3 / (1 + 4403 \cdot \exp(-126 \cdot \text{O}_3)).$$

The weighted values were summed over each calendar month, then adjusted for missing data as described in the second draft PA (EPA, 2023, page 4F-3). Monthly W126 index values were not calculated for months where fewer than 75% of the possible daytime hourly concentrations were available. Next, moving 3-month sums were calculated from the monthly index values, and the highest of these 3-month sums was determined to be the annual W126 index. Three-month periods spanning multiple years (e.g., November to January, December to February) were not considered in the calculations. From the description of the W126 calculation presented in the second draft PA (EPA, 2023) 3-month W126 values were determined for the January-March, February-April, March-May, April-June, May-July, June-August, July-September, August-October, September-November, and October-December periods and the annual maximum 3-month value was then identified. Similarly, the second draft PA (EPA, 2023, page 4F-3) notes that the N100 values were calculated as the maximum number of hours with an hourly O₃ concentration of 100 ppb or greater in the three consecutive calendar months yielding the highest number each year. For the analyses described in the second draft of the PA (EPA, 2023), although not mentioned in the document, the 3-month period for the three consecutive calendar months when the maximum number of hours with an hourly O₃ concentration of 100 ppb or greater occurred apparently did not necessarily match in some cases the same 3-month period when the highest 3-month W126 value was experienced. Similarly, the month in which the annual 4th highest maximum daily average 8-hour concentration occurred did not necessarily include the 3-month periods when either the 3-month maximum N100 values or the 3-month maximum W126 levels occurred.

As noted in Section 3 of these comments, actual data show that the highest O₃ exposures occur at some sites across the U.S. during the springtime (March to mid-June). Using hourly average O₃ data from 57 National Park Service Parks, the EPA in the 2014 PA (EPA, 2014c) (Welfare Appendix, page 7A-12) provided the highest 3-month W126 values and the timeframe corresponding to those W126 exposures for the Parks with O₃ monitors for the period 2006-2010. Table 7A-2 as was provided in Section 3 is shown in the pages below. Note that several of the O₃ monitors in the Parks experienced their highest W126 exposures during the spring months (defined as March, April, May or April, May, June). Similarly, as noted in Section 3, Neufeld et al. (2019) analyzed O₃ trends from 1989 to 2016 for six monitoring sites in and adjacent to Great Smoky Mountain NP (GRSM) and ranging in elevation from 564m to 2030m. The W126 exposures increased between the years 1989–~2002 and substantially decreased afterwards. As emissions were reduced at most of the six sites analyzed by Neufeld et al. (2019), the maximum 3-month W126 exposures shifted from mid-summer to the April–June period. Neufeld et al. (2019) noted that for the sites analyzed in their analysis, the shift in seasons following reduction of the peak concentrations may reflect a greater percentage of the remaining higher O₃ exposures being associated with background processes, such as stratospheric intrusions.

While the EPA in the second draft PA (EPA, 2023) has focused on the 3-month period when the highest W126 exposures occur, the 3-month periods of March-May and April-June may not be appropriate periods for assessing the risk of O₃ exposures affecting tree seedling biomass loss. The vegetation may not be as susceptible to O₃ biomass loss as during the warmer months. The observations reported by the EPA (2014c) and Neufeld et al. (2019) suggest that the EPA limit its characterization of W126 and N100 values to those periods when O₃ exposures have their greatest impacts on tree seedling biomass loss. As indicated in the second draft PA

(EPA, 2023, pages 4A-21 – 4A-26 and pages 4A-28 – 4A-30), most of the tree seedling experiments began in June and were terminated in September/October (please see Tables 4A-7 and 4A-8 in the second draft PA).

Table 7A-2 Ozone Exposure in 57 O₃ Monitors Located in Parks*

Monitor site ID	Park Name	W126					3-Month Timeframe for W126				
		2006	2007	2008	2009	2010	2006	2007	2008	2009	2010
230090102	Acadia National Park	10.59	7.89	7.64	7.02	5.24	MJJ	AMJ	MJJ	MAM	MAM
230090103	Acadia National Park	6.37	6.41	4.72	5.21	4.13	MJJ	AMJ	MJJ	MAM	MAM
311651001	Agate Fossil Beds National Monument	--	8.27	12.76	5.85	--	--	JAS	MJJ	JJA	--
460710001	Badlands National Park	--	--	2.23	2.54	3.85	--	--	JAS	AMJ	JJA
460711001	Badlands National Park	16.74	8.01	--	--	--	JJA	JJA	--	--	--
480430101	Big Bend National Park	11.62	10.60	10.55	8.62	8.47	AMJ	MAM	MAM	MAM	MAM
370110002	Blue Ridge Parkway	9.88	11.46	8.81	4.71	8.19	AMJ	AMJ	AMJ	AMJ	AMJ
490370101	Canyonlands National Park	18.06	16.93	17.06	12.23	13.24	MJJ	MJJ	AMJ	MAM	AMJ
250010002	Cape Cod National Seashore	13.47	13.16	12.89	5.25	7.03	MJJ	MJJ	MJJ	AMJ	MJJ
350153001	Carlsbad Caverns National Park	--	8.65	17.50	11.37	7.09	--	AMJ	AMJ	MJJ	AMJ
160310001	City of Rocks National Reserve	--	--	--	--	6.02	--	--	--	--	JJA
80771001	Colorado National Monument	--	11.61	15.04	4.13	8.75	--	JJA	MJJ	JAS	AMJ
450790021	Congaree National Park	12.31	10.78	9.45	3.97	6.32	MAM	MAM	MAM	FMA	MAM
450210002	Cowpens National Battlefield	14.30	7.87	16.05	3.24	8.81	MJJ	AMJ	JJA	FMA	MAM
160230101	Craters of the Moon National Monument	--	10.17	10.88	5.68	7.82	--	JJA	MJJ	MAM	JAS
210131002	Cumberland Gap National Historical Park	--	18.36	10.12	3.58	7.31	--	MJJ	MJJ	MJJ	MJJ
60270101	Death Valley National Park	29.18	32.55	25.57	15.30	10.61	MJJ	MJJ	MJJ	JJA	JAS
560111013	Devil's Tower National Monument	--	--	7.09	5.42	5.44	--	--	JAS	JAS	JJA
490471002	Dinosaur National Monument	--	10.33	13.34	8.39	13.80	--	MJJ	MJJ	MJJ	MJJ
300298001	Glacier National Park	2.90	2.29	3.98	3.53	2.44	JJA	MAM	MAM	AMJ	AMJ
300351001	Glacier National Park	--	--	--	4.91	3.93	--	--	--	MJJ	MJJ
40058001	Grand Canyon National Park	21.66	18.68	17.02	10.10	14.95	MJJ	AMJ	AMJ	JJA	AMJ
320330101	Great Basin National Park	15.54	15.79	16.94	10.19	11.44	JJA	MJJ	MJJ	AMJ	AMJ
370870036	Great Smoky Mountains National Park	11.46	13.35	11.50	4.59	7.89	AMJ	AMJ	AMJ	AMJ	AMJ
470090102	Great Smoky Mountains National Park	12.97	12.69	10.44	5.31	10.27	AMJ	MAM	AMJ	MAM	MAM
471550101	Great Smoky Mountains National Park	18.87	20.66	14.15	9.03	15.16	AMJ	AMJ	MJJ	MAM	MAM
471550102	Great Smoky Mountains National Park	19.59	23.51	16.23	7.32	11.94	MJJ	JJA	MJJ	MJJ	ASO

7A-12

Source: EPA (2014c).

Monitor site ID	Park Name	W126					3-Month Timeframe for W126				
		2006	2007	2008	2009	2010	2006	2007	2008	2009	2010
180890022	Indiana Dunes National Lakeshore	8.79	12.21	3.66	2.42	3.91	JJA	AMJ	JAS	MJJ	JJA
60650008	Joshua Tree National Park	24.36	19.97	27.43	19.66	23.39	AMJ	AMJ	MJJ	AMJ	AMJ
60651004	Joshua Tree National Park	--	26.37	30.05	18.81	20.47	--	MJJ	AMJ	JJA	JJA
60719002	Joshua Tree National Park	55.48	52.46	50.99	39.93	43.92	MJJ	MJJ	JJA	JJA	JJA
60893003	Lassen Volcanic National Park	18.97	15.10	18.98	7.64	9.63	JAS	JJA	MJJ	JJA	JAS
80830101	Mesa Verde National Park	23.44	17.57	13.41	15.05	11.94	MJJ	MJJ	AMJ	JJA	AMJ
60711001	Mojave National Preserve	--	28.50	38.92	19.91	19.39	--	MJJ	MJJ	JAS	JJA
530530012	Mount Rainier Wilderness	3.19	3.30	1.18	2.20	1.86	MAM	MAM	JAS	FMA	MAM
530090016	Olympic National Park	--	--	--	--	0.52	--	--	--	--	JAS
530091004	Olympic National Park	--	0.28	0.93	--	--	--	JAS	JAS	--	--
482731001	Padre Island National Seashore	--	8.19	3.66	--	--	--	AMJ	AMJ	--	--
40170119	Petrified Forest National Park	19.16	16.60	19.40	9.04	12.71	AMJ	AMJ	AMJ	AMJ	AMJ
60690003	Pinnacles National Monument	17.14	14.85	19.78	11.41	9.79	JAS	AMJ	MJJ	JAS	JAS
40190021	Saguaro National Park	19.57	17.06	20.13	11.01	15.31	MJJ	MJJ	AMJ	MAM	AMJ
360910004	Saratoga National Historical Park	6.68	10.38	9.26	5.40	5.98	JJA	MJJ	AMJ	MAM	MJJ
311570005	Scotts Bluff National Monument	--	--	--	--	6.20	--	--	--	--	JJA
61070006	Sequoia-Kings Canyon National Park	50.09	53.38	57.24	29.13	26.93	JJA	JJA	JJA	JAS	JAS
61070009	Sequoia-Kings Canyon National Park	66.07	62.88	56.91	55.51	53.79	JAS	JJA	MJJ	JAS	JAS
511130003	Shenandoah National Park	16.43	14.40	12.07	7.63	10.84	AMJ	AMJ	AMJ	MAM	JAS
380070002	Theodore Roosevelt National Park	7.71	5.54	5.55	3.95	4.19	JAS	JJA	AMJ	AMJ	AMJ
380530002	Theodore Roosevelt National Park	9.45	6.29	6.31	4.22	5.17	JJA	JJA	MJJ	AMJ	MAM
40070010	Tonto National Monument	26.39	23.24	25.40	13.67	16.90	MJJ	MJJ	AMJ	AMJ	AMJ
271370034	Voyageurs National Park	5.33	5.19	3.86	4.94	7.66	AMJ	AMJ	MAM	MAM	MAM
460330132	Wind Cave National Park	20.52	12.20	5.92	5.75	5.61	JJA	JJA	JJA	JJA	JAS
560391011	Yellowstone National Park	12.98	9.96	8.84	7.63	11.54	AMJ	AMJ	MAM	MAM	AMJ
60430003	Yosemite National Park	33.78	29.68	42.51	25.70	27.34	JJA	MJJ	JJA	JAS	JAS
60431002	Yosemite National Park	--	12.60	10.03	--	--	--	AMJ	MJJ	--	--
60431003	Yosemite National Park	--	11.61	--	--	--	--	JAS	--	--	--
60431004	Yosemite National Park	--	6.95	15.52	6.58	9.43	--	MJJ	JJA	JAS	JAS
60431005	Yosemite National Park	--	--	27.83	5.18	14.28	--	--	JAS	JAS	JAS

*Nine parks have more than 1 monitor

Source: EPA (2014c).

The use of the W126 to characterize O₃ exposure concentrations regarding potential vegetation effects, particularly growth, has received strong support from CASAC in previous reviews (Henderson, 2006; Samet, 2010; Frey, 2014, Cox, 2020). The history of the use of the W126 as a secondary standard is summarized in the second draft PA (EPA, 2023, pages 1-9 – 1-11) as follows:

The EPA initiated the fourth periodic review of the air quality criteria and standards for O₃ and other photochemical oxidants with a call for information in September 2000 (65 FR 57810, September 26, 2000). In 2007, the EPA proposed to revise the level of the primary standard within a range of 0.075 to 0.070 ppm (72 FR 37818, July 11, 2007). The EPA proposed to revise the secondary standard either by setting it identical to the proposed new primary standard or by setting it as a new seasonal standard using a cumulative form. Documents supporting these proposed decisions included the 2006 AQCD (U.S. EPA, 2006) and 2007 Staff Paper (U.S. EPA, 2007) and related technical support documents. The EPA completed the review in March 2008 by revising the levels of both the primary and secondary standards from 0.08 ppm to 0.075 ppm while retaining the other elements of the prior standards (73 FR 16436, 15 March 27, 2008).

In May 2008, state, public health, environmental, and industry petitioners filed suit challenging the EPA's final decision on the 2008 O₃ standards. On September 16, 2009, the EPA announced its intention to reconsider the 2008 O₃ standards,⁹ and initiated a rulemaking to do so. At the EPA's request, the court held the consolidated cases in abeyance pending the EPA's reconsideration of the 2008 decision.

In January 2010, the EPA issued a notice of proposed rulemaking to reconsider the 2008 final decision (75 FR 2938, January 19, 2010). In that notice, the EPA proposed that further revisions of the primary and secondary standards were necessary to provide a requisite level of protection to public health and welfare. The EPA proposed to revise the level of the primary standard from 0.075 ppm to a level within the range of 0.060 to 0.070 ppm, and to revise the secondary standard to one with a cumulative, seasonal form. At the EPA's request, the CASAC reviewed the proposed rule at a public teleconference on January 25, 2010 and provided additional advice in early 2011 (Samet, 2010, Samet, 2011). Later that year, in view of the need for further consideration and the fact that the Agency's next periodic review of the O₃ NAAQS required under CAA section 109 had already begun (as announced on September 29, 2008),¹⁰ the EPA decided to consolidate the reconsideration with its statutorily required periodic review.¹¹

In light of the EPA's decision to consolidate the reconsideration with the ongoing periodic review, the D.C. Circuit proceeded with the litigation on the 2008 O₃ NAAQS decision. On July 23, 2013, the court upheld the EPA's 2008 primary standard, but remanded the 2008 secondary standard to the EPA (*Mississippi v. EPA*, 744 F.3d 1334 [D.C. Cir. 2013]). With respect to the primary standard, the court first rejected arguments that the EPA should not have lowered the level of

the existing primary standard, holding that the EPA reasonably determined that the existing primary standard was not requisite to protect public health with an adequate margin of safety, and consequently required revision. The court went on to reject arguments that the EPA should have adopted a more stringent primary standard. ***With respect to the secondary standard, the court held that the EPA's explanation for the setting of the secondary standard identical to the revised 8-hour primary standard was inadequate under the CAA because the EPA had not adequately explained how that standard provided the required public welfare protection*** (emphasis added).

At the time of the court's decision, the EPA had already completed significant portions of its next statutorily required periodic review of the O₃ NAAQS. This review had been formally initiated in 2008 with a call for information in the Federal Register (73 FR 56581, September 29, 2008). In late 2014, based on the ISA, Risk and Exposure Assessments (REAs) for health and welfare, and PA¹² developed for this review, the EPA proposed to revise the 2008 primary and secondary standards by reducing the level of both standards to within the range of 0.070 to 0.065 ppm (79 FR 75234, December 17, 2014).

The EPA's final decision in this review was published in October 2015, establishing the now-current standards (80 FR 65292, October 26, 2015). In this decision, based on consideration of the health effects evidence on respiratory effects of O₃ in at-risk populations, the EPA revised the primary standard from a level of 0.075 ppm to a level of 0.070 ppm, while retaining all the other elements of the standard (80 FR 65292, October 26, 2015). The EPA's decision on the level for the standard was based on the weight of the scientific evidence and quantitative exposure/risk information. The level of the secondary standard was also revised from 0.075 ppm to 0.070 ppm based on the scientific evidence of O₃ effects on welfare, particularly the evidence of O₃ impacts on vegetation, and quantitative analyses available in the review.¹³ The other elements of the standard were retained. This decision on the secondary standard also incorporated the EPA's response to the D.C. Circuit's remand of the 2008 secondary standard in *Mississippi v. EPA*, 744 F.3d 1344 (D.C. Cir. 2013). The 2015 revisions to the NAAQS were accompanied by revisions to the data handling procedures, and the ambient air monitoring requirements¹⁴ (80 FR 65292, October 26, 2015).¹⁵

After publication of the final rule, a number of industry groups, environmental and health organizations, and certain states filed petitions for judicial review in the D.C. Circuit. The industry and state petitioners argued that the revised standards were too stringent, while the environmental and health petitioners argued that the revised standards were not stringent enough to protect public health and welfare as the Act requires. On August 23, 2019, the court issued an opinion that denied all the petitions for review with respect to the 2015 primary standard while also concluding that the EPA had not provided a sufficient rationale for aspects of its decision on the 2015 secondary standard and

remanding that standard to the EPA (*Murray Energy Corp. v. EPA*, 936 F.3d 597 [D.C. Cir. 2019]).

In its August 2019 decision, the D.C. Circuit addressed arguments regarding the adequacy of the EPA Administrator's 2015 decision to use a 3-year average of the W126 index as the benchmark and use the current form of the standard instead of the CASAC-recommended W126 metric. The Court noted:

The Environmental Petitioners raise two challenges pertaining to the W126 index, both as a benchmark and as a form and averaging time. First, they argue that EPA impermissibly departed from CASAC's advice by setting the secondary standard level using a three-year average W126 benchmark without lowering the level to protect against single-year exposures associated with median annual tree growth loss of 6%, which CASAC had advised was "unacceptably high." J.A. 518. They also contend that EPA arbitrarily disregarded CASAC's advice to adopt the W126 index as the form and averaging time for the secondary standard. We remand to EPA on the first issue and do not reach the second.

The Three-Year Average Benchmark. CASAC advised EPA that basing the secondary standard on a "single-year period" would provide "more protection for annual crops and for the anticipated cumulative effects on perennial species" than a three-year average. J.A. 518. It explained that EPA's proposal to use a three-year averaging period was "not supported by the available data," J.A. 536, and that if EPA chose to "base the secondary standard on a three-year averaging period," then "the level of the standard should be revised downward such that the level for the highest three-month summation in any given year of the three-year period would not exceed [its] scientifically recommended" range of single-year, W126 exposure levels, J.A. 518. This was necessary, CASAC explained, to "protect against single unusually damaging years that will be obscured in the average." J.A. 536.

EPA argues it gave effect to CASAC's recommendation by using a three-year average benchmark of 17 ppm-hrs, which, after adjusting for the cottonwood data, was "somewhat below" the 19 ppm-hrs associated with 6% annual growth loss that CASAC had advised was "unacceptably high." 80 Fed. Reg. at 65,406-07 (quoting J.A. 518). But CASAC had advised a maximum level associated with 5.2% annual biomass loss, see J.A. 631, and it expressly cautioned that 6% median growth loss in a single year was unacceptable, see J.A. 518. EPA's use of a benchmark that averages out to less than 6% biomass loss over three years does not accord with CASAC's advice. Indeed, as commenters informed EPA during the rulemaking, see J.A. 1836-40, EPA's own air quality data suggests that many large national parks and wilderness areas that have met EPA's chosen three-year average 17 ppm-hrs benchmark— areas that Congress considers significant to the public welfare, see 42 U.S.C. §§ 7470(2), 7472(a)—have meanwhile recorded single-year W126 values at and above 19 ppm-hrs, which is associated with

“unacceptably high” annual biomass loss of 6% and higher. See J.A. 1061-64; 80 Fed. Reg. at 65,391. EPA critiques that data as marred by outdated handling procedures, but the agency acknowledged that other data derived through updated procedures produced results “similar to” those showing harmful exposure spikes. J.A. 1213. Critically, EPA points us to no data or analysis (based on new or old procedures) suggesting that the chosen benchmark prevents single seasonal exposures of 19 ppm-hrs or higher. In short, EPA has not demonstrated how its chosen benchmark protects against “unusually damaging years that will be obscured in the average.” J.A. 536.

EPA alternatively defends its decision to use the three-year-average benchmark as providing a focus on public welfare effects of “potentially greater” significance than effects “associated with a single year” of exposure. 80 Fed. Reg. at 65,404. This position, however, is inconsistent with EPA’s other actions. In establishing the secondary standard, for example, EPA heavily relied on data showing annual 6% median tree growth loss at 19 ppm-hrs, see *id.* at 65,406, and acknowledged the potential for a single season of high ozone exposure to “alter biomass allocation and plant reproduction in seasons subsequent to [that season’s] exposure,” thereby leading to “a negative impact on species regeneration in subsequent years,” *id.* at 65,371-72; see also J.A. 740-41. EPA additionally recognized that “ozone effects in plants are cumulative,” *id.* at 65,373 (quoting EPA, *Integrated Science Assessment 2-44* (2013)), meaning that the adverse vegetative effects from single, high-ozone years are not offset by subsequent low-ozone years. EPA has identified no contrary evidence in the record demonstrating why these single-year effects matter less than a three-year average.

We therefore remand this issue for EPA to either lower the standard to protect against unusually damaging cumulative seasonal exposures that will be obscured in its three-year average, or explain its conclusion that the unadjusted average is an appropriate benchmark notwithstanding CASAC’s contrary advice. Alternatively, EPA could adopt the single-year W126 exposure index as the form and averaging time, which would presumably moot any problems with the way it translated that index to use as a benchmark.

The Form and Averaging Time. CASAC also recommended that EPA use the single-year W126 index as the form and averaging time for the secondary standard. J.A. 518. EPA instead chose to retain the existing form and averaging time—the three-year average of the fourth-highest daily maximum eight-hour concentration. Adopting the W126 index as the form and averaging time was unnecessary, EPA explained, because the ozone exposure levels associated with the existing form and averaging time and a three-year average of the W126 index are “highly correlated,” especially at lower levels, and “future control programs designed to help meet a primary [ozone] standard based on the” current form and averaging time should “provide similar improvements in terms of the 3-year average of the annual W126 metric.” J.A. 1253; see also 80 Fed. Reg. at 65,400-01, 408-09.

The Environmental Petitioners argue that EPA did not justify its decision not to adopt the W126 index as the form and averaging time. We lack any basis to assess the reasonableness of EPA's actions, however, because a critical piece of the puzzle is missing. To review: EPA chose not to use the W126 index as the form and averaging time because it found that ozone exposure levels associated with the existing form and averaging time are "highly correlated" to a three-year average of the W126 index. But, as discussed, EPA never explained why it is reasonable to focus on a three-year average of the W126 index in the first place. Therefore, we cannot assess the relevance of the claim that the two are "highly correlated." EPA's reconsideration on remand of the three-year averaging issue should supply us with the information necessary to resolve this question, or, if EPA chooses to follow CASAC's advice to lower the standard to control for unusually high single years, potentially moot the Environmental Petitioners' concern that the current form tolerates even three-year average W126 levels higher than 17 ppm-hrs during periods when a 0.07 ppm, 8-hour level is met. Accordingly, we decline to reach this question.

The CASAC was clear on its recommendations to the Administrator concerning the use of the W126 exposure index as a standard to protect vegetation. In his 2010 comments on CASAC's recommendations for the reconsidered O₃ primary and secondary standards (as ordered by the Obama Administration in 2009), Samet (2010) stated for the secondary standard

CASAC also supports EPA's secondary ozone standard as proposed as a new cumulative, seasonal standard expressed as an annual index of the sum of weighted hourly concentrations (i.e., the W126 form), cumulated over 12 hours per day (8am to 8pm) during the consecutive 3-month period within the ozone season with the maximum index value, set as a level within the range of 7 to 15 ppm-hours. This W126 metric can be supported as an appropriate option for relating ozone exposure to vegetation responses, such as visible foliar injury and reductions in plant growth. We found the Agency's reasoning, as stated in the *Federal Register* notice of January 19, 2010, to be supported by the extensive scientific evidence considered in the last review cycle. In choosing the W126 form for the secondary standard, the Agency acknowledges the distinction between the effects of acute exposures to ozone on human health and the effects of chronic ozone exposures on welfare, namely that vegetation effects are more dependent on the cumulative exposure to, and uptake of, ozone over the course of the entire growing season (defined to be a minimum of at least three months). In this proposal, the Agency is responding to the clear need for a secondary standard that is different from the primary standard in averaging time, level and form.

In its 2014 letter to the Administrator (Frey, 2014) concerning the draft PA regarding the substitution of the W126 exposure index with the 8-h primary standard, CASAC indicated within its letter:

The CASAC concurs with the justification in this section that the form of the standard should be changed from the current 8-hr form to the cumulative W126 index and finds that the discussion provides an appropriate and sufficient rationale.

This section clearly demonstrates that ozone-induced injury may occur in areas that meet the current standard. As noted above, the correlative similarity between the current standard and a level of the W126 index of 15 ppm-hrs must not be interpreted to mean that just meeting the current standard is equivalent to just meeting a W126 level of 15 ppm-hrs. Most of the analyses found effects below 15 ppm-hrs (many at 10 or even 7 ppm-hrs). Based on review of relevant science, the CASAC concludes that the upper bound of the range that should be considered for the W126 standard should not exceed 15 ppm-hrs. The CASAC does not support a level higher than 15 ppm-hrs. Levels above 15 ppm-hrs should not be included in the revised PA as options for an alternate secondary standard. For example, at 17 ppm-hrs, the median tree species has 6% relative biomass loss, and the median crop species has over 5% loss. These levels are unacceptably high.

The CASAC does not recommend the use of a three-year averaging period. We favor a single-year averaging period, which will provide more protection for annual crops and for the anticipated cumulative effects on perennial species. The scientific analyses considered in this review, and the evidence upon which they are based, are from single-year results. If a 3-year averaging period is established, then the upper limit will need to be reduced to protect against one-year ozone peaks. We consider this further in the response to charge questions for Chapter 6.

The suggestion in Section 6.2 to use a 3-year averaging period is not supported by the available data. We have not supported it in the past nor do we support it here. The primary justification for a 3-year averaging period is to improve the program stability of the classification of regions as being in or out of compliance. The proposed form includes a 3-month period, so it is not nearly as sensitive to extreme events as an hourly or 8-hour averaging period. The case has not been made that welfare benefits from the stability of a 3-year average are greater than those from using the biologically relevant 1-year value. If a 3-year averaging period is implemented, it should be at a lower level than a single-year standard to protect against single unusually damaging years that will be obscured in the average.

In reaching its scientific judgment regarding the indicator, form, averaging time, and range of levels for a revised secondary standard, the CASAC has focused on the scientific evidence for the identification of the kind and extent of adverse effects on public welfare. The CASAC acknowledges that the choice of a level within the range recommended based on scientific evidence is a policy judgment under the statutory mandate of the Clean Air Act. Specifically, the Clean Air Act grants discretion to the Administrator to specify a standard that is "requisite to protect the public welfare from any known or anticipated adverse effects

associated with the presence of [the] pollutant in the ambient air” (Section 302(h), 42 U.S.C., §7602(h)). As a policy recommendation, separate from its advice above regarding scientific findings, the CASAC advises that a level of 15 ppm-hrs is requisite to protect crop yield loss, but that lower levels provide additional protection of crop yield loss. Furthermore, there are specific economically significant crops that may not be protected at 15 ppm-hrs but that would be protected at lower levels. Based on scientific judgment of CASAC, a level of 10 ppm-hrs is required to reduce foliar injury. A level of 7 ppm-hours is protective of relative biomass loss for trees. Furthermore, 7 ppm-hours offers additional protection against crop yield loss and foliar injury. Therefore, 7 ppm hours is protective of ecosystem services. Thus, lower levels from the recommended range offer a greater degree of protection of more endpoints than do higher levels from the range.

If, as a policy matter, the Administrator prefers to base the secondary standard on a three year averaging period for the purpose of program stability, then the level of the standard should be revised downward such that the annual level in any given year of the three year period would not exceed the scientifically recommended range of annual levels of 7 ppm-hrs to 15 ppm-hrs. For example, if in a three year period the highest annual W126 value is 15 ppm-hrs, and the lowest W126 value associated with a three year average is 13 ppm-hrs, then the appropriate level for the three year average would be 13 ppm-hrs to protect against a peak annual level of 15 ppm-hrs. The final Policy Assessment should quantify the ratio of the annual to three year average W126 values to determine what downward adjustment from the annual levels recommended here is needed if a three year form is selected.

As noted in the second draft PA (EPA, 2023, page 1-12), the EPA announced its initiation of the periodic review of the air quality criteria for photochemical oxidants and the O₃ NAAQS in June 2018. The ISA was completed and made available to the public in April 2020. The comments from CASAC and the public were also considered in completing the PA and the advice regarding the standards was described and considered in the final 2020 PA and in the EPA’s decision making. On August 14, 2020, the EPA (Federal Register, 2020a) proposed to retain both the primary and secondary O₃ standards, without revision. In December 2020, the EPA (Federal Register, 2020b) issued its final decision to retain the existing standards without revision. In the 2020 decision, the EPA (1) preferred the use of the W126, as well as peak hourly concentrations of interest, for assessing vegetation risk associated with O₃ exposures and (2) applied the current 8-h form of the O₃ NAAQS to control for those W126 values and peak hourly concentrations associated with vegetation effects.

Fig. 4-6, which is a reproduction of Table 4A-4 in the second draft PA (EPA, 2023), shows a summary from the Lee and Hogsett (1996) analyses of the relative biomass loss for eleven individual tree seedlings and median at various W126 index values identified in the second draft PA (EPA, 2023, page 4A-16). Fig. 4-6 provides estimates of the relative loss for tree biomass at various W126 index values using the composite E-R functions for each species for each integer W126 index value between 7 ppm-hrs and 30 ppm-hrs. The cross species median

of the species-specific composite functions is calculated for all 11 tree species. This table also provides estimates of the number of species for trees that would be below various reference values (e.g., 2% RBL for trees) at various W126 index values.

Table 4A-4. Relative biomass loss for eleven individual tree seedlings and median at various W126 index values.

W126	Douglas Fir	Loblolly	Virginia Pine	Red maple	Sugar maple	Red Alder	Ponderosa Pine	Aspen	Tulip Poplar	Eastern White Pine	Black Cherry	Median (11 species)	Number of Species ≤ 2%	Number of Species ≤ 5%	Number of Species ≤ 10%	Number of Species ≤ 15%
30	0.1%	2.9%	1.7%	3.8%	28.1%	10.4%	12.8%	18.6%	27.7%	25.2%	53.8%	12.8%	2	4	4	6
29	0.0%	2.8%	1.7%	3.6%	23.7%	10.0%	12.3%	17.9%	26.1%	24.0%	52.6%	12.3%	2	4	5	6
28	0.0%	2.7%	1.6%	3.5%	19.9%	9.6%	11.8%	17.2%	24.5%	22.8%	51.4%	11.8%	2	4	5	6
27	0.0%	2.7%	1.6%	3.3%	16.4%	9.2%	11.4%	16.5%	23.0%	21.6%	50.1%	11.4%	2	4	5	6
26	0.0%	2.6%	1.5%	3.1%	13.4%	8.8%	10.9%	15.8%	21.4%	20.5%	48.8%	10.9%	2	4	5	7
25	0.0%	2.5%	1.4%	3.0%	10.9%	8.4%	10.4%	15.2%	19.9%	19.3%	47.5%	10.4%	2	4	5	7
24	0.0%	2.4%	1.4%	2.8%	8.7%	8.0%	10.0%	14.5%	18.4%	18.2%	46.2%	8.7%	2	4	7	8
23	0.0%	2.3%	1.3%	2.7%	6.9%	7.6%	9.5%	13.8%	17.0%	17.1%	44.8%	7.6%	2	4	7	8
22	0.0%	2.2%	1.3%	2.5%	5.3%	7.2%	9.0%	13.1%	15.6%	15.9%	43.3%	7.2%	2	4	7	8
21	0.0%	2.1%	1.2%	2.4%	4.1%	6.8%	8.6%	12.4%	14.3%	14.9%	41.9%	6.8%	2	5	7	10
20	0.0%	2.0%	1.2%	2.2%	3.1%	6.4%	8.1%	11.8%	13.0%	13.8%	40.3%	6.4%	3	5	7	10
19	0.0%	1.9%	1.1%	2.1%	2.3%	6.0%	7.6%	11.1%	11.8%	12.7%	38.8%	6.0%	3	5	7	10
18	0.0%	1.8%	1.0%	1.9%	1.7%	5.7%	7.2%	10.4%	10.6%	11.7%	37.2%	5.7%	5	5	7	10
17	0.0%	1.7%	1.0%	1.8%	1.2%	5.3%	6.7%	9.8%	9.4%	10.7%	35.6%	5.3%	5	5	9	10
16	0.0%	1.6%	0.9%	1.6%	0.9%	4.9%	6.3%	9.1%	8.4%	9.7%	33.9%	4.9%	5	6	10	10
15	0.0%	1.5%	0.9%	1.5%	0.6%	4.5%	5.8%	8.4%	7.4%	8.8%	32.2%	4.5%	5	6	10	10
14	0.0%	1.4%	0.8%	1.4%	0.4%	4.2%	5.4%	7.8%	6.4%	7.9%	30.4%	4.2%	5	6	10	10
13	0.0%	1.3%	0.8%	1.2%	0.3%	3.8%	4.9%	7.1%	5.5%	7.0%	28.6%	3.8%	5	7	10	10
12	0.0%	1.2%	0.7%	1.1%	0.2%	3.5%	4.5%	6.5%	4.7%	6.2%	26.7%	3.5%	5	8	10	10
11	0.0%	1.1%	0.6%	1.0%	0.1%	3.1%	4.1%	5.9%	3.9%	5.4%	24.8%	3.1%	5	8	10	10
10	0.0%	1.0%	0.6%	0.9%	0.1%	2.8%	3.6%	5.2%	3.2%	4.6%	22.9%	2.8%	5	9	10	10
9	0.0%	0.9%	0.5%	0.7%	0.0%	2.4%	3.2%	4.6%	2.6%	3.9%	20.9%	2.4%	5	10	10	10
8	0.0%	0.8%	0.5%	0.6%	0.0%	2.1%	2.8%	4.0%	2.0%	3.2%	18.8%	2.0%	5	10	10	10
7	0.0%	0.7%	0.4%	0.5%	0.0%	1.8%	2.4%	3.4%	1.5%	2.6%	16.7%	1.5%	7	10	10	10

Figure 4-6. Relative biomass loss (RBL) estimates for eleven individual tree seedlings and median at various W126 index values using data from Lee and Hogsett (1996) (EPA, 2023, page 4A-16).

Fig. 4-7, which is a reproduction of Table 4A-10 in the second draft PA (EPA, 2023), shows a summary of the relative biomass loss for sixteen individual tree seedlings and median at various W126 index values identified in the second draft PA (EPA, 2023, page 4A-34). Fig.4-7 provides estimates of the relative loss for tree biomass at various W126 index values using the composite E-R functions from the analysis by Lee et al. (2022) for each species for each integer W126 index value between 7 ppm-hrs and 30 ppm-hrs. The cross species median of the species-specific composite functions is calculated for all 16 tree species. This table also provides estimates of the number of species for trees that would be below various reference values (e.g., 2% RBL for trees) at various W126 index values. In its 2014 letter to the Administrator (Frey, 2014) concerning the draft PA regarding the substitution of the W126 exposure index with the 8-h primary standard, CASAC indicated levels above 15 ppm-hrs that should not be included as options for an alternate secondary standard. A W126 value of 17 ppm-hrs would not adequately protect the most sensitive tree species (highlighted in yellow) identified by Lee et al. (2022).

Table 4A-10. RBL for sixteen tree species (Lee et al., 2022), and the median, for W126 index from 30 to 7 ppm-hrs.

W126	Douglas Fir	Virginia Pine	Red maple	Sugar maple	Red Alder	Ponderosa Pine	Quaking Aspen	Tulip Poplar	Eastern White Pine	Black Cherry	American Sycamore	Winged Sumac	Sweetgum	Chestnut Oak	Table Mountain	Yellow Buckeye	Median (16 species)	# of Species ≤ 2%	# of Species ≤ 5%	# of Species ≤ 10%	# of Species ≤ 15%
30	2.9%	4.6%	7.1%	5.2%	15.4%	17.3%	25.9%	35.3%	4.2%	45.8%	19.7%	53.6%	25.7%	3.6%	2.5%	0.0%	11.2%	1	6	7	7
29	2.9%	4.4%	6.9%	5.1%	14.9%	16.7%	24.8%	34.3%	4.0%	44.7%	19.1%	51.1%	19.2%	3.5%	2.4%	0.0%	10.9%	1	6	7	8
28	2.8%	4.3%	6.7%	4.9%	14.4%	16.2%	23.7%	33.4%	3.9%	43.5%	18.5%	48.6%	14.1%	3.4%	2.3%	0.0%	10.4%	1	7	7	9
27	2.7%	4.1%	6.4%	4.7%	13.9%	15.7%	22.6%	32.4%	3.8%	42.4%	17.9%	46.0%	10.1%	3.3%	2.3%	0.0%	8.3%	1	7	7	9
26	2.6%	4.0%	6.2%	4.5%	13.4%	15.1%	21.5%	31.4%	3.6%	41.2%	17.3%	43.4%	7.1%	3.2%	2.2%	0.0%	6.7%	1	7	8	9
25	2.5%	3.8%	6.0%	4.4%	13.0%	14.6%	20.4%	30.4%	3.5%	40.0%	16.7%	40.8%	4.9%	3.0%	2.1%	0.0%	5.5%	1	8	8	10
24	2.4%	3.7%	5.7%	4.2%	12.5%	14.1%	19.3%	29.4%	3.3%	38.7%	16.1%	38.2%	3.4%	2.9%	2.0%	0.0%	5.0%	1	8	8	10
23	2.3%	3.5%	5.5%	4.0%	12.0%	13.5%	18.2%	28.3%	3.2%	37.5%	15.5%	35.6%	2.2%	2.8%	1.9%	0.0%	4.8%	2	8	8	10
22	2.2%	3.4%	5.3%	3.9%	11.5%	13.0%	17.2%	27.3%	3.1%	36.2%	14.8%	33.0%	1.5%	2.7%	1.8%	0.0%	4.6%	3	8	8	11
21	2.1%	3.2%	5.0%	3.7%	11.0%	12.4%	16.1%	26.2%	2.9%	34.9%	14.2%	30.5%	0.9%	2.6%	1.8%	0.0%	4.4%	3	8	8	11
20	2.0%	3.1%	4.8%	3.5%	10.5%	11.9%	15.0%	25.2%	2.8%	33.5%	13.6%	28.0%	0.6%	2.4%	1.7%	0.0%	4.2%	4	8	8	11
19	1.9%	2.9%	4.6%	3.3%	10.0%	11.3%	14.0%	24.1%	2.7%	32.1%	12.9%	25.5%	0.4%	2.3%	1.6%	0.0%	4.0%	4	8	8	12
18	1.8%	2.8%	4.3%	3.2%	9.5%	10.8%	13.0%	23.0%	2.5%	30.7%	12.3%	23.1%	0.2%	2.2%	1.5%	0.0%	3.8%	4	8	9	12
17	1.7%	2.6%	4.1%	3.0%	9.0%	10.2%	12.0%	21.8%	2.4%	29.3%	11.7%	20.8%	0.1%	2.1%	1.4%	0.0%	3.5%	4	8	9	12
16	1.6%	2.5%	3.9%	2.8%	8.5%	9.6%	11.0%	20.7%	2.2%	27.9%	11.0%	18.6%	0.1%	2.0%	1.3%	0.0%	3.3%	5	8	10	12
15	1.5%	2.3%	3.6%	2.6%	8.0%	9.0%	10.1%	19.5%	2.1%	26.4%	10.4%	16.4%	0.0%	1.8%	1.3%	0.0%	3.1%	5	8	10	12
14	1.4%	2.2%	3.4%	2.5%	7.5%	8.5%	9.1%	18.4%	2.0%	24.9%	9.7%	14.4%	0.0%	1.7%	1.2%	0.0%	2.9%	6	8	12	13
13	1.3%	2.0%	3.2%	2.3%	7.0%	7.9%	8.2%	17.2%	1.8%	23.3%	9.1%	12.4%	0.0%	1.6%	1.1%	0.0%	2.7%	7	8	12	13
12	1.2%	1.8%	2.9%	2.1%	6.4%	7.3%	7.3%	16.0%	1.7%	21.7%	8.4%	10.6%	0.0%	1.5%	1.0%	0.0%	2.5%	7	8	12	13
11	1.1%	1.7%	2.7%	1.9%	5.9%	6.7%	6.4%	14.7%	1.5%	20.1%	7.7%	8.9%	0.0%	1.3%	0.9%	0.0%	2.3%	8	8	13	14
10	1.0%	1.5%	2.4%	1.8%	5.4%	6.1%	5.6%	13.5%	1.4%	18.5%	7.0%	7.4%	0.0%	1.2%	0.8%	0.0%	2.1%	8	8	13	14
9	0.9%	1.4%	2.2%	1.6%	4.9%	5.5%	4.8%	12.2%	1.3%	16.8%	6.4%	6.0%	0.0%	1.1%	0.8%	0.0%	1.9%	8	10	13	14
8	0.8%	1.2%	2.0%	1.4%	4.3%	4.9%	4.0%	10.9%	1.1%	15.1%	5.7%	4.7%	0.0%	1.0%	0.7%	0.0%	1.7%	9	12	13	14
7	0.7%	1.1%	1.7%	1.2%	3.8%	4.3%	3.3%	9.6%	1.0%	13.3%	5.0%	3.6%	0.0%	0.9%	0.6%	0.0%	1.5%	9	13	14	15

Figure 4-7. Relative biomass loss (RBL) estimates for sixteen individual tree seedlings and median at various W126 index values using data from Lee et al. (2022) (EPA, 2023, page 4A-34).

Table 6 from Lee et al. (2022) summarizes the predicted relative biomass loss (PRBL) as a function of the 12-h W126 O₃ metric for the sixteen tree species based on the random coefficients predicted total dry weight plant response. Species are listed in order from most sensitive to least sensitive based on the W126 value associated with 5% biomass loss. Ozone rating based on the test of a significant O₃ effect and W126 value associated with 5% biomass loss: 1) sensitive if W126 < 10 ppm-h; 2) intermediate if 10 < W126 < 28 ppm-h; and 3) insensitive if W126 > 28 ppm-h and/or a linear slope not significantly different from zero.

For the 10 individual tree seedlings (black cherry, Douglas fir, ponderosa pine, quaking aspen, red alder, red maple, sugar maple, tulip poplar, Virginia pine, and white pine) analyzed by both Lee et al. (2022) and Lee and Hogsett (1996), there were differences in the RBL results when reviewing the results from both studies. Lee et al. (2022) expanded the tree experiment dataset analyzed by Lee and Hogsett (1996) and analyzed the updated datasets using a different approach which yielded species-specific E-R functions for RBL. The newly available datasets include new analyses of data from 10 experiments analyzed by Lee and Hogsett (1996) and

experiments for six additional species (American sycamore, winged sumac, sweetgum, chestnut oak, table mountain pine, and yellow buckeye).

The second draft of the PA (EPA, 2023, pages 4-41 – 4-42) notes

Since the 2020 ISA, the tree experiment dataset analyzed by Lee and Hogsett (1996) has been expanded. Additional analyses of both previously and newly available experimental datasets have been performed by a different approach that also yields species-specific E-R functions for RBL (Lee et al., 2022). The newly available datasets include additional experiments for three species among the Lee and Hogsett (1996) eleven, and also experiments for six additional species (Appendix 4A, Table 4A-8). The analyses by Lee et al (2022) derived parameters for functions to estimate RBL from W126 index based on linear or Weibull models parameterized to describe tree seedling biomass (as log total dry weight) for each species at the chamber mean level as a function of 92-day W126 index. The Weibull model is then used to derive the function for RBL (termed “PRBL” [predicted RBL] in Lee et al al., 2022), as further described in Appendix 4A.⁵³

The approaches of the two analyses differ in a number of aspects. As summarized above, the Lee and Hogsett (1996) analysis (1) derived experiment-specific functions describing the responses for each set of O₃ exposures, in terms of the cumulative W126 index over the treatment days (e.g., 55 days or 550 days); (2) predicted RBL at a set of 92-day W126 index values (e.g., 10, 20 ... 60 ppm-hrs) from the experiment-specific datasets; and (3) then derived a species-specific composite function based on the median species response from the experiment-specific function predictions (see Appendix 4A, Attachment 1). In contrast to that approach, Lee et al. (2022) applied a statistical model to exposure and response datasets for each species. Before applying the model, they scaled the W126 index for single-year exposures to 92 days (through application of a factor of 92 divided by number of treatment days), and for multiyear exposures they paired the scaled 92-day W126 index for the last year of exposure with the responses to the multiple years (assessed in the harvest at end of last exposure). The resultant functions are illustrated in Figure 4-5.

As might be expected, the two approaches yield different functions (details provided in Appendix 4A, sections 4A.1.1 and 4A.2.2). The overall pattern across the studied species, however, is not appreciably different, as is illustrated by consideration of RBL estimates for W126 index values below 20 ppm-hrs. For example, for a W126 index of 15 ppm-hrs, RBL estimates from both analyses are above 5% for five species, and above 25% for one of the five. The RBL estimates at 15 ppm-hrs for three of the other four species are between five and 10% via both sets of functions. For the fifth species (tulip poplar), the RBL estimate from Lee and Hogsett (1996) falls between five and 10% while the estimate from Lee et al (2022) is between 15 and 20 %. The difference in the functions and associated estimates for this species appear to relate to differences in the analysis approaches as data from the same set of experiments are analyzed in both cases.

The second draft PA (EPA, 2023, page 4-43) notes an important observation about the relationship between the W126 and N100 occurrences

In addition to statistically analyzing the association of growth reduction with O₃ exposure in terms of W126 index and reporting RBL E-R functions for 16 species, the study by Lee et al. (2022) reported findings on two aspects of the analyses that are informative to issues considered in reviewing the secondary O₃ standard. The first set of findings concerns consideration of tree seedling growth response to controlled conditions in which cumulative O₃ exposure was influenced by different prevalences of high and repeated hourly concentrations. More specifically, analyses described an effect of differences in the hourly concentration pattern, which included differences in the number of hours at or above 100 ppb (N100), on response to O₃ exposure with similar W126 index values. For both species analyzed (quaking aspen and ponderosa pine), the total biomass response was significantly lower for the treatment with similar W126 index values with lower *versus* higher N100 (Lee et al., 2022). Specifically, statistical modeling concluded significant differences in models describing responses for the treatments with higher vs lower N100 for similar magnitude of W126 index. That is, the analyses indicate that high hourly concentrations can exert an impact on growth additional to what might be related to the cumulative exposure quantified by the W126 index.

As described above, the second draft PA (EPA, 2023) was concerned with the observation that Lee et al. (2022) reported that tree seedling growth response was related to cumulative O₃ exposures which appeared to be influenced by different occurrences of high and repeated hourly concentrations. The second draft PA (EPA, 2023) noted that Lee et al. (2022) described an effect where differences in the number of hours at or above 100 ppb (N100) influenced the response of quaking aspen and ponderosa pine although the W126 exposures were similar in each experiment. Figure 3 in Lee et al. (2022) illustrates the O₃ exposure profiles for open-top chamber studies conducted at the U.S. EPA laboratory in Corvallis, Oregon during the 1989–1995 period. The requested hourly O₃ concentrations over a 30-day period are shown for the episodic low, episodic high, daily peak, high-elevation low, and high-elevation high exposure profiles. Lee et al. (2022) noted that the baseline O₃ profile for the studies conducted in Corvallis, Oregon was constructed to reflect the episodic pattern of ambient O₃ concentrations in rural areas for six Midwestern states (i.e., Illinois, Indiana, Ohio, Michigan, Wisconsin, and Minnesota) as per Hogsett et al. (1985). As noted in Lee et al. (2022), for some of the seedling studies performed in Corvallis, Oregon, a second profile was designed as a daily peak profile of equivalent peak concentration with an identical diurnal pattern each day as described in Hogsett et al. (1985).

Figure 6 in Hogsett et al. (1985) illustrates the exposure profile used. The authors described a field-exposure research facility constructed to provide a controlled environment to determine the influence of the various components of O₃ exposure (concentration, frequency, and duration) on plant response. The facility used modified open-top chambers and an automated control system for continuous delivery and control of single pollutants over a growing season.

Two types of exposure profiles were designed (i.e., an episodic regime and a daily peak profile, which resembled a sinusoidal form). The episodic exposure profile was constructed using the EPA's 1978 SAROAD database for a selected Midwestern site in Conneaut, Ohio. The daily sinusoidal peak profiles were created solely for the purpose of producing cumulative exposure values identical to the values experienced in the episodic regimes. **In other words, no attempt was made to have the daily sinusoidal peak profiles mimic the pattern of hourly average O₃ exposures experienced under ambient conditions in the field.** The Hogsett et al. (1985) study described the response of alfalfa and tall fescue growth to episodic and daily peak exposure profiles that experienced equivalent integrated exposure indices over the growing season. Hogsett et al. (1985) reported that over the period of three cuttings (133 days), alfalfa growth was reduced more when exposed to the episodic profile than when exposed to the daily O₃ peak profile. The authors reported that the tall fescue growth was reduced only slightly over a period of three cuttings (90 days) when exposed to either regime.

Thus, the pattern of the daily sinusoidal peak profiles used in some of the experiments performed at the Corvallis facility, as described by Lee et al. (2022), was similar to the pattern applied by Hogsett et al. (1985) for the purpose of producing cumulative exposure values identical to the values experienced in the episodic regimes so as to document whether the higher hourly average O₃ concentrations were more important than the mid- and lower-level values. No attempt was made to design the daily sinusoidal peak profiles to mimic the pattern of hourly average O₃ exposures experienced under ambient conditions in the field. The two exposure protocols, experiencing the same cumulative exposure value, were applied for the purpose of exploring whether differences in growth responses could be observed between the two profiles experiencing the same cumulative exposure.

The second draft PA (EPA, 2023, pages 4-54 – 4-55) notes

There is uncertainty in the shape of species-specific E-R relationships and magnitude of growth impacts on seedlings in air quality conditions that meet the current standard.⁷² While nearly all experiments include a treatment that may relate to this, nearly all experiments additionally include just as many or more treatments, both in terms of W126 index and N100, that differ markedly from air quality conditions common in areas meeting the current standard in that they exhibit higher O₃ exposures, in terms of W126 index and/or N100 (Appendix 4A, Tables 4A-7 and 4A-8). For example, with regard to W126 index, the majority of the experiments have W126 values exceeding 25 ppm-hrs at the highest treatment levels, with some ranging into the hundreds of ppm-hrs. Similarly, as illustrated in Figure 4-7, in the bulk of the treatments for nearly all species for which RBL E-R functions have been derived, N100 is above 10 (Appendix 4A, Tables 4A-7 and 4A-8).⁷³ The extent to which these higher exposures are influencing the shape of the E-R curve and associated predicted RBL at lower exposures is unknown from the available studies. It has been observed, however, for the species evaluated (ponderosa pine and quaking aspen), that an hourly O₃ pattern that included a much higher N100 but the same W126 index yielded a greater growth impact than a pattern that had a lower N100 (see section 4.3.3.1.2 above; Lee et al., 2022).⁷⁴ This suggests that a portion of the growth impacts observed in many of the

experimental treatments for a given W126 index may arise from unusually high hourly concentrations rather than from a comparable W126 index that might occur with lower peaks, thus influencing the shape of the resultant E-R curve.

Footnote 74 in the draft PA (EPA, 2023, page 4-54) notes

⁷⁴ With regard to peak concentrations, even for the experimental treatments with W126 index levels of a magnitude common at U.S. sites that meet the current standard (e.g., less than 20 ppm-hrs), the values for N100 extend up above 10, to more than 40 in one instance (Appendix 4A, Table 4A-7, black cherry and aspen). Across the full set of treatments, including those newly available in Lee et al. (2022), values for N100 extend into the hundreds, up above 500, in a single treatment over 121 days. ***As discussed in section 4.4.1 below, such hourly concentrations are not common for U.S. sites that meet the current standard, at which N100 is virtually always less than 10 (and generally less than 5 [see Figure 4-10 below])*** (emphasis added).

As noted above, the second draft PA highlighted EPA's concern that differences in the number of hours at or above 100 ppb (N100), with similar artificially created W126 index values, could result in differences in vegetation growth response. However, this result would be expected as per the experimental results reported by Hogsett et al. (1985), who demonstrated that the higher hourly average O₃ concentrations were more important than the mid- and lower-level values based on comparing realistic episodic exposure regimes with daily sinusoidal peak profiles (which were not constructed to mimic ambient conditions).

Lee et al. (2022) describe the plant biomass differences between the daily peak high O₃ treatment and the episodic high treatment as follows in Section 3.3 of their paper:

The cumulative O₃ effect on plant biomass was adequately described using the Weibull function and the 12-h W126 metric to characterize the exposure-response relationships for all species and O₃ treatments, but a more complex exposure-response relationship was observed for quaking aspen and ponderosa pine exposed to two or more different O₃ profiles (Table 5). Total plant biomass response to the daily peak high ozone treatment for the 1989 quaking aspen study 11 was significantly less than that for the episodic high treatment having similar W126 value ($p = 0.0007$) (Fig. 8). The total plant biomass response to the daily peak high ozone treatment for the 1989 ponderosa pine study (Study 12D) was less than that for the episodic high treatment having similar W126 value ($p = 0.056$), while a single line was adequate for ponderosa pine exposed for a single year to two or more different O₃ profiles for the other OTC studies (Fig. 7). Plant biomass responses of ponderosa pine exposed to the Midwestern episodic profile offset by 12-h or not offset were adequately fit to a single line for one OTC study (Study 34), indicating the greater role of daytime peak O₃ concentrations than nighttime peak O₃ concentrations in eliciting a plant biomass response.

Lee et al. (2022) describe above the Lee and Hogsett (1999) results for plant biomass responses of ponderosa pine exposed to episodic profiles offset by 12 hours. The nighttime O₃ treatment in the Lee and Hogsett (1999) study was not designed to mimic ambient exposures. Rather, the treatment was applied in the study for the purpose of assessing whether daytime O₃ exposures were more important than nighttime exposures. The plant biomass response of ponderosa pine exposed to the episodic profile applied in one treatment during the daytime hours versus the same episodic profile applied in another treatment during the nighttime hours resulted in greater responses to the daylight O₃ exposures, although both treatments resulted in statistically significant biomass loss differences from the control. Lee and Hogsett (1999) reported for their study that the tissue biomass of ponderosa pine seedlings was significantly reduced when seedlings were exposed to either daytime or nighttime episodic profiles, and the biomass reductions were significantly greater with daytime peak concentrations than with nighttime peak concentrations. Both the daylight and nighttime profiles were designed to experience an equivalent W126 cumulative exposure over the 118-day growing season experimental period. Clearly the nighttime W126 exposure did not mimic the conditions that one might anticipate occurring under ambient conditions.

As noted in the second draft PA (EPA, 2023, page 4-5), in her 2015 decision, the Administrator focused on the relative biomass loss estimates for a revised standard that would generally limit W126 cumulative exposures to those for which the median RBL estimate for seedlings of the 11 species with established E-R functions would be somewhat below 6%. She noted that the median RBL estimate was 6% for a cumulative seasonal W126 exposure index of 19 ppm-hrs. The Administrator noted the CASAC view regarding 6%, most particularly the CASAC's characterization of this level of effect in the median studied species as "unacceptably high".

As noted above, in its 2014 letter to the Administrator (Frey, 2014) concerning the draft PA regarding the substitution of the W126 exposure index with the 8-h primary standard, CASAC indicated within its letter:

Most of the analyses found effects below 15 ppm-hrs (many at 10 or even 7 ppm-hrs)... The CASAC does not support a level higher than 15 ppm-hrs...

...As a policy recommendation, separate from its advice above regarding scientific findings, the CASAC advises that a level of 15 ppm-hrs is requisite to protect crop yield loss, but that lower levels provide additional protection of crop yield loss. Furthermore, there are specific economically significant crops that may not be protected at 15 ppm-hrs but that would be protected at lower levels. Based on scientific judgment of CASAC, a level of 10 ppm-hrs is required to reduce foliar injury. A level of 7 ppm-hours is protective of relative biomass loss for trees. Furthermore, 7 ppm-hours offers additional protection against crop yield loss and foliar injury. Therefore, 7 ppm hours is protective of ecosystem services. Thus, lower levels from the recommended range offer a greater degree of protection of more endpoints than do higher levels from the range.

Fig. 4-8 below from the second draft PA (EPA, 2023, page 4-69) illustrates the relationship between the W126 metric and the design values for the current 8-h O₃ NAAQS over the period 2018-2020. The W126 index is analyzed for both the 3-year design value period (left), as well as the annual period (right).

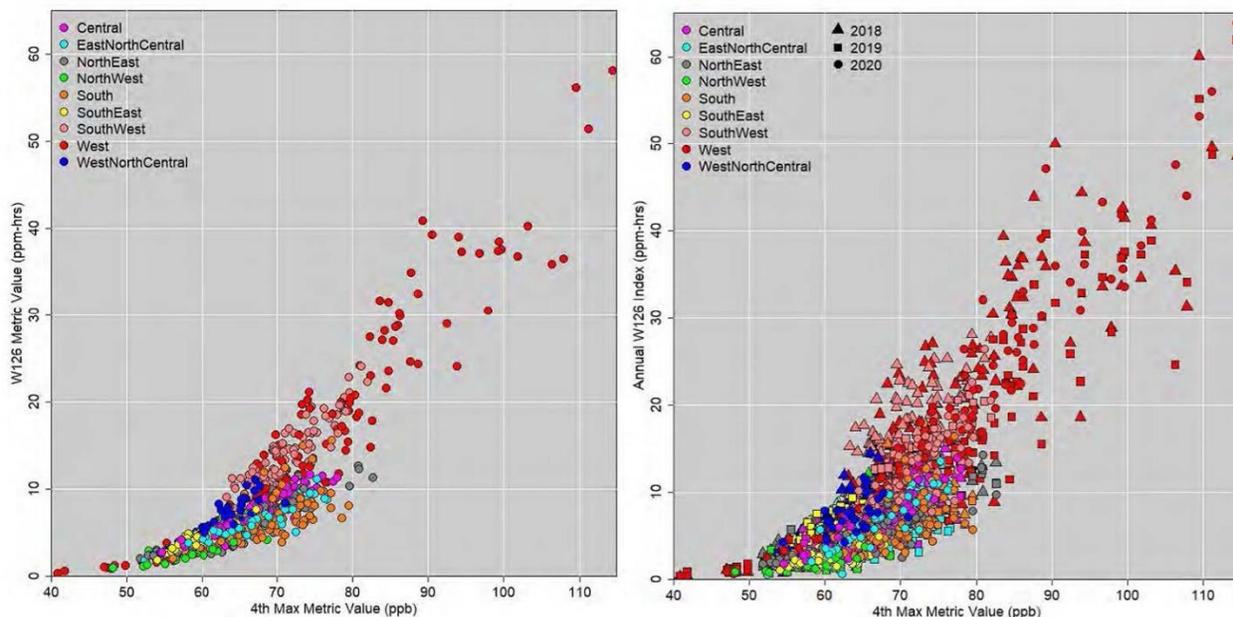


Figure 4-12. Relationship between the W126 index and design values for the current standard (2018-2020). The W126 index is analyzed in terms of averages across the 3-year design value period (left) and annual values (right).

Figure 4-8. Relationship between the W126 index and design values for the current standard (2018-2020). The W126 index is analyzed in terms of averages across the 3-year design value period (left) and annual values (right) (EPA, 2023, page 4-69).

Frey (2014) in his CASAC summary letter noted that lower levels from the recommended W126 range of exposures offered a greater degree of protection of more biological endpoints than the higher levels proposed by the EPA in the draft 2014 PA that CASAC was reviewing. As noted above, CASAC believed that a level of 7 ppm-hrs was protective of relative biomass loss for trees and that 7 ppm-hrs offered additional protection against crop yield loss and foliar injury and therefore, was protective of ecosystem services. As summarized earlier, Lee et al. (2022) noted that the 3-month 12-h W126 estimated to result in a 5% biomass loss was 2.5-9.2 ppm-hrs for sensitive species. At the 7 ppm-hrs annual level, Fig. 4-8 above (Fig. 4-12 in EPA (2023 page 4-69)), the right side of the figure shows that the range of values for the current 8-h form of the O₃ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the O₃ NAAQS of 0.070 ppm (i.e., 70 ppb), numerous occurrences greater than 7 ppm-hrs are evident and that these occurrences are not geographically isolated to the western part of the US (Fig. 4-8 above, right side). Therefore, at a W126 level of 7 ppm-hrs there does not appear to be a strong relationship between the annual W126 and the current form (3-year average) of 8-h O₃ NAAQS. The W126 metric and the 8-h O₃ NAAQS behave differently at levels of 7 ppm-hrs and below. A strategy of using the current O₃ NAAQS to limit annual W126 values to 7 ppm-hrs or below is questionable.

The second draft PA (EPA, 2023, page 4-75) notes that

In considering the prevalence of peak concentrations occurring at monitoring sites, it can be seen that O₃ concentrations at or above 100 ppb occur at lower prevalence at sites that meet the current standard than at sites that meet a range of W126 index values. As shown in Table 4-1, during the highest year for the different N100 or D100 thresholds, the percentage of sites exceeding those thresholds is greater for the sites restricted to meet the different annual W126 levels, with the exception of 7 ppm-hrs, than it is for sites meeting the current standard (design values [3-year 4th Max] at or below 70 ppb) for which the percentages are similar to those for the sites meeting a W126 of 7 ppm-hrs. This observation can also be made for the average percentages across the 3-year period. Further, in looking at the three most recent 3-year periods (extending from 2016 through 2020), a similar finding holds (Table 4-2).

Fig. 4-9 is a reproduction of Table 4-1 in the second draft PA (EPA, 2023, page 4-75) that illustrates the percent of monitoring sites during the 2018 to 2020 period with 4th max or W126 metrics at or below various thresholds that have N100 or D100 values above various thresholds. (EPA, 2023, page 4-75). At a W126 level of 7 ppm-hrs and below, 8% of the sites experience N100 values > 0 occurrences. Nine percent of the sites experience N100 values > 0 at a level of the 3-year 4th Max ≤ 70 ppb. Therefore, the 3-year 4th Max offers no better control for the N100 values than the W126.

In summary, based on the above discussions associated with limiting the W126 and N100 values, the O₃ NAAQS would not necessarily control better than the annual W126 exposure for annual W126 values of 7 ppm-hrs and below. It appears that both the W126 and the 3-year 4th Max can control the N100 values for W126 values of 7 ppm-hrs and below. However, as noted earlier, at an annual W126 level of 7 ppm-hrs, there does not appear to be a strong relationship between the W126 and the current form of the 8-h O₃ NAAQS and that a strategy of using the O₃ NAAQS to limit annual W126 values to 7 ppm-hrs would be questionable. As noted in the previous subsection, in some cases, those O₃ exposure regimes responsible for eliciting adverse vegetation effects may be associated with an extremely low number of occurrences of hourly average O₃ concentrations ≥ 100 ppb or perhaps, in some cases, even zero occurrences ≥ 100 ppb.

Table 4-1. Percent of monitoring sites during the 2018 to 2020 period with 4th max or W126 metrics at or below various thresholds that have N100 or D100 values above various thresholds.

	Total Number of Sites	Number of sites where:			Number of sites where:		
		N100 > 0	N100 > 5	N100 > 10	D100 > 0	D100 > 2	D100 > 5
<i>Average percent of sites exceeding N100 or D100 threshold per year*</i>							
3-year 4 th Max ≤ 70	877	6%	0.4%	<0.1%	6%	0.3%	0%
Annual W126 ≤ 25	1134-1144	11%	1.7%	0.5%	11%	1.7%	0.3%
Annual W126 ≤ 19	1091-1129	10%	1.3%	0.3%	10%	1.3%	0.2%
Annual W126 ≤ 17	1067-1117	9.3%	1.3%	0.2%	9.3%	1.3%	0.2%
Annual W126 ≤ 15	1031-1091	9%	1.2%	0.2%	9%	1.2%	0.1%
Annual W126 ≤ 7	626-860	5.3%	0.4%	0%	5.3%	0.4%	0%
Annual 4 th Max ≤ 70	802-1000	3.7%	0%	0%	3.7%	0%	0%
<i>Percent of sites exceeding N100 or D100 threshold in maximum year of the three</i>							
3-year 4 th Max ≤ 70	See above	9%	0.6%	0.1%	9%	0.5%	0%
Annual W126 ≤ 25		15%	2%	0.6%	15%	2%	0.4%
Annual W126 ≤ 19		13%	2%	0.4%	13%	28%	0.3%
Annual W126 ≤ 17		13%	2%	0.3%	13%	2%	0.3%
Annual W126 ≤ 15		13%	2%	0.3%	13%	2%	0.3%
Annual W126 ≤ 7		8%	1%	0%	8%	1%	0%
Annual 4 th Max ≤ 70		4%	0%	0%	4%	0%	0%
* For the annual metrics, the entries for each N100 or D100 column may be for different years in the 3-year period. Thus the "Total Number of Sites" column presents the range in number of sites that meet the annual 4 th Max or W126 thresholds in each of the three years (as presented in Table 4F-2, Appendix 4F).							

Figure 4-9. Percent of monitoring sites during the 2018 to 2020 period with 4th max or W126 metrics at or below various thresholds that have N100 or D100 values above various thresholds. (EPA, 2023, page 4-75).

Using the 6% RBL criterion noted by the Administrator in 2015, two of the most sensitive species were black cherry and aspen. In the second draft PA (EPA, 2023, pages 4A-21 – 4A-26 and pages 4A-28 – 4A-30), Tables 4A-7 and 4A-8 are provided. Tables 4A-7 and 4A-8 include the N100 values associated with each of the tree seedling experiments. The second draft PA (EPA, 2023, pages 4-38 and 4-39) notes

The established E-R functions for most of the 11 species were derived using data from multiple studies or experiments, many of which employed open top chambers, an established experimental approach, involving a wide range of exposure and/or growing conditions. For example, many of the experimental treatments for exposures to elevated O₃ on which the established E-R functions for the 11 tree seedling species are based involved W126 index levels extending greater than 80 ppm-hrs (for 12-hour 92-day W126 index) and had many (tens to

hundreds) of hours of O₃ concentrations above 100 ppb (Appendix 4A, Tables 4A-7 and 4A-8).^{46 47}

Footnotes 46 and 47 indicate the following:

⁴⁶ Among the experiments on which the E-R functions are based, N100 values for exposure levels most common at U.S. sites that meet the current standard (e.g., W126 index less than 25 ppm-hrs for a single season) range above 10, to more than 40. Across all treatment levels in these experiments, N100 ranged up above 500 (Appendix 4A, Table 4A-7).

⁴⁷ Similarly, the experimental exposures in studies supporting some of the established E-R functions for 10 crop species also include many hours with hourly O₃ concentrations at or above 100 ppb (Lefohn and Foley, 1992).

Tables 4A-7 and 4A-8 in the second draft PA (EPA, 2023, pages 4A-21 – 4A-26 and 4A-28 – 4A-30) show that W126 exposures associated with levels approximating 7 ppm-hrs experience less infrequent occurrences of hourly average O₃ concentrations ≥ 100 ppb. For some of the tree seedling experiments, the N100 values were either 0 or close to 0 when the W126 value approximated 7 ppm-hrs. For the most sensitive tree species, Lee et al. (2022) in their study noted that the N100 values ranged from 0 to 7 for 12-h 92-day W126 exposures of 2.5-9.2 ppm-hrs exposures. ***While many of the tree seedling experiments experienced high numbers of N100 values in the highest treatments, such was not the case for those treatments that experienced the lower W126 values of interest (e.g., a W126 value of 7 ppm-hrs).***

The second draft PA (EPA, 2023) includes the results of Lee et al. (2022). The 16 species included in the Lee et al. (2022) analysis are widespread across the U.S., are ecologically important, and include a variety of deciduous and coniferous, and faster and slower growing trees. The authors applied a reference level of 5% annual biomass loss based on several reasons. One of the reasons, as noted by the authors, was that based on CASAC science advice, US EPA policy assessments in the past have used 5-6% benchmarks for a biomass loss that was unacceptable (EPA 2014a). The 3-month 12-h W126 estimated to result in a 5% biomass loss was 2.5-9.2 ppm-hrs for sensitive species, 20.8-25.2 ppm-hrs for intermediate species, and > 28.7 ppm-hrs for insensitive species. The most sensitive tree species in the Lee et al. (2022) analyses included black cherry, ponderosa pine, quaking aspen, red alder, American sycamore, tulip poplar, and winged sumac. The authors noted that these species are ecologically important and widespread across the U.S. The species-specific exposure-response relationships, according to Lee et al. (2020), would allow U.S. agencies and other groups to better estimate biomass losses based on O₃ exposures in North America and could be used in risk assessment and scenario analyses. The geographic distribution for quaking aspen, black cherry, tulip poplar, and ponderosa pine, which are included in the most sensitive tree species category noted in Lee et al. (2022), is identified in Figs. 4-10 – 4-13 (Tables 4B-1, 4B-10, 4B-5, and 4B-6). When combined, these four tree species cover a large part of the U.S.

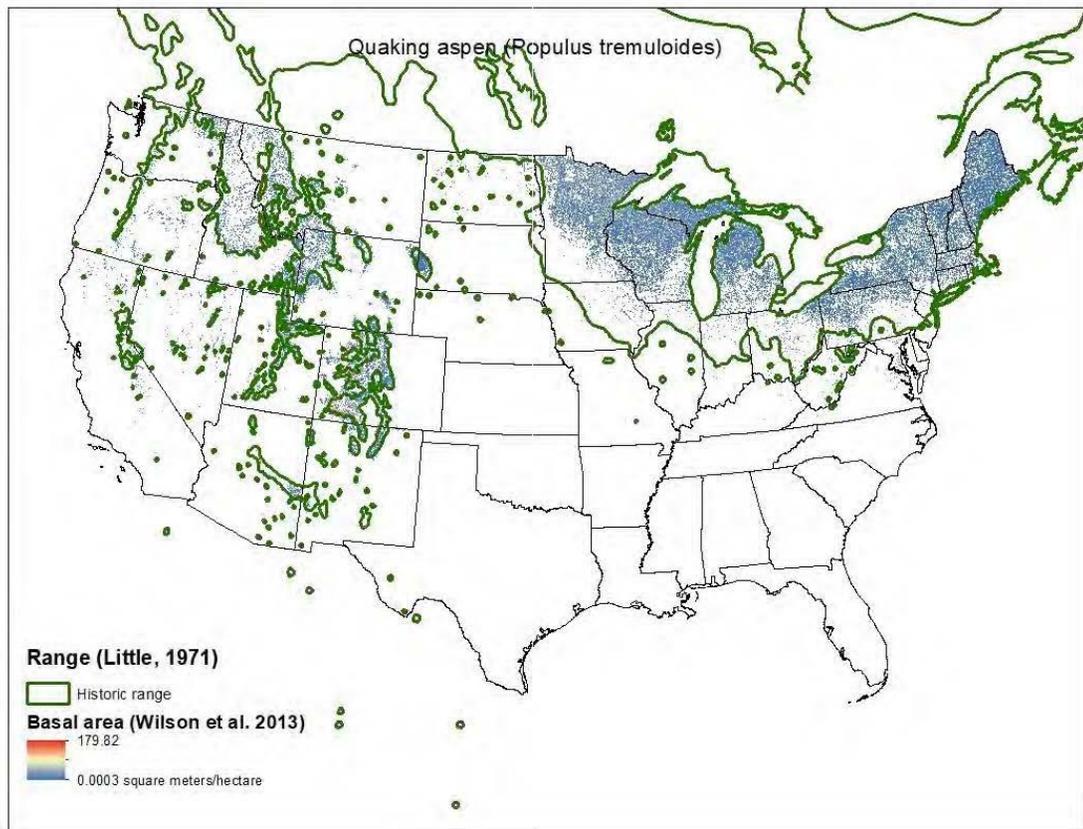


Table 4B-1. Distribution of quaking aspen (*Populus tremuloides*) in the continental U.S.

Figure 4-10. Distribution of quaking aspen in the continental U.S. (EPA, 2023, page 4B-3).

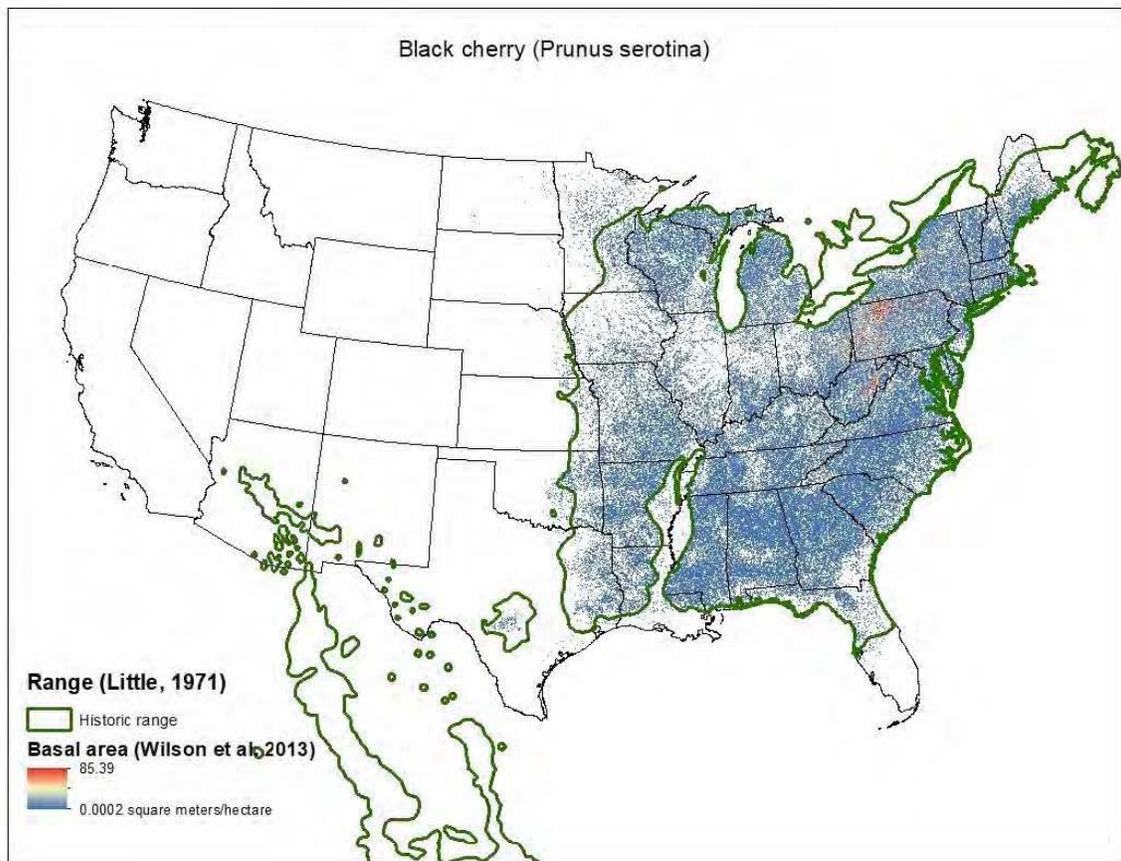


Table 4B-10. Distribution of black cherry (*Prunus serotina*) in the continental U.S.

Figure 4-11. Distribution of black cherry in the continental U.S. (EPA, 2023, page 4B-12).

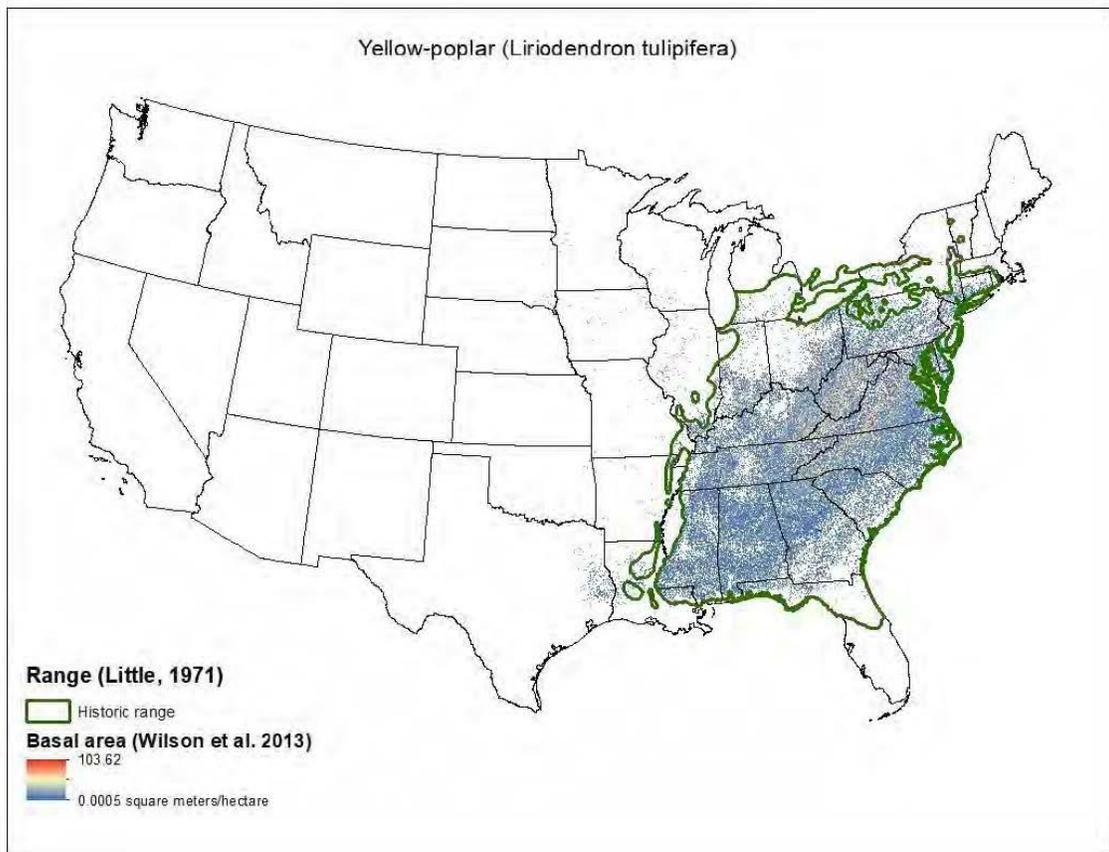


Table 4B-5. Distribution of tulip poplar (*Liriodendron tulipifera*) in the continental U.S.

Figure 4-12. Distribution of tulip poplar in the continental U.S. (EPA, 2023, page 4B-7).

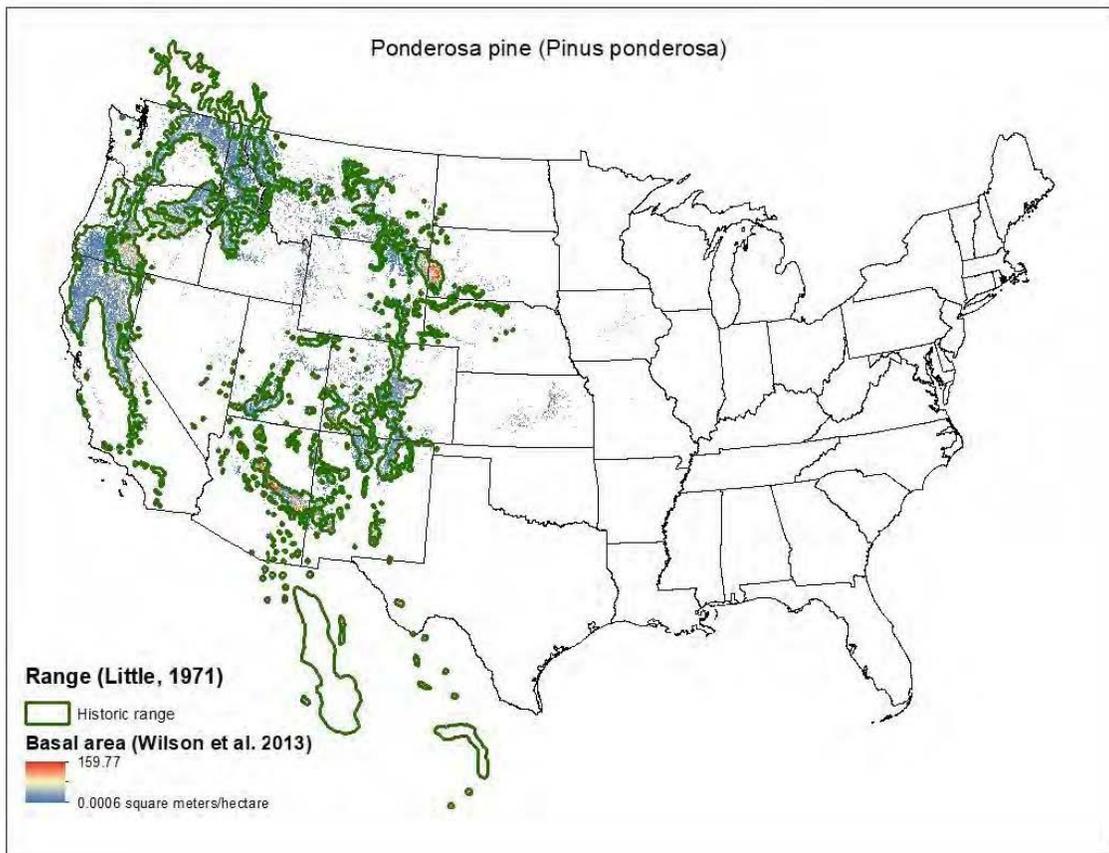


Table 4B-6. Distribution of ponderosa pine (*Pinus ponderosa*) in the continental U.S.

Figure 4-13. Distribution of ponderosa pine in the continental U.S. (EPA, 2023, page 4B-8).

Lee et al. (2022) noted the following about the occurrence of the N100 values in their analyses:

Since the time the exposure studies were performed, air quality policies have been implemented that decreased the number and maximum concentration of episodic peaks in the U.S. (US EPA, 2020b). It has been suggested that one way to measure this is to count the daytime hourly O₃ concentrations of 0.10 ppm and higher (N100), because they may play an important role in eliciting a plant response to O₃ exposure for seedlings as well as for crops (Lefohn and Foley 1992; Lefohn et al., 1997). As seen in Tables 3-5, the N100 count decreases at lower W126 exposure levels. The most sensitive species in our analysis had a biomass loss of 5% at a W126 of 2.5-9.2 ppm-hrs and N100 ranged from 0 to 7 at those exposures. However, there were few experimental exposures between 2.5 and 9.2 ppm-hrs. In that same 2.5-9.2 ppm-hrs range, U.S. O₃ monitoring data from 2016 to 2018 indicate that N100 ranged from 0-10, with an N100 of 0 at most monitors (US EPA, 2020b). While the episodic peaks have been declining

across the U.S. in the past few decades, there are still areas with N100 peaks where the sensitive species occur. Therefore, we believe the exposure response relationships reported in this study remain relevant to current O₃ exposures, especially for the sensitive species.

The frequency of the occurrence of elevated O₃ concentrations (i.e., N100 values) has been documented in the NCLAN and tree seedling experiments. Since the reporting of this phenomenon in the literature, researchers have discussed the use of the W126 exposure index, coupled with the N100 metric, so that the exposure-response models developed in the crop and tree seedling experiments are useful for predicting vegetation effects under current ambient O₃ exposures. The results of Lefohn and Foley (1992), Lefohn et al. (1997), Davis and Orendovici (2006), Kohut (2007), Smith, (2012), and Lee et al. (2022) provide important insight into the application of a unique secondary O₃ NAAQS that consists of the combination of a 3-month 12-h W126 index (e.g., 7 ppm-hrs) coupled with a specific N100 threshold value (e.g., 4 hours). The dual-parameter form of the O₃ NAAQS would be similar to the Air Quality Related Value (AQRV) for vegetation currently employed by the Forest County Potawatomi Community (FCPC) as described in Section 4.2.

4.4 The Use of the 4th Highest Daily Maximum 8-h Concentration as a Substitute for the W126 Exposure Index Coupled with N100 Values as the Secondary O₃ NAAQS

In the 2020 decision (Federal Register, 2020b), the EPA (1) preferred the use of the W126, as well as peak hourly concentrations of interest, for assessing vegetation risk associated with O₃ exposures and then (2) applied the current 8-h form of the O₃ NAAQS to control for those W126 values and peak hourly concentrations associated with vegetation effects. As noted previously, Lefohn et al. (1997) identified, for the most sensitive tree seedlings in the Southern Appalachian area, a 24-h 92-day W126 value of 5.9 ppm-hrs coupled with an N100 value of 6. The Forest County Potawatomi Community established the O₃ thresholds for the 3-year average of the 3-month (June, July, and August) 24-h cumulative W126 value at 7.0 ppm-hr and the 3-year average of the 3-month (June, July, and August) number of hours \geq 100 ppb (N100) at 4 for vegetation (<https://lnr.fcpotawatomi.com/wp-content/uploads/2015/01/FCPC-AQRV-Threshold-Effects-Levels-Webpage-version1.pdf>). Lee et al. (2022) reported for the most sensitive species in their analysis for a biomass loss of 5% a W126 value of 2.5-9.2 ppm-hrs and N100 values ranging from 0 to 7 at those exposures.

At the 7 ppm-hrs annual level, Fig. 4-8 (Fig. 4-12 in EPA (2023, page 4-69)), the right side of the figure shows that the range of values for the current form of the O₃ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the O₃ NAAQS of 0.070 ppm (i.e., 70 ppb), numerous occurrences greater than 7 ppm-hrs are evident and that these occurrences are not geographically isolated to the western part of the US. The W126 metric and the 8-h O₃ NAAQS each behave differently at levels of 7 ppm-hrs and below. Focusing on the N100 values, the second draft PA (EPA, 2023) notes for the 2018 to 2020 period that at a W126 level of 7 ppm-hrs and below, 8% of the sites experience N100 values $>$ 0 occurrences. Nine percent of the sites experience N100 values $>$ 0 occurrences at a level of the 3-year 4th Max \leq 70 ppb. Therefore, the 3-year 4th Max offers no better control for the N100 values than the W126.

As noted earlier, Lee et al. (2022) identified, for the most sensitive species in their analysis (i.e., black cherry, ponderosa pine, quaking aspen, red alder, American sycamore, tulip poplar, and winged sumac) at a biomass loss of 5%, a W126 value of 2.5-9.2 ppm-hrs and N100 values ranging from 0 to 7 at those exposures. The authors also noted that these sensitive species were ecologically important and widespread across the U.S. The selection of a 3-month, 12-h W126 at least at the level of 7 ppm-hrs with an N100 value > 0 would appear to be protective of some of the most sensitive species analyzed in the Lee et al. (2022) study. The actual determination of the specific W126 level, as well as the N100 value, would be the decision made by the EPA Administrator.

Therefore, at a W126 level of 7 ppm-hrs, there does not appear to be (1) a strong relationship between the annual W126 and the current form of the 8-h O₃ NAAQS and (2) an advantage of using the current 8-h form of the 3-year 4th Max ≤ 70 ppb to control for the N100 values. Thus, the case for preferring the current form of the O₃ NAAQS to limit annual W126 values to 7 ppm-hrs (or perhaps below this level), as well as for using the current O₃ NAAQS to reduce N100 values to zero, is not strong.

4.5 1-Year W126 versus the 3-Year Average of the W126 Exposure Metric

In its August 2019 decision, the D.C. Circuit addressed arguments regarding the adequacy of the EPA Administrator's 2015 decision to use a 3-year average of the W126 index as the benchmark instead of the 1-year average of the W126 metric. The Court noted:

The Environmental Petitioners raise two challenges pertaining to the W126 index, both as a benchmark and as a form and averaging time. First, they argue that EPA impermissibly departed from CASAC's advice by setting the secondary standard level using a three-year average W126 benchmark without lowering the level to protect against single-year exposures associated with median annual tree growth loss of 6%, which CASAC had advised was "unacceptably high." J.A. 518. They also contend that EPA arbitrarily disregarded CASAC's advice to adopt the W126 index as the form and averaging time for the secondary standard. We remand to EPA on the first issue and do not reach the second.

The Three-Year Average Benchmark. CASAC advised EPA that basing the secondary standard on a "single-year period" would provide "more protection for annual crops and for the anticipated cumulative effects on perennial species" than a three-year average. J.A. 518. It explained that EPA's proposal to use a three-year averaging period was "not supported by the available data," J.A. 536, and that if EPA chose to "base the secondary standard on a three-year averaging period," then "the level of the standard should be revised downward such that the level for the highest three-month summation in any given year of the three-year period would not exceed [its] scientifically recommended" range of single-year, W126 exposure levels, J.A. 518. This was necessary, CASAC explained, to

“protect against single unusually damaging years that will be obscured in the average.” J.A. 536.

EPA argues it gave effect to CASAC’s recommendation by using a three-year average benchmark of 17 ppm-hrs, which, after adjusting for the cottonwood data, was “somewhat below” the 19 ppm-hrs associated with 6% annual growth loss that CASAC had advised was “unacceptably high.” 80 Fed. Reg. at 65,406-07 (quoting J.A. 518). But CASAC had advised a maximum level associated with 5.2% annual biomass loss, see J.A. 631, and it expressly cautioned that 6% median growth loss in a single year was unacceptable, see J.A. 518. EPA’s use of a benchmark that averages out to less than 6% biomass loss over three years does not accord with CASAC’s advice. Indeed, as commenters informed EPA during the rulemaking, see J.A. 1836-40, EPA’s own air quality data suggests that many large national parks and wilderness areas that have met EPA’s chosen three-year average 17 ppm-hrs benchmark— areas that Congress considers significant to the public welfare, see 42 U.S.C. §§ 7470(2), 7472(a)—have meanwhile recorded single-year W126 values at and above 19 ppm-hrs, which is associated with “unacceptably high” annual biomass loss of 6% and higher. See J.A. 1061-64; 80 Fed. Reg. at 65,391. EPA critiques that data as marred by outdated handling procedures, but the agency acknowledged that other data derived through updated procedures produced results “similar to” those showing harmful exposure spikes. J.A. 1213. Critically, EPA points us to no data or analysis (based on new or old procedures) suggesting that the chosen benchmark prevents single seasonal exposures of 19 ppm-hrs or higher. In short, EPA has not demonstrated how its chosen benchmark protects against “unusually damaging years that will be obscured in the average.” J.A. 536.

EPA alternatively defends its decision to use the three-year-average benchmark as providing a focus on public welfare effects of “potentially greater” significance than effects “associated with a single year” of exposure. 80 Fed. Reg. at 65,404. This position, however, is inconsistent with EPA’s other actions. In establishing the secondary standard, for example, EPA heavily relied on data showing annual 6% median tree growth loss at 19 ppm-hrs, see *id.* at 65,406, and acknowledged the potential for a single season of high ozone exposure to “alter biomass allocation and plant reproduction in seasons subsequent to [that season’s] exposure,” thereby leading to “a negative impact on species regeneration in subsequent years,” *id.* at 65,371-72; see also J.A. 740-41. EPA additionally recognized that “ozone effects in plants are cumulative,” *id.* at 65,373 (quoting EPA, *Integrated Science Assessment 2-44* (2013)), meaning that the adverse vegetative effects from single, high-ozone years are not offset by subsequent low-ozone years. EPA has identified no contrary evidence in the record demonstrating why these single-year effects matter less than a three-year average.

We therefore remand this issue for EPA to either lower the standard to protect against unusually damaging cumulative seasonal exposures that will be obscured in its three-year average, or explain its conclusion that the unadjusted average is

an appropriate benchmark notwithstanding CASAC's contrary advice. Alternatively, EPA could adopt the single-year W126 exposure index as the form and averaging time, which would presumably moot any problems with the way it translated that index to use as a benchmark.

The second draft PA (EPA, 2023, page 4-8) notes in footnote 9 the following:

⁹ In its discussion regarding the EPA's use of a 3-year average W126 index, the 2019 court decision remanding the 2015 standard back to the EPA referenced advice from the CASAC in the 2015 review on protection against "unusually damaging years." Use of this term occurs in the 2014 CASAC letter on the second draft PA (Frey, 2014). Most prominently, the CASAC defined as damage "injury effects that reach sufficient magnitude as to reduce or impair the intended use or value of the plant to the public, and thus are adverse to public welfare" (Frey, 2014, p. 9). We also note that the context for the CASAC's use of the phrase "unusually damaging years" in the 2015 review is in considering the form and averaging time for a revised secondary standard in terms of a W126 index (Frey, 2014, p. 13), which as discussed below is relatively less controlling of high-concentration years (whether as a single year index or averaged over three years) than the current secondary standard and its fourth highest daily maximum 8-hour metric (85 FR 87327, December 31, 2020).

As noted in footnote 9 above, CASAC's use of the phrase "unusually damaging years" (Frey, 2014) focused on the W126 index in the 2015 review when considering the form and averaging time for a revised secondary standard. In the Administrator's 2020 decision (Federal Register, 2020b), the EPA introduced the concept that the W126 index was relatively less controlling of high-concentrations years (whether as a single year index or averaged over three years) than the current 8-h secondary standard. The EPA in its 2020 decision introduced air quality analyses that investigated the annual occurrence of elevated hourly O₃ concentrations of concern. The results of the air quality analyses first appeared in the Administrator's final decision. The second draft PA (EPA, 2023), page 4-9) notes that

With regard to the EPA's use of a 3-year average W126 index to assess protection from RBL, the 2020 decision additionally took into account the 2019 court remand on this issue, including the remand's reference to protection against "unusually damaging years." (85 FR 87325-87328, December 31, 2020). Accordingly, the EPA considered air quality analyses of peak hourly concentrations in the context of considering protection against "unusually damaging years." With regard to this caution, and in the context of controlling exposure circumstances of concern (e.g., for growth effects, among others), the EPA considered air quality analyses that investigated the annual occurrence of elevated hourly O₃ concentrations which may contribute to vegetation exposures of concern (2020 PA, Appendix 2A, section 2A.2; Wells, 2020). These air quality analyses illustrate limitations of the W126 index (whether in terms of a 3-year average or a single year) for the purpose of controlling peak concentrations,¹¹ and also the strengths of the current standard in this regard. The air quality analyses

show that the form and averaging time of the existing standard, in addition to controlling cumulative exposures in terms of W126 (as found in the 2015 review), is much more effective than the W126 index in limiting peak concentrations (e.g., hourly O₃ concentrations at or above 100 ppb)¹² and in limiting number of days with any such hours (Wells, 2020, e.g., Figures 4, 5, 8, 9 compared to Figures 6, 7, 10 and 11).¹³ Thus, the W126 index, by its very definition, and as illustrated by the air quality data analyses, does not provide specificity with regard to year-to-year variability in elevated hourly O₃ concentrations with the potential to contribute to the “unusually damaging years” that the CASAC had identified for increased concern in the 2015 review. As a result, the 2020 decision found that a standard based on a W126 index (either a 3-year or a single-year index) would not be expected to provide effective control of the peak concentrations that may contribute to “unusually damaging years” for vegetation.¹⁴ Based on all of the above, the 2020 decision concluded that control of such years is a characteristic of the existing standard (the effectiveness of which is demonstrated by the air quality analyses), and that that use of a seasonal W126 averaged over a 3-year period, which is the design value period for the current standard, to estimate median RBL using the established E-R functions, in combination with a broader consideration of air quality patterns, such as peak hourly concentrations, is appropriate for considering the public welfare protection provided by the standard (85 FR 87340-87341, December 31, 2020).

Footnote 14 above in the second draft of the PA (EPA, 2023, page 4-10) states

¹⁴ From these analyses, the Administrator concluded that the form and averaging time of the current standard is effective in controlling peak hourly concentrations and that a W126 index based standard would be much less effective in providing the needed protection against years with such elevated and potentially damaging hourly concentrations.

As noted at the end of Section 4.4 of these comments, at a W126 level of 7 ppm-hrs, there does not appear to be (1) a strong relationship between the annual W126 and the current 3-year form of the 8-h O₃ NAAQS and (2) an advantage of using the current 8-h form of the 3-year 4th Max ≤ 70 ppb to control for the N100 values. Thus, the case for preferring the current form of the O₃ NAAQS to limit annual W126 values to 7 ppm-hrs (or perhaps below this level), as well as for using the current O₃ NAAQS to reduce N100 values to zero does not appear to be strong. The Agency’s 2020 decision that the current form of the 3-year average of the 4th highest daily maximum 8-h standard would control for a seasonal W126 in combination with a broader consideration of air quality patterns, such as peak hourly concentrations, appears to be based on annual W126 values greater than 7 ppm-hrs.

In response to the Court’s decision, the Administrator noted in his O₃ NAAQS rulemaking (Federal Register, 2020b) that his response addressed the August 2019 decision by the D.C. Circuit on the secondary standard established in 2015 and issues raised. The EPA expanded certain analyses in its 2020 review compared with those conducted in the previous

review, including discussion on issues raised in the remand, and provided additional explanation of rationales for proposed conclusions on these points in his 2020 review.

As noted by the Administrator (Federal Register, 2020b, page 87326):

The EPA concludes that the 3-year average W126 index is a reasonable metric for assessing the level of protection provided by the current standard from cumulative seasonal exposures related to RBL, while noting that our evaluation for the protection provided by the current standard has also been informed by our consideration of other metrics (as described further below). In reaching this conclusion, we have taken into account the available evidence base and air quality analyses, with a focus on two types of considerations, as well as consideration of the context for RBL as a proxy for an array of other vegetation effects (discussed in section III.B.2.b(v) below). The first of the two consideration types concerns the E-R functions and their use with a 3-year average W126 index, and the second concerns the control by the W126 index metric of exposures that might be termed “unusually damaging.” With regard to the first, we find our use of the 3-year average W126 index appropriate in light of the approach used in deriving the E-R functions from the underlying data (from exposures of varying durations, including of multiple years), and the evidence available for evaluating these functions across multiyear exposures.²¹⁰ Additionally, with regard to the second consideration, we recognize limitations associated with a reliance solely on W126 index as a metric to control exposures that might be termed “unusually damaging.” For example, two different air quality patterns for which the associated W126 index is the same may have very different incidence of elevated O₃ concentrations, and accordingly pose different risks to vegetation. As discussed below, however, the occurrence of such concentrations (and any associated risk of damage) are controlled by the current secondary standard. In light of this evidence, and recognizing the role for both peak and cumulative exposures in eliciting growth and related vegetation and ecosystem effects, the EPA concludes that focusing solely on W126 index (either in terms of a single year or 3-year average) in considering the public welfare protection provided by the current standard would not be considering all the relevant scientific information. *To the extent that the prior CASAC advised that the EPA should focus solely on single-year W126 index values in evaluating the protection provided by the secondary standard, the EPA disagrees that this would provide the needed protection, for the reasons explained more fully below* (emphasis added). In this regard, we additionally note that the current CASAC concluded that focusing on three-year average W126 index values in considering the public welfare protection offered by the secondary standard “appears of reasonable thought and scientifically sound” (Cox, 2020a, p. 19).

Footnote 210 states:

²¹⁰ Additionally, as described in section III.B.1.c above and III.B.2.b(v) below, the EPA’s identification of 17 ppm-hrs for a target W126 index of 17 ppm-hrs

(e.g., versus 18 ppm-hrs) was in consideration of the prior CASAC recommendation for considering a “lower” level ppm-hrs.

It is important to note that CASAC’s use of the phrase “unusually damaging years” (Frey, 2014) focused on the W126 index in the 2015 review when considering the form and averaging time for a revised secondary standard. CASAC was specifically concerned about the use of the 3-year average W126 versus the annual W126 index on the protection of vegetation from “unusually damaging years.” In the paragraph above, EPA has expanded the term “unusually damaging years” to include concern for the control of high-concentration years. As noted above, the EPA concluded in its final rulemaking of the O₃ NAAQS in 2020 that focusing solely on W126 index (either in terms of a single year or 3-year average) in considering the public welfare protection provided by the current standard would not be considering all the relevant scientific information. Thus, the EPA has expanded the focus of the prevention of “unusually damaging years,” as originally defined by CASAC (Frey, 2014), from the W126 index (either in terms of a single year or 3-year average) to now include one additional consideration: the control of high-concentration years (i.e., the number of hourly average O₃ concentrations \geq 100 ppb). For this additional aspect, the second draft PA (EPA, 2023, page 4-9) notes that EPA considers air quality analyses of peak hourly concentrations in the context of considering protection against “unusually damaging years.”

The Administrator continues (Federal Register, 2020b, pages 87326-87327)

With regard to the comment that cited a recommendation from the prior CASAC on protection of vegetation against “unusually damaging years” and the part of the court remand referencing that CASAC recommendation, we have considered the CASAC discussion using this term, in the context of the court remand. Use of this term by the prior CASAC occurs in the 2014 letter on the second draft PA in the 2015 review (Frey, 2014b). Most prominently, the prior CASAC defined as damage “injury effects that reach sufficient magnitude as to reduce or impair the intended use or value of the plant to the public, and thus are adverse to public welfare” (Frey, 2014b, p. 9). The prior CASAC additionally provided advice with regard to surrogate metrics for judging such “damage,” e.g., use of RBL for judging effects on trees and their related functions and ecosystem services, use of crop RYL for judging public welfare effects of crop effects (Frey, 2014b, p. 10). We also note that the context for the prior CASAC’s use of the phrase “unusually damaging years” is in considering the form and averaging time for a revised secondary standard in terms of a W126 index (Frey, 2014b, p. 13), which as discussed below is relatively less controlling of high-concentration years, rather than in the context of the current secondary standard and its fourth highest daily maximum 8-hour metric.

While the prior CASAC did not provide any specificity or details as to the exposure circumstances and damage intended by its more general phrasing, nor did it cite to specific evidence in scientific publications, *we agree with the general concept that particular air quality patterns in a year may pose particular risk of vegetation damage, in terms of both or either growth-related effects or visible*

foliar injury (discussed in section III.B.2(iii) below) (emphasis added). Across past O₃ NAAQS reviews, the air quality criteria for vegetation effects have emphasized the risk posed to vegetation from higher hourly average O₃ concentrations (e.g., “[h]igher concentrations appear to be more important than lower concentrations in eliciting a response” [ISA, p. 8-180]; “higher hourly concentrations have greater effects on vegetation than lower concentrations” [2013 ISA, p. 91-4] “studies published since the 2006 O₃ AQCD do not change earlier conclusions, including the importance of peak concentrations, ... in altering plant growth and yield” [2013 ISA, p. 9-117]). In fact, the EPA has recognized the W126 index for E-R models for growth and yield (in the current and prior ISA and prior AQCD) in part due to its preferential weighting of higher concentrations (ISA, p. 8-130).

We note, however, that while the W126 index weights higher hourly concentrations, it cannot, given its definition as an index that sums three months of weighted hourly concentrations into a single value, always differentiate between air quality patterns with high peak concentrations and those without such concentrations. This is illustrated by the following two hypothetical examples. In the first example, two air quality monitors have a similar pattern of generally lower average hourly concentrations, but differ in the occurrence of higher concentrations (e.g., hourly concentrations at or above 100 ppb). The W126 index describing these two monitors would differ. In the second example, one monitor has appreciably more hourly concentrations above 100 ppb compared to a second monitor; but the second monitor has higher average hourly concentrations than the first. In the second example, the two monitors may have the same W126 index, even though the air quality patterns observed at those monitors are quite different, particularly with regard to the higher concentrations, which have been recognized to be important in eliciting responses (as noted above).

Thus, the EPA disagrees with a view implied by many of the commenters (who object to the EPA’s proposed decision) that the sole focus for assessing public welfare protection, related to vegetation damage, and air quality control provided by the secondary standard should be on the W126 index. This view ignores both the limitations of the W126 index itself in distinguishing among different patterns of hourly O₃ concentrations and the fact that the current secondary standard has, by virtue of its form, a metric that does. With regard to these limitations of the W126 index, as described above, two different locations or years may have different patterns of hourly concentrations but the same W126 index value. This was recognized in the study by Lefohn et al (1997), which observed the appreciable differences between the prevalence of hourly concentrations at or above 100 ppb in exposures on which the E-R functions are based and those common in ambient air.²¹⁴

Footnote 214 states:

²¹⁴ For example, many of the experimental exposures of elevated O₃ on which the established E-R functions for the 11 tree seedling species are based, had hundreds of hours of O₃ concentrations above 100 ppb, far more than are common in (unadjusted) ambient air, including in areas that meet the current standard (Lefohn et al 1997; PA, Appendix 2A, section 2A.2; Wells, 2020). Similarly, the experimental exposures in studies supporting some of the established E-R functions for 10 crop species also include many hours with hourly O₃ concentrations at or above 100 ppb (Lefohn and Foley, 1992).

The Administrator continues (Federal Register, 2020b, pages 87328)

...Thus, based on the findings of both the analyses in the PA (PA, Appendix 2A) and the additional analyses (Wells, 2020), the EPA disagrees with the commenter that the proposed decision ignores the importance of elevated hourly O₃ concentrations in eliciting effects on vegetation. Rather, the proposed decision, and final decision to retain the existing standard, which controls peak concentrations and also cumulative seasonal exposure in terms of W126 index, explicitly considers this importance and address it in a way that is more effective than a standard expressed in terms of the W126 index would be, even based on a single-year W126 well below 17 ppm-hrs (as shown in the additional air quality analyses [Wells, 2020]).

In summary, we find that a 3-year average is appropriate for use in assessing protection for RBL based on the established tree seedling E-R functions, in light of the discussion above, while also finding it important to consider additional aspects of O₃ air quality, that influence vegetation exposures of potential concern, in reaching conclusions about the adequacy of the current standard. We disagree with the commenters and the prior CASAC that focus on a single year W126 index is needed to protect against years with O₃ concentrations with the potential to be “unusually damaging,” Rather, as described here, the metric of the current standard provides strong protection against elevated hourly concentrations that might contribute to “unusually damaging” years with the potential to be adverse to the public welfare, as well as providing protection against effects of cumulative exposures seen in experimental studies. Accordingly, we disagree with those commenters that express the view that the current standard does not provide such protection.

As noted in footnote 214 (Federal Register, 2020b, page 87327), Lefohn et al. (1997) and Lefohn and Foley (1992) identified numerous occurrences of hourly average O₃ concentrations \geq 100 ppb (N100) in some of the treatments in the tree seedling experiments, as well as the crop experiments. It is important to note that not all the treatments in the tree seedling and crop experiments experienced elevated O₃ concentrations. At the more moderate treatments, the exposures contained infrequent occurrences of hourly average O₃ concentrations \geq 100 ppb (see for example, Tables 4A-7 and 4A-8 in the second draft PA (EPA, 2023, pages 4A-21 – 4A-26 and 4A-28 – 4A-30). ***The EPA appears to be more focused on the highest treatments in the tree seedling and crop experiments than the more moderate treatments, where the more sensitive***

species experienced biomass and yield losses under exposures experiencing infrequent occurrences of hourly average O₃ concentrations ≥ 100 ppb. In its 2020 decision, the Administrator preferred to use the W126, as well as peak hourly concentrations of interest, for assessing vegetation risk associated with O₃ exposures. The Administrator also noted that he preferred to apply the current 8-h form of the O₃ NAAQS to control for those W126 and hourly average O₃ concentrations ≥ 100 ppb values of concern for the protection of vegetation from O₃ exposures.

The second draft PA (EPA, 2023, page 4-5) notes that the Administrator in the 2015 decision, using RBL estimates as a proxy, focused her attention on a revised standard that would generally limit cumulative exposures to those for which the median RBL estimate for seedlings of the 11 tree species with established E-R functions would be somewhat below 6%. However, the use of a median RBL estimate implies that the resultant median RBL will reflect sensitive, intermediate, and insensitive species. Rather than using a median RBL that reflects all sensitivities, it is more relevant to identify the RBL estimates at specific biomass loss percentage levels for those sensitive species that are ecologically important and widespread throughout the U.S. Associated with these RBL estimates will be ranges of W126 values accumulated over an approximate 90 days, as well as the number of hourly average O₃ concentrations ≥ 100 ppb occurring over the period of interest. In section 4.4 of these comments, it was noted that Lefohn et al. (1997) had identified for the most sensitive tree seedlings in the Southern Appalachian area a 24-h 92-day W126 value of 5.9 ppm-hrs coupled with an N100 value of 6. The Forest County Potawatomi Community established the O₃ thresholds for the 3-year average of the 3-month (June, July, and August) 24-h cumulative W126 value at 7.0 ppm-hr and the 3-year average of the 3-month (June, July, and August) number of hours ≥ 100 ppb (N100) at 4 for vegetation (<https://lnr.fcpotawatomi.com/wp-content/uploads/2015/01/FCPC-AQRV-Threshold-Effects-Levels-Webpage-version1.pdf>). Lee et al. (2022) identified, for the most sensitive species in their analysis (i.e., black cherry, ponderosa pine, quaking aspen, red alder, American sycamore, tulip poplar, and winged sumac) at a biomass loss of 5%, a W126 value of 2.5-9.2 ppm-hrs and N100 values ranging from 0 to 7 at those exposures. The authors also noted that these sensitive species were ecologically important and widespread across the U.S.

CASAC's use of the phrase "unusually damaging years" (Frey, 2014) focused on the W126 index in the 2015 review when considering the form and averaging time for a revised secondary standard. CASAC was specifically concerned about the use of the 3-year average W126 versus the annual W126 index. The second draft PA (EPA, 2023, page 4D-10) comments on the variation of the annual W126 index from the 3-year (2018-2020) W126 index. The second draft PA (EPA, 2023, page 4-10) notes

Figure 4D-6 shows a scatter plot of the deviations in the 2018, 2019, and 2020 annual W126 index values (y-axis) from the 2018-2020 average W126 metric values (x-axis). This figure shows that the magnitude of the annual W126 index deviations from the 3-year average tend to increase as the W126 metric value increases. About 40% of the annual W126 index values are within +/- 1 ppm-hr of the 3-year average value, about 73% are within +/- 2 ppm-hrs of the 3-year average value, and about 96% are within +/- 5 ppm-hrs of the 3-year average value. Figure 4D-7 also presents the deviations in the 2018, 2019, and 2020

annual W126 index values from their respective 2018-2020 averages for the sites meeting the current standard. For these sites, 42% of annual W126 index values are within 1 ppm-hr of the 3-yr average, 78% are within 2 ppm-hrs, and 99% are within 5 ppm-hrs (Figure 4D-7). From these two figures it can be seen that lower 4th max metric values generally correspond to smaller inter-annual variation within W126 metric values, especially for sites meeting the current standard.

Figs. 4-14 (Fig. 4D-6 in the second draft PA) and 4-15 (Fig. 4D-7 in the second draft PA) illustrate the variation of the annual W126 values from the 2018-2020 average W126 metric values, as well as from the current 8-h NAAQS. Both figures illustrate that for the annual W126 index, the deviation can be large relative to the absolute value of the W126 index at the 7 ppm-hrs level.

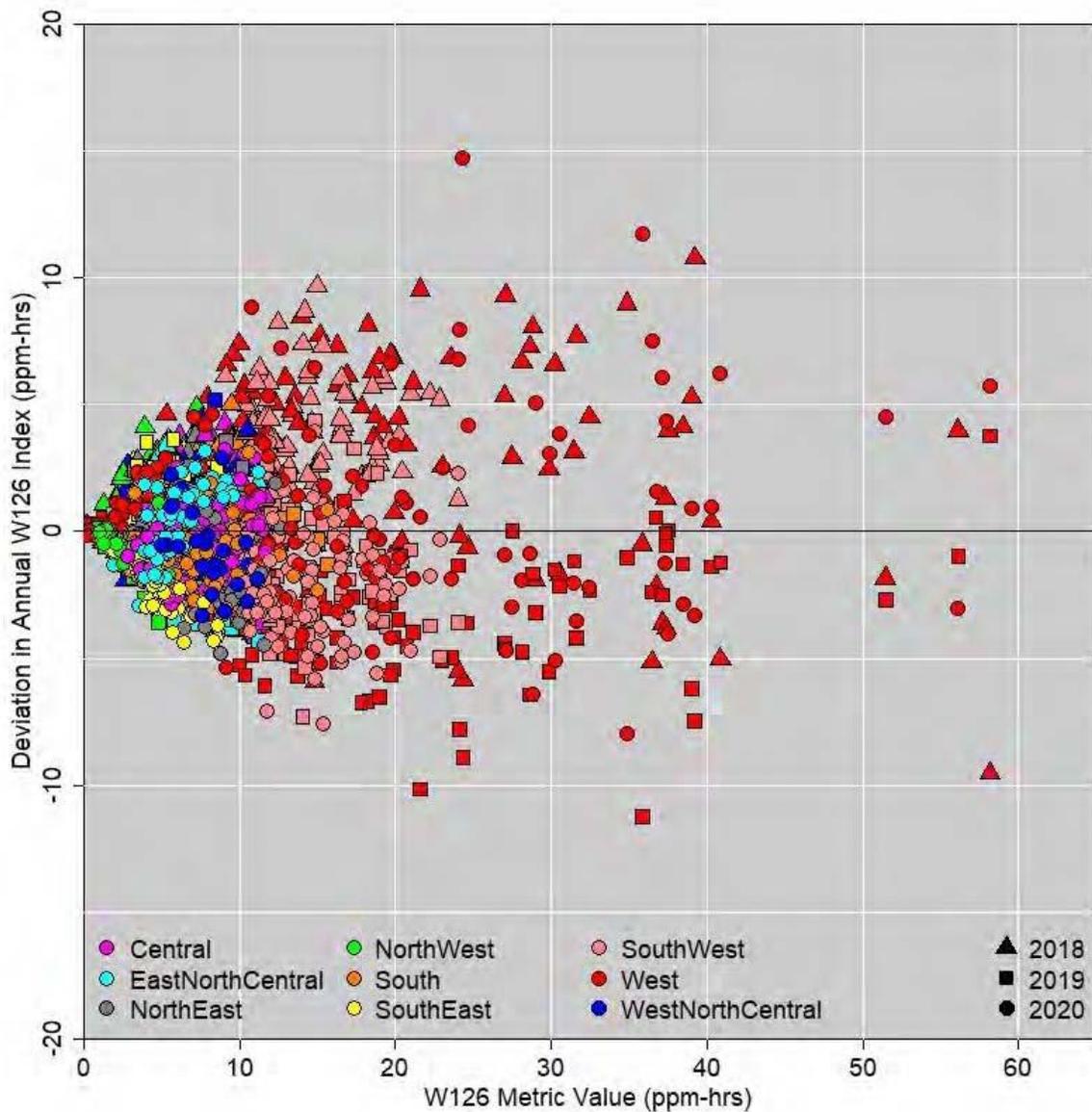


Figure 4D-6. Deviation in annual W126 index values from their respective 3-year averages for all U.S. monitoring sites in 2018-2020.

Figure 4-14. Deviation in annual W126 index value from their respective 3-year averages for all U.S. monitoring sites in 2018-2020. (EPA, 2023, page 4D-14).

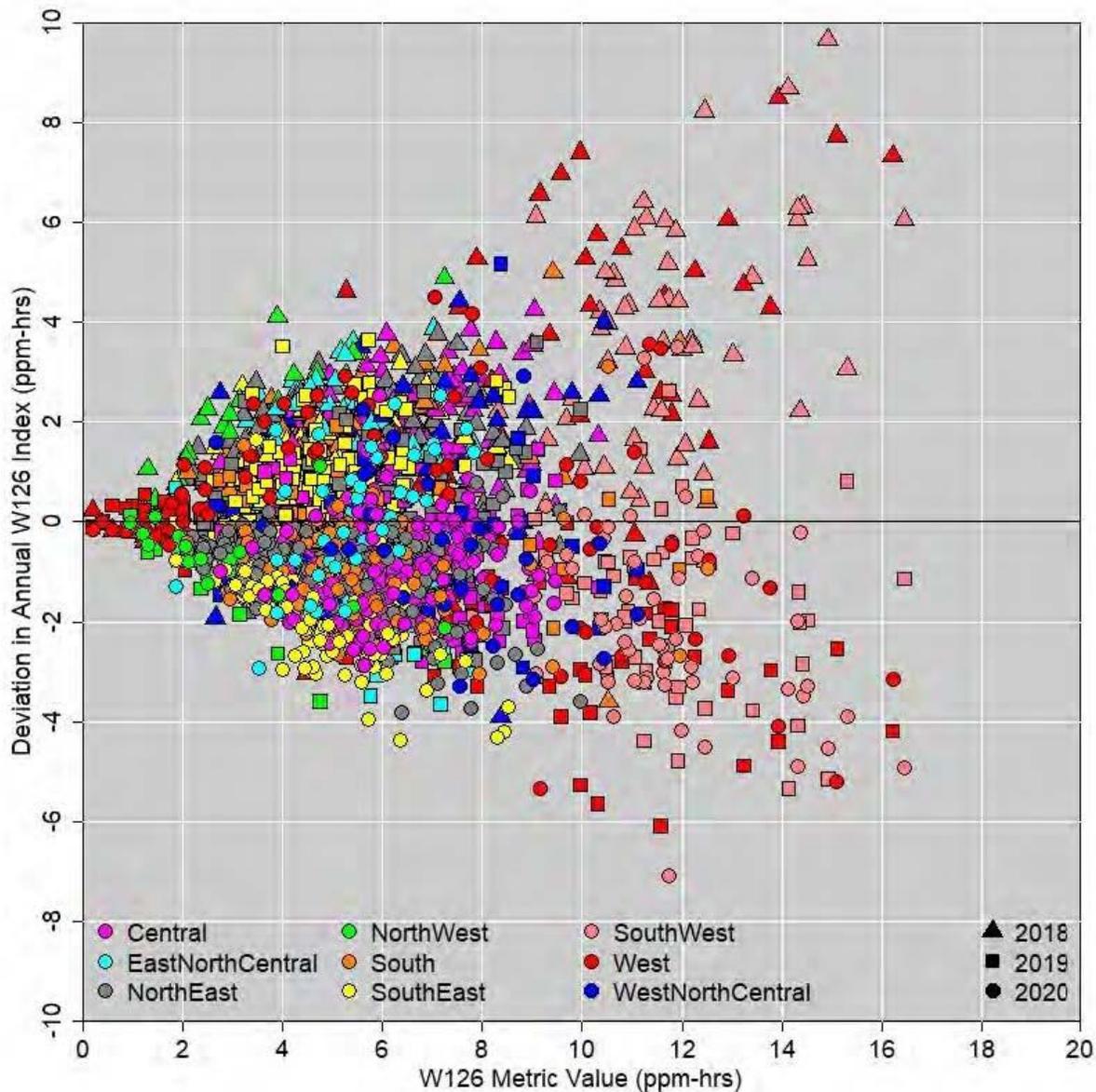


Figure 4D-7. Deviation in annual W126 index values from their respective 3-year averages for all U.S. monitoring sites with 4th max metric values at or below 70 ppb in 2018-2020.

Figure 4-15. Deviation in annual W126 index value from their respective 3-year averages for all U.S. monitoring sites with 4th max metric values at or below 70 ppb in 2018-2020. (EPA, 2023, page 4D-15).

In addition to the deviation analyses for supporting the use of the 3-year versus the annual W126 index, the second draft PA includes a focus on the uncertainty of how estimates for a single year exposure relate to subsequent years in multiyear exposures. The second draft PA (EPA, 2023, page 4-56) notes

Uncertainty is also associated with our understanding of the extent to which RBL estimates for a single year exposure pertain to subsequent years in multiyear exposures. There is limited availability of studies of seasonal growth effects on trees across multiple years (particularly more than two) that have also reported detailed O₃ concentration data throughout the exposure. However, as reported by Lee et al. (2022), tree seedling data for experiments of multiple year exposures do not consistently demonstrate additional response in a second year of exposure. In those analyses, which associated the 2-year exposure response with the single-year W126 index for the most recent year, the 2-year response for three of four species analyzed was not significantly greater than the response after the first year. For the fourth species, while the 2-year response was greater than the single-year response, the effect was not additive; that is, the response at the end of two years was less than twice the response after the first year (Lee et al., 2022).

An available study of multi-year growth effects for aspen (King et al., 2005), which was summarized and assessed in the 2020 and 2013 ISAs with regard the extent to which it confirmed O₃-related biomass impacts estimated using the established E-R functions for aspen. The second draft PA (EPA, 2023, page 4-100) notes that both ISAs reached similar conclusions about general support for the E-R functions across a multiyear study of trees in naturalistic settings.

The second draft PA (EPA, 2023, page 4-101 – 4-103) continues to focus on evidence to support a 3-year average W126 versus the annual W126 exposure metric as follows:

In further considering the evidence and information and its support for use of a single or multiple year W126 index, we consider the concept of cumulative multiyear exposures and associated impacts. In particular, we ask the question of whether applying the E-R functions to a W126 index averaged over multiple years would over- or under-estimate cumulative exposure response, whereas use of a single seasonal exposure metric would not. Multiyear studies reporting results for each year of the study are the most informative to the question of plant annual and cumulative responses to individual years (high and low) over multiple-year periods. However, as summarized in section 4.3.4 above, the evidence is still limited with regard to studies of O₃ effects that report seasonal observations across multiyear periods and that also include detailed hourly O₃ concentration records (to allow for derivation of cumulative exposure index values). There is some recent evidence relevant to this question, e.g., that allows for some evaluation of the predictability of growth impacts from single-year versus multiple-year average exposure estimates. This evidence, described in Section 4.3.3 above, indicated variability among species with regard to the extent to which a multiyear response differed from a single year. Specifically, Lee et al. (2022) found that most species assessed did not exhibit a greater response for two years of O₃ exposure than a single year. One species exhibited a greater reduction in growth after two years exposure than a single year, but the effect was less than

additive. These responses would suggest an over-estimation of cumulative exposure response using a single year W126.

We additionally note aspects of the evidence that reflect variability in organism response under different experimental conditions and the extent to which this variability is represented in the available data, which might also indicate an appropriateness of assessing environmental conditions using a mean across seasons (in recognition of the existence of such year-to-year variability in conditions and responses). For example, among the species for which there are more than two or three experimental datasets comprising the support for the species' E-R function in Lee and Hogsett (1996) (e.g., 14 experimental datasets for aspen [seven for which the E-R function for wild aspen has been derived and seven supporting a function for aspen clones] and 11 for ponderosa pine), the data illustrate appreciable variability in response across experiments (Appendix 4A, Figures 4A-6 and 4A-10). Contributions to this variability may come from several factors, including variability in seasonal response related to variability in non-O₃ related environmental influences on growth, such as rainfall, temperature and other meteorological variables, as well as biological variability across individual seedlings. Stochastic analysis of the experimental-specific E-R functions of Lee and Hogsett (1996) illustrate this variability and its influence on species median RBL (Appendix 4A, Figure 4A-13). An additional variability could also be due to influential aspects of the O₃ air quality on plant growth that are not completely captured by the W126 index, e.g., different patterns of hourly concentrations that yield the same W126 index (see section 4.3.4 and below). **Such variability in the data underlying these E-R functions may further support an average (e.g., 3-year average) over interpretation with a single year precision** (emphasis added).

An additional aspect of considering the evidence and information, is how well the data underlying the E-R functions represent and reflect conditions that are currently being experienced in the U.S., and most importantly, conditions that reflect current air quality patterns when meeting the current standard. Accordingly, we note the appreciable differences between the prevalence of hourly concentrations at or above 100 ppb in exposures on which the E-R functions are based and those common in ambient air.¹¹⁰ As discussed in section 4.3.4 above, the O₃ tree seedling treatments for 92-day W126 index levels below approximately 20 ppm-hrs had N100 counts ranging up above 40. Across the treatments, N100 values range up above 500 (Appendix 4A, Table 4A-7 and Table 4A-8). We find it reasonable to interpret this information, and its contribution to uncertainty in the application of the underlying E-R functions, as supporting a less precise interpretation, such as an average across multiple seasons. We additionally note that this information also highlights the potential significance of higher peak concentrations in terms of estimated effects of concern on vegetation.

Thus, while the E-R functions are based on strong evidence of cumulative seasonal O₃ exposure reducing tree growth, and while they provide for quantitative characterization of the extent of such effects across cumulative seasonal O₃ exposure levels of appreciable magnitude, there is uncertainty associated with the resulting RBL predictions that might be described as an imprecision or inexactitude. The evidence does not indicate single-year seasonal exposure in combination with the established E-R functions to be a better predictor of RBL than a seasonal exposure based on a multiyear average. Accordingly, it is reasonable to conclude that the evidence provides support for use of a multiyear average in assessing the level of protection provided by the current standard from cumulative seasonal exposures related to RBL of concern based on the established E-R functions.¹¹¹ The multiyear, specifically 3-year, average metric also appears to be reasonable for use in the context of the use of RBL as a proxy to represent an array of vegetation-related effects. Thus, upon consideration of all of the factors raised above, we find the use of a multiyear average, and more specifically a 3-year average, W126 index in assessing protection for RBL based on the established tree seedling E-R functions to be reasonable. We also note, as discussed in response to the prior question, the importance of also considering an additional aspect of O₃ air quality, specifically the occurrence of elevated hourly concentrations that influence vegetation exposures of potential concern, in reaching conclusions about the adequacy of the current standard.

The second draft PA (EPA, 2023, pages 4-100 – 4-103) appears to summarize the EPA's argument for using a 3-year average W126 versus a single W126 year. The bolded 4 items below appear to summarize EPA's support for the use of the 3-year average versus a single year for protecting vegetation. Below each of the 4 bolded items is a response. The Agency's evidence is as follows:

- 1. Does the application of E-R functions to a W126 index averaged over multiple years over- or under-estimate cumulative exposure response, in comparison to the use of a single seasonal exposure metric? Multiyear studies reporting results for each year of the study are the most informative to the question of plant annual and cumulative responses to individual years (high and low) over multiple-year periods?**

The second draft PA (EPA, 2023) appears to focus on whether multiple year W126 averages over- or under-estimate cumulative exposure responses in comparison to the use of a single seasonal exposure. The evidence is limited to studies of O₃ effects that report seasonal observations across multiyear periods and that also include detailed hourly O₃ concentration records. There is some recent evidence relevant to this question. However, this evidence indicates variability among species about the extent to which a multiyear response differed from a single year. Most of the E-R functions, using the W126 exposure index, were developed for a single year for an approximate 90-day period. The purpose of developing the E-R functions was to provide an estimate of RYL or RBL losses associated with O₃ exposures that could be useful for assessing risk under ambient O₃ conditions across the U.S. As has been mentioned earlier, the

CASAC (Frey, 2014) did not recommend the use of a three-year averaging period. CASAC favored a single year averaging period, which would provide more protection for annual crops and for the anticipated cumulative effects on perennial species. As mentioned above, the scientific analyses considered in CASAC's review, and the evidence upon which the analyses were based, were from single-year results. The CASAC noted that if a 3-year averaging period were established, then the upper limit would need to be reduced to protect against one-year O₃ peaks. CASAC (Frey, 2014) believed that that the case had not been made that welfare benefits from the stability of a 3-year average were greater than those from using the biologically relevant 1-year value. **As noted earlier, CASAC was concerned about single unusually damaging years that would be obscured if multiyear averages were applied.** There does not exist strong experimental evidence that vegetation can be better protected by averaging cumulative O₃ exposures across multiple years. In June 2014, CASAC (Frey, 2014) was concerned about single unusually damaging years that would be obscured by using a multiyear W126 average exposure. Multiyear averaging obscures single-year cumulative exposures, as well as single-year peak concentrations, whose values may be high enough that they need to be reduced to protect vegetation.

- 2. EPA (2023) additionally notes aspects of the evidence that reflect variability in organism response under different experimental conditions and the extent to which this variability is represented in the available data, which might also indicate an appropriateness of assessing environmental conditions using a mean across seasons (in recognition of the existence of such year-to-year variability in conditions and responses. Contributions to this variability may come from several factors (e.g., rainfall, temperature, other meteorological variables, and biological variability across individual seedlings). In addition, variability could also be due to aspects of the O₃ air quality on plant growth not completely captured by different patterns of hourly concentrations. Such variability in the data underlying these E-R functions may further support an average (e.g., 3-year average) over interpretation with a single year precision (emphasis added).**

The variability of growth response from year to year to O₃ exposures is a reason to focus on each year separately. CASAC (Frey, 2014) noted

Care should be taken not to overstate uncertainties. For example, there is quite a lot of certainty in estimates of biomass loss for forest tree seedling species and crop species for which E-R functions have been developed. Because several dominant crop species have E-R functions, there is a quite a lot of certainty about impacts of ozone on crop yield across most annual cropland in the U.S.

There is considerable uncertainty in extrapolating from the 12 forest tree species to all forest tree species in the U.S. It should be anticipated that there are species of vegetation that are highly sensitive to ozone that do not have E-R functions, and others that are insensitive. It is scientifically justifiable to extrapolate from the known E-R curves, assuming that they are representative of the un-sampled population.

As noted earlier, CASAC favored a single year averaging period, which would provide more protection for annual crops and for the anticipated cumulative effects on perennial species. The scientific analyses considered in CASAC's review, and the evidence upon which the analyses were based, were from single-year results. The CASAC (Frey, 2014) noted that if a 3-year averaging period were established, then the upper limit would need to be reduced to protect against one-year O₃ peaks. CASAC (Frey, 2014) believed that that the case had not been made that welfare benefits, based on the desire to mathematically stabilize the variability of cumulative exposure using a 3-year average, were greater than using a biologically relevant 1-year cumulative exposure. **CASAC was concerned about single unusually damaging years that would be obscured if a multiyear average were applied.** The use of multiyear averaging obscures the identification of elevated cumulative exposures, as well as elevated peak concentrations, whose values may be high enough that they need to be reduced to protect vegetation.

- 3. An additional aspect of considering the evidence and information, is how well the data underlying the E-R functions represent and reflect conditions that are currently being experienced in the U.S., and most importantly, conditions that reflect current air quality patterns when meeting the current standard. The O₃ tree seedling treatments for 92-day W126 index levels below approximately 20 ppm-hrs had N100 counts ranging up above 40. Across the treatments, N100 values range up above 500. We find it reasonable to interpret this information, and its contribution to uncertainty in the application of the underlying E-R functions, as supporting a less precise interpretation, such as an average across multiple seasons.**

The second draft PA (EPA, 2023) focuses on the relevance of the O₃ exposures applied in the experimental treatments for the tree seedling experiments. The second draft PA notes that the O₃ tree seedling treatments for 92-day W126 index levels below approximately 20 ppm-hrs experienced N100 counts ranging up above 40; across the treatments, N100 values ranged up above 500. As discussed earlier, some of the experimental exposure protocols applied in both the NCLAN and tree seedling experiments were designed to explore the different growth effects on vegetation associated with episodic and daily peak exposures (see for example Hogsett et al., 1985) as well as daytime versus nighttime exposure responses. **The simulated (1) daily peak experimental exposure profiles and (2) daytime versus nighttime profiles were not designed to mimic ambient exposures experienced in the U.S.** In many instances, the focus of some of the experiments was on comparing vegetation growth effects associated with (1) episodic treatments that contained growth effects associated with infrequent high hourly average O₃ concentrations with (2) daily peak treatments that contained more frequent high but less magnitude than the concentrations used in the episodic exposures. For example, Lee et al. (2022) report that the total plant biomass response to the daily peak O₃ treatment for the 1989 quaking aspen study (Study 11) was significantly less than that for the episodic treatment with both treatments experiencing similar W126 values. The daily peak experimental exposure profiles, some of which contained very high numbers of elevated hourly average O₃ concentrations, as used in some of the NCLAN and tree seedling experiments, were not designed to mimic ambient exposures experienced in the U.S.

Besides the daily peak treatments which were not designed to mimic ambient exposures, there were treatments in the vegetation experiments where N100 values occurred. As noted in the second draft of the PA (EPA, 2023), Lefohn and Foley (1992) and Lefohn et al. (1997) reported that some of the O₃ treatments in the NCLAN and tree seedling experiments experienced hourly average O₃ concentrations ≥ 100 ppb. While many of the tree seedling experiments experienced high numbers of N100 values in the higher experimental treatments, such was not the case for those treatments that experienced the lower W126 values of interest (e.g., a W126 value of 7 ppm-hrs). Tables 4A-7 and 4A-8 in the second draft PA (EPA, 2023, pages 4A-21 – 4A-26 and 4A-28 – 4A-30) show that W126 exposures associated with levels approximating 7 ppm-hrs experience less infrequent occurrences of hourly average O₃ concentrations ≥ 100 ppb. Lee et al. (2022) report that the most sensitive species in their analysis experienced a biomass loss of 5% at a W126 of 2.5–9.2 ppm-hrs with N100 ranging from 0 to 7 at those exposures. For some of the tree seedling experiments, the N100 values were either 0 or close to 0 when the W126 value approximated 7 ppm-hrs. In that same 2.5–9.2 ppm-hrs range, Lee et al. (2022) report that U.S. O₃ monitoring data from 2016 to 2018 indicated that N100 ranged from 0 to 10, with an N100 of 0 at most monitors. The authors noted that while the episodic peaks have been declining across the U.S. in the past few decades, there are still areas with N100 peaks where the sensitive species identified by Lee et al. (2022) occur.

- 4. Thus, while the E-R functions are based on strong evidence of cumulative seasonal O₃ exposure reducing tree growth, and while they provide for quantitative characterization of the extent of such effects across cumulative seasonal O₃ exposure levels of appreciable magnitude, there is uncertainty associated with the resulting RBL predictions that might be described as an imprecision or inexactitude. The evidence does not indicate single-year seasonal exposure in combination with the established E-R functions to be a better predictor of RBL than a seasonal exposure based on a multiyear average. Accordingly, it is reasonable to conclude that the evidence provides support for use of a multiyear average in assessing the level of protection provided by the current standard from cumulative seasonal exposures related to RBL of concern based on the established E-R functions.**

In the second draft PA (EPA, 2023), EPA makes the case that the multiyear, specifically the 3-year, average metric appears to be reasonable for use in the context of the use of RBL as a proxy to represent an array of vegetation-related effects. The EPA makes the case that there is uncertainty associated with the resulting RBL predictions that the Agency believes can be described as imprecision or inexactitude.

In its 2014 letter to the Administrator (Frey, 2014) concerning the draft PA regarding the substitution of the W126 exposure index with the 8-h primary standard, CASAC indicated within its letter:

The CASAC concurs with the justification in this section that the form of the standard should be changed from the current 8-hr form to the cumulative W126 index and finds that the discussion provides an appropriate and sufficient rationale.

This section clearly demonstrates that ozone-induced injury may occur in areas that meet the current standard. As noted above, the correlative similarity between the current standard and a level of the W126 index of 15 ppm-hrs must not be interpreted to mean that just meeting the current standard is equivalent to just meeting a W126 level of 15 ppm-hrs. ***Most of the analyses found effects below 15 ppm-hrs (many at 10 or even 7 ppm-hrs)*** (emphasis added). Based on review of relevant science, the CASAC concludes that the upper bound of the range that should be considered for the W126 standard should not exceed 15 ppm-hrs. The CASAC does not support a level higher than 15 ppm-hrs. Levels above 15 ppm-hrs should not be included in the revised PA as options for an alternate secondary standard. For example, at 17 ppm-hrs, the median tree species has 6% relative biomass loss, and the median crop species has over 5% loss. These levels are unacceptably high.

The CASAC does not recommend the use of a three-year averaging period. We favor a single-year averaging period, which will provide more protection for annual crops and for the anticipated cumulative effects on perennial species. The scientific analyses considered in this review, and the evidence upon which they are based, are from single-year results. If a 3-year averaging period is established, then the upper limit will need to be reduced to protect against one-year ozone peaks. We consider this further in the response to charge questions for Chapter 6.

The suggestion in Section 6.2 to use a 3-year averaging period is not supported by the available data. We have not supported it in the past nor do we support it here. The primary justification for a 3-year averaging period is to improve the program stability of the classification of regions as being in or out of compliance. The proposed form includes a 3-month period, so it is not nearly as sensitive to extreme events as an hourly or 8-hour averaging period. The case has not been made that welfare benefits from the stability of a 3-year average are greater than those from using the biologically relevant 1-year value. If a 3-year averaging period is implemented, it should be at a lower level than a single-year standard to protect against single unusually damaging years that will be obscured in the average.

In reaching its scientific judgment regarding the indicator, form, averaging time, and range of levels for a revised secondary standard, the CASAC has focused on the scientific evidence for the identification of the kind and extent of adverse effects on public welfare. The CASAC acknowledges that the choice of a level within the range recommended based on scientific evidence is a policy judgment under the statutory mandate of the Clean Air Act. Specifically, the Clean Air Act grants discretion to the Administrator to specify a standard that is “requisite to protect the public welfare from any known or anticipated adverse effects associated with the presence of [the] pollutant in the ambient air” (Section 302(h), 42 U.S.C., §7602(h)). As a policy recommendation, separate from its advice above regarding scientific findings, the CASAC advises that a level of 15 ppm-hrs

is requisite to protect crop yield loss, but that lower levels provide additional protection of crop yield loss. Furthermore, there are specific economically significant crops that may not be protected at 15 ppm-hrs but that would be protected at lower levels. Based on scientific judgment of CASAC, a level of 10 ppm-hrs is required to reduce foliar injury. ***A level of 7 ppm-hours is protective of relative biomass loss for trees. Furthermore, 7 ppm-hours offers additional protection against crop yield loss and foliar injury. Therefore, 7 ppm hours is protective of ecosystem services*** (emphasis added). Thus, lower levels from the recommended range offer a greater degree of protection of more endpoints than do higher levels from the range.

If, as a policy matter, the Administrator prefers to base the secondary standard on a three year averaging period for the purpose of program stability, then the level of the standard should be revised downward such that the annual level in any given year of the three year period would not exceed the scientifically recommended range of annual levels of 7 ppm-hrs to 15 ppm-hrs. For example, if in a three year period the highest annual W126 value is 15 ppm-hrs, and the lowest W126 value associated with a three year average is 13 ppm-hrs, then the appropriate level for the three year average would be 13 ppm-hrs to protect against a peak annual level of 15 ppm-hrs. The final Policy Assessment should quantify the ratio of the annual to three year average W126 values to determine what downward adjustment from the annual levels recommended here is needed if a three year form is selected.

The Administrator's 2020 decision (Federal Register, 2020b) introduced the concept that the W126 index was relatively less controlling of high-concentrations years (whether as a single year index or averaged over three years) than the current 8-h secondary standard. At the 7 ppm-hrs annual W126 level, Fig. 4-8 (Fig. 4-12 in the second draft PA (EPA, 2023, page 4-69)), which was previously presented, the right side of the figure shows that the range of values for the current form of the O₃ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the O₃ NAAQS of 0.070 ppm (i.e., 70 ppb), numerous occurrences greater than 7 ppm-hrs are evident and that these occurrences are not geographically isolated to the western part of the US. The W126 metric and the 8-h O₃ NAAQS each behave differently at levels of 7 ppm-hrs and below. Focusing on the N100 values, the second draft PA (EPA, 2023, page 4-75, Table 4-1) notes for the 2018 to 2020 period that at a W126 level of 7 ppm-hrs and below, 8% of the sites experience N100 values > 0 occurrences. Nine percent of the sites experience N100 values > 0 occurrences at a level of the 3-year 4th Max ≤ 70 ppb. Therefore, the 3-year 4th Max offers no better control for the N100 values than the W126. At an annual W126 level of 7 ppm-hrs, there does not appear to be an advantage of applying the current 8-h form of the 3-year 4th Max to control for the N100 values. Thus, a strategy of using the O₃ NAAQS to limit annual W126 values to 7 ppm-hrs (and below) and N100 values to zero appears to be questionable. As noted earlier, many of the treatments in the tree seedling and crop experiments experienced infrequent or no occurrences of hourly average O₃ concentrations ≥ 100 ppb.

In his 2020 decision, the Administrator (EPA, 2020, page 87334) notes

With regard to the commenters' objection to the EPA's use of a 3-year average in assessing RBL, we note, as an initial matter, that the EPA's focus on a 3-year average of 17 ppm-hrs as a target level relates to an RBL estimate of 5.3%, a value that was also chosen in 2015 in recognition of the prior CASAC advice both with regard to 6% RBL and about considering a lower W126 index target for a 3-year average due to the prior CASAC's concern about "unusually damaging years." In the current review, the CASAC has explicitly considered the EPA's interpretation of 6% in identifying a target of 17 ppm-hrs as a 3-year average, and expressed its view that this target "is still effective in particularly protecting the public welfare in light of vegetation impacts from ozone" (Cox, 2020a, Consensus Responses to Charge Questions p. 21). Accordingly, the EPA disagrees with the comments that 6% RBL and a 3-year average W126 index target of 17 ppm-hrs are too high to inform the Administrator's judgments on O₃ air quality that protects the public welfare; rather, the Administrator continues to find this useful in informing his judgments regarding the public welfare protection provided by the standard, together with a broader consideration of air quality patterns associated with meeting the current standard, such as control of peak hourly concentrations, as described in section III.B.3 below. Further, we refer to the discussion above of how the existing standard, with its current averaging time and form provides the protection from the occurrence of elevated hourly concentrations that may characterize what the prior CASAC described as "unusually damaging years." As discussed above, the available air quality data demonstrate the strong protection provided by the current standard from elevated concentrations that may occur in some years. *As noted above, these analyses indicate that while the current form and averaging time of the existing standard provides control of these concentrations and the associated peak exposures, reliance solely on a standard in the form of the W126 index based standard, as advocated by the commenters, even with a level as low as 7 ppm-hrs cannot be relied on to provide it* (emphasis added).

From the above, it appears that the Administrator in his 2020 decision (Federal Register, 2020, page 87334) believed that to protect vegetation at the 3-year average of a W126 value of 17 ppm-hrs, the current 8-form of the O₃ NAAQS could control for the occurrences of elevated hourly average O₃ concentrations. He noted that "...these analyses indicate that while the current form and averaging time of the existing standard provides control of these concentrations and the associated peak exposures, reliance solely on a standard in the form of the W126 index-based standard, as advocated by the commenters, even with a level as low as 7 ppm-hrs cannot be relied on to provide it."

At the annual W126 level of 7 ppm-hrs, the W126 exposure index appears to protect as well as the current form of the 8-h O₃ NAAQS from elevated O₃ hourly average concentrations. However, 17 ppm-hrs for the 3-year average W126 is not protective of the most sensitive and important vegetation. As noted earlier, Lee et al. (2022) reported for the most sensitive species in their analysis a biomass loss of 5% at W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures. There does not exist strong experimental evidence that vegetation can be better protected by

averaging cumulative O₃ exposures across multiple years. The variability of growth response from year to year to O₃ exposures is a reason to focus on each year separately.

4.6 A Suggested Secondary O₃ NAAQS Standard to Protect Vegetation

Based on the second draft of the PA (EPA, 2023), the EPA has reached the conclusion that the evidence supports a secondary O₃ NAAQS standard that (1) protects against cumulative growth impacts, such as those characterized by the W126 index, as well as (2) protects against high, peak hourly concentrations associated with vegetation effects. In addition, the EPA has reached the conclusion that there are aspects of O₃ air quality not controlled by a cumulative index (i.e., W126 exposure metric) that contribute to the risk of vegetation effects. The EPA believes that the W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb.

Identifying those O₃ exposure regimes experienced in the experimental studies that elicited important vegetation growth and yield effects and relating these specific regimes of interest with ambient air quality levels experienced in the U.S. is important. The EPA notes that some of the experimental treatments used to determine the RBL E-R functions are based upon W126 index values which are experienced in areas of the U.S. that meet the current standard. However, some of these W126 experimental exposures that resulted in the lower W126 values of concern (e.g., W126 values of 7 ppm-hrs) are associated with hourly average O₃ concentrations at/above 100 ppb exhibited in the experiments, which are not necessarily exhibited in some of the areas meeting the current 8-h standard. The EPA believes that the W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly average concentrations, such as those at or above 100 ppb.

The second draft of the PA (EPA, 2023) on page 4-133 notes

We note the inability of the W126 index to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. This is illustrated, for example, by the fact that some of the experimental exposures on which the RBL E-R functions are based reflect W126 index values that may currently occur in areas of the U.S. that meet the current standard and yet the experimental exposures include tens of hours at/above 100 ppb (which have been found to contribute to greater RBL for the same W126 index, as discussed in sections 4.3.3.1.2 and 4.3.4.1) while more than a handful of such hours are rare in areas meeting the standard.

EPA notes above there exist a “handful” of hourly average O₃ concentrations ≥ 100 ppb in areas currently meeting the secondary O₃ NAAQS. In meeting the secondary O₃ NAAQS, the standard must protect against (1) cumulative growth impacts, such as those characterized by the W126 index, as well as (2) high, peak hourly concentrations associated with vegetation effects. The second draft PA (EPA, 2023) notes on page 4-133 that the Administrator may judge it more appropriate to consider an alternative option to the existing secondary standard. Page 4-133 states

However, if the Administrator judges it more appropriate to consider an alternative option to the existing standard to achieve the desired air quality, such an option could be a combination of two standards: (1) a W126 index-based standard for which a range of levels supported by the evidence might extend from 17 ppm-hrs up to somewhat below 24 or 25 ppm-hrs, based on consideration of the current information for visible foliar injury and CYL, as discussed above, well as consideration of median RBL (in its role as a surrogate or proxy) across the range of species for which E-R functions are available; and (2) a standard that controls hourly, peak concentrations, such as an average of upper percentile annual N100 values, and for which a precise range of levels might be expected to achieve a level of peak concentration control comparable to that achieved by the existing standard.

The EPA refers to the N100 indicator metric above. The N100 metric is the number of hourly average O₃ concentrations \geq 100 ppb as applied by Lefohn and Foley (1992) and Lefohn et al. (1997).

The choice of whether to focus on the results associated with (1) the most recent analyses performed by Lee et al. (2022), (2) the analyses summarized by Lee and Hogsett (1996) or (3) the combination of the two analyses is a critical decision point. There are differences between the predicted E-R functions associated with the different methodologies applied by Lee and Hogsett (1996) and the most recent analysis by Lee et al. (2022) published in the peer-reviewed literature. The choice of which study to use or the combination of both studies will affect the decision about whether the current form of the secondary O₃ NAAQS is appropriate for protecting vegetation. Calculating averages mathematically obscures the differences associated with each of the two analyses. While Lee et al. (2022) described the two models, the authors did not state in their paper whether one model was preferred more than the other. However, one might assume that the lead author of both papers (i.e., Dr. Henry Lee) prefers the model selected in the most current analyses as described in Lee et al. (2022) when compared to the model applied in the Lee and Hogsett (1996) analyses.

For the discussion of the form and level of the secondary O₃ NAAQS, focus is on Fig. 4-7 presented earlier in these comments, which is a reproduction of Table 4A-10 in the second draft PA (EPA, 2023). The figure summarizes the results from Lee et al. (2022). The figure shows a summary of the relative biomass loss for sixteen individual tree seedlings and median at various W126 index values identified in the second draft PA (EPA, 2023, page 4A-34). Fig.4-7 below provides estimates of the relative loss for tree biomass at various W126 index values using the composite E-R functions from the analysis by Lee et al. (2022) for each species for each integer W126 index value between 7 ppm-hrs and 30 ppm-hrs. The cross species median of the species-specific composite functions is calculated for all 16 tree species. This table also provides estimates of the number of species for trees that would be below various reference values (e.g., 2% RBL for trees) at various W126 index values. In its 2014 letter to the Administrator (Frey, 2014) concerning the draft PA regarding the substitution of the W126 exposure index with the 8-h primary standard, CASAC indicated levels above 15 ppm-hrs that should not be included as options for an alternate secondary standard.

As noted earlier in Section 4 of these comments, Lee et al. (2022) reported for the most sensitive species in their analysis a biomass loss of 5% at W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures. A W126 value of 17 ppm-hrs, which is identified in the second draft of the PA (EPA, 2023) as an important level to protect vegetation, is not protective of the most sensitive tree species (highlighted in yellow) identified by Lee et al. (2022).

Table 4A-10. RBL for sixteen tree species (Lee et al., 2022), and the median, for W126 index from 30 to 7 ppm-hrs.

W126	Douglas Fir	Virginia Pine	Red maple	Sugar maple	Red Alder	Ponderosa Pine	Quaking Aspen	Tulip Poplar	Eastern White Pine	Black Cherry	American Sycamore	Winged Sumac	Sweetgum	Chestnut Oak	Table Mountain Yellow Buckeye	Median (16 species)	# of Species ≤ 2%	# of Species ≤ 5%	# of Species ≤ 10%	# of Species ≤ 15%	
30	2.9%	4.6%	7.1%	5.2%	15.4%	17.3%	25.9%	35.3%	4.2%	45.8%	19.7%	53.6%	25.7%	3.6%	2.5%	0.0%	11.2%	1	6	7	7
29	2.9%	4.4%	6.9%	5.1%	14.9%	16.7%	24.8%	34.3%	4.0%	44.7%	19.1%	51.1%	19.2%	3.5%	2.4%	0.0%	10.9%	1	6	7	8
28	2.8%	4.3%	6.7%	4.9%	14.4%	16.2%	23.7%	33.4%	3.9%	43.5%	18.5%	48.6%	14.1%	3.4%	2.3%	0.0%	10.4%	1	7	7	9
27	2.7%	4.1%	6.4%	4.7%	13.9%	15.7%	22.6%	32.4%	3.8%	42.4%	17.9%	46.0%	10.1%	3.3%	2.3%	0.0%	8.3%	1	7	7	9
26	2.6%	4.0%	6.2%	4.5%	13.4%	15.1%	21.5%	31.4%	3.6%	41.2%	17.3%	43.4%	7.1%	3.2%	2.2%	0.0%	6.7%	1	7	8	9
25	2.5%	3.8%	6.0%	4.4%	13.0%	14.6%	20.4%	30.4%	3.5%	40.0%	16.7%	40.8%	4.9%	3.0%	2.1%	0.0%	5.5%	1	8	8	10
24	2.4%	3.7%	5.7%	4.2%	12.5%	14.1%	19.3%	29.4%	3.3%	38.7%	16.1%	38.2%	3.4%	2.9%	2.0%	0.0%	5.0%	1	8	8	10
23	2.3%	3.5%	5.5%	4.0%	12.0%	13.5%	18.2%	28.3%	3.2%	37.5%	15.5%	35.6%	2.2%	2.8%	1.9%	0.0%	4.8%	2	8	8	10
22	2.2%	3.4%	5.3%	3.9%	11.5%	13.0%	17.2%	27.3%	3.1%	36.2%	14.8%	33.0%	1.5%	2.7%	1.8%	0.0%	4.6%	3	8	8	11
21	2.1%	3.2%	5.0%	3.7%	11.0%	12.4%	16.1%	26.2%	2.9%	34.9%	14.2%	30.5%	0.9%	2.6%	1.8%	0.0%	4.4%	3	8	8	11
20	2.0%	3.1%	4.8%	3.5%	10.5%	11.9%	15.0%	25.2%	2.8%	33.5%	13.6%	28.0%	0.6%	2.4%	1.7%	0.0%	4.2%	4	8	8	11
19	1.9%	2.9%	4.6%	3.3%	10.0%	11.3%	14.0%	24.1%	2.7%	32.1%	12.9%	25.5%	0.4%	2.3%	1.6%	0.0%	4.0%	4	8	8	12
18	1.8%	2.8%	4.3%	3.2%	9.5%	10.8%	13.0%	23.0%	2.5%	30.7%	12.3%	23.1%	0.2%	2.2%	1.5%	0.0%	3.8%	4	8	9	12
17	1.7%	2.6%	4.1%	3.0%	9.0%	10.2%	12.0%	21.8%	2.4%	29.3%	11.7%	20.8%	0.1%	2.1%	1.4%	0.0%	3.5%	4	8	9	12
16	1.6%	2.5%	3.9%	2.8%	8.5%	9.6%	11.0%	20.7%	2.2%	27.9%	11.0%	18.6%	0.1%	2.0%	1.3%	0.0%	3.3%	5	8	10	12
15	1.5%	2.3%	3.6%	2.6%	8.0%	9.0%	10.1%	19.5%	2.1%	26.4%	10.4%	16.4%	0.0%	1.8%	1.3%	0.0%	3.1%	5	8	10	12
14	1.4%	2.2%	3.4%	2.5%	7.5%	8.5%	9.1%	18.4%	2.0%	24.9%	9.7%	14.4%	0.0%	1.7%	1.2%	0.0%	2.9%	6	8	12	13
13	1.3%	2.0%	3.2%	2.3%	7.0%	7.9%	8.2%	17.2%	1.8%	23.3%	9.1%	12.4%	0.0%	1.6%	1.1%	0.0%	2.7%	7	8	12	13
12	1.2%	1.8%	2.9%	2.1%	6.4%	7.3%	7.3%	16.0%	1.7%	21.7%	8.4%	10.6%	0.0%	1.5%	1.0%	0.0%	2.5%	7	8	12	13
11	1.1%	1.7%	2.7%	1.9%	5.9%	6.7%	6.4%	14.7%	1.5%	20.1%	7.7%	8.9%	0.0%	1.3%	0.9%	0.0%	2.3%	8	8	13	14
10	1.0%	1.5%	2.4%	1.8%	5.4%	6.1%	5.6%	13.5%	1.4%	18.5%	7.0%	7.4%	0.0%	1.2%	0.8%	0.0%	2.1%	8	8	13	14
9	0.9%	1.4%	2.2%	1.6%	4.9%	5.5%	4.8%	12.2%	1.3%	16.8%	6.4%	6.0%	0.0%	1.1%	0.8%	0.0%	1.9%	8	10	13	14
8	0.8%	1.2%	2.0%	1.4%	4.3%	4.9%	4.0%	10.9%	1.1%	15.1%	5.7%	4.7%	0.0%	1.0%	0.7%	0.0%	1.7%	9	12	13	14
7	0.7%	1.1%	1.7%	1.2%	3.8%	4.3%	3.3%	9.6%	1.0%	13.3%	5.0%	3.6%	0.0%	0.9%	0.6%	0.0%	1.5%	9	13	14	15

Figure 4-7. Relative biomass loss (RBL) estimates for sixteen individual tree seedlings and median at various W126 index values using data from Lee et al. (2022) (EPA, 2023, page 4A-34).

In Table 6 in Lee et al. (2022), the authors summarize the predicted relative biomass loss (PRBL) as a function of the 12-h W126 O₃ metric for the sixteen tree species based on the random coefficients predicted total dry weight plant response. Species are listed in order from most sensitive to least sensitive based on the W126 value associated with 5% biomass loss. Ozone rating based on the test of a significant O₃ effect and W126 value associated with 5% biomass loss: 1) sensitive if W126 < 10 ppm-h; 2) intermediate if 10 < W126 < 28 ppm-h; and 3) insensitive if W126 > 28 ppm-h and/or a linear slope not significantly different from zero. The most sensitive tree species in the Lee et al. (2022) analyses included black cherry, ponderosa pine, quaking aspen, red alder, American sycamore, tulip poplar, and winged sumac. The W126 values for the most sensitive species are listed in Table 4-1 as follows:

Table 4-1. W126 value associated with PRBL of 5% (ppm-h) with most sensitive tree species. Source: Adopted from Lee et al. (2022).

Species	Sensitivity	W126 at PRBL of 5%
Black Cherry	Sensitive	2.5
Tulip Poplar	Sensitive	3.5
Ponderosa Pine 1 (harvest 1)	Sensitive	9.6
Ponderosa Pine 2 (harvest 2)	Sensitive	6.0
American Sycamore	Sensitive	7.0
Winged Sumac	Sensitive	8.5
Quaking Aspen	Sensitive	9.0
Red Alder	Sensitive	9.2

The median and mean values for the most sensitive species listed in Table 4-1 are 7.8 ppm-hrs and 6.9 ppm-hrs, respectively. The authors noted that these species are ecologically important and widespread across the U.S. Based on the results in Lee et al. (2022), it is suggested that the part of the secondary O₃ NAAQS that protects against a cumulative growth impact for the most sensitive and vulnerable vegetation species should be set at an annual 3-month maximum 12-h W126 level of 7 ppm-hrs. It is suggested that annual 3-month maximum W126 exposures at and below 7 ppm-hrs will adequately protect the most sensitive species. While it is suggested that the annual 90-day W126 12-h metric be calculated over consecutive 3-month periods beginning on the first day of April through the last day in October, the beginning and end dates for the 3-month cumulative period should be based on scientific, as well as public policy considerations to be made by the Administrator.

In the second draft PA (EPA, 2023, page 4F-3) the EPA specifies how the W126 was calculated in the PA as follows:

The W126 metric values were calculated using the hourly O₃ concentration data in parts per million (80 FR 65374, October 26, 2015). For daytime hours (defined as the 12-hour period from 8:00 AM to 8:00 PM Local Standard Time each day), the hourly concentration values at each O₃ monitoring site were weighted using the following equation:

$$\text{Weighted O}_3 = \text{O}_3 / (1 + 4403 * \exp (-126 * \text{O}_3)).$$

These weighted values were summed over each calendar month, then adjusted for missing data (e.g.; if 80% of the daytime hourly concentrations were available, the sum would be multiplied by 1/0.8 = 1.25) to obtain the monthly W126 index values. Monthly W126 index values were not calculated for months where fewer than 75% of the possible daytime hourly concentrations were available. Next, moving 3-month sums were calculated from the monthly index values, and the highest of these 3-month sums was determined to be the annual W126 index.

Three-month periods spanning multiple years (e.g., November to January, December to February) were not considered in these calculations. The annual W126 index values were averaged across each consecutive 3-year period to obtain the final W126 metric values, with units in parts per million-hours (ppm-hrs). The W126 metric values were rounded to the nearest unit ppm-hr for applications requiring direct comparison to a W126 level (e.g., Table 1), otherwise, all decimal digits were retained. For consistency with the 4th max metric calculations, the W126 metric values were considered valid if hourly O₃ concentration values were available for at least 90% of the daytime hours during the O₃ monitoring season on average across the three years, with a minimum of 75% of the daytime hours during the O₃ monitoring season in any calendar year.

It is suggested that a similar procedure for the determination of the annual W126 exposure index be performed over the beginning and end date period established for the period that determines the maximum 3-month (i.e., 90-day) W126 value. Although not suggested in these comments, should the EPA make the decision to apply a multi-year average of the annual maximum 3-month 12-h W126 metric, it is suggested that a multi-year average be established below the W126 level of 7 ppm-hrs to protect the most sensitive species from single unusually damaging years.

The second component of the secondary O₃ NAAQS as suggested in the second draft of the PA (2023) is a metric that protects against high, peak hourly concentrations associated with vegetation effects. As noted earlier, Lee et al. (2022) reported for the most sensitive species in their analysis a biomass loss of 5% at 12-h W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures. Lee et al. (2022) reported there were few experimental exposures between 2.5 and 9.2 ppm-hrs. Table 2, Table 3, and Table 4 in Lee et al. (2022) summarize the W126 and N100 values for each treatment. Reviewing Tables 4A-7 (pages 4A-21 to 4A-26) and 4A-8 (pages 4A-28 to 4A-30) in the second draft PA (EPA, 2023) and Tables 2, 3, and 4 in Lee et al. (2022), the treatments closest to the W126 levels at RBL values of 5% appear to indicate that most of the sensitive species may have experienced at least one occurrence of an hourly average O₃ concentration ≥ 100 ppb. However, as Lee et al. (2022) note, there is a great deal of uncertainty associated with identifying the exact estimate of the number of hourly average O₃ concentrations ≥ 100 ppb experienced for each of the sensitive species at the RBL value of 5%. There were few experimental treatments associated with W126 values between 2.5 ppm-hrs and 9.2 ppm-hrs. Given the inability to identify the specific number of hourly average O₃ concentrations ≥ 100 ppb associated with the W126 values at the PRBL of 5% identified in Table 4-1 above, it is suggested that the N100 metric be designated as equal to or greater than 1.

The suggestion that the N100 metric be designated as equal to or greater than 1 is based on the observation in the second draft of the PA (EPA, 2023, Table 4-1, page 4-75) that for annual W126 values ≤ 7 ppm-hrs for the period 2018-2020, 5.3% of the sites experienced an N100 value ≥ 1 . In other words, most of the sites where annual W126 ≤ 7 ppm-hrs were experienced exhibited N100 values of 0. Given the high uncertainty associated with quantifying the number of hourly average O₃ concentrations ≥ 100 ppb experienced at the W126 level at PRBL values of 5% for the most sensitive species identified by Lee et al. (2022), it is suggested

that an N100 value equal to or greater than 1 be selected rather than assuming that most of the sensitive species in the experiments experienced N100 values of 0. As noted in Section 7 (research needs) of these comments, additional experiments are recommended using hour-by-hour exposures that mimic ambient exposures. The EPA may wish to consider initiating major research initiatives similar to the National Crop Loss Assessment Network (NCLAN) and tree seedling experiments the Agency funded many years ago. To reduce uncertainty at the lower range of exposures in the development of E-R response functions, high, middle, and low exposures will be required to establish the response curves with additional focus on the lower range (i.e., more treatments than previously applied in the crop and tree seedling experiments). Specifically, more attention should be paid to the lower range of O₃ exposures that include a low number or 0 occurrences of hourly average O₃ concentrations ≥ 100 ppb. In other words, a better definition of the relationship between vegetation effects, W126 exposure, and N100 values in the lower part of the effects curve is required so as to modify, if necessary, the suggested secondary O₃ NAAQS of an N100 metric equal to or greater than 1 that is coupled to the annual 3-month maximum 12-h W126 level of 7 ppm-hrs.

In the second draft PA (EPA, 2023, page 4F-3) the EPA specifies how the N100 metric was calculated in the PA as follows:

The N100 metric was calculated as the maximum number of hours with an hourly O₃ concentration of 100 ppb or greater in the three consecutive calendar months yielding the highest number in a given year. Similarly, the D100 metric was calculated as the maximum number of days with at least one hourly O₃ concentration of 100 ppb or greater in the three consecutive calendar months yielding the highest number in a given year. These metrics were considered valid if the annual data completeness rate for the O₃ monitoring season was at least 75 percent.

Therefore, it is suggested that an annual 3-month maximum 12-h W126 value at and below 7 ppm-h, assessed over the April 1 through October 31 period, associated with a N100 value equal to or greater than 1 will be protective of the most sensitive tree species analyzed in the Lee et al. (2022) analyses. Should the EPA make the decision to apply a multi-year average of the annual maximum 3-month 12-h W126 metric, then it is suggested to establish a multi-year average below a W126 level of 7 ppm-hrs to protect the most sensitive species from single unusually damaging years. It is suggested that that the multi-year average of the N100 value, if greater than 0, be assigned a value of 1.

Based on the second draft of the PA (EPA, 2023), the EPA has reached the conclusion that the evidence supports a secondary O₃ NAAQS standard that (1) protects against cumulative growth impacts, such as those characterized by the W126 index, as well as (2) protects against high, peak hourly concentrations associated with vegetation effects. The EPA notes in the second draft PA (EPA, 2023, page 4-133) that there exist a “handful” of hourly average O₃ concentrations ≥ 100 ppb in areas currently meeting the secondary O₃ NAAQS. As discussed in this subsection, for the most sensitive tree species, Lee et al. (2022) reported in their analysis a biomass loss of 5% at W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures. The N100 values

in the range from 0 to 7 represents a “handful” of hourly average O₃ concentrations ≥ 100 ppb. The suggestion of an annual 3-month maximum 12-h W126 value equal to or less than 7 ppm-h , which is associated with a N100 value equal to or greater than 1, will protect the most sensitive tree species and appears to satisfy the EPA’s observation the current evidence supports a secondary O₃ NAAQS standard that (1) protects against cumulative growth impacts, such as those characterized by the W126 index, as well as (2) protects against high, peak hourly concentrations associated with vegetation effects.

One might assume that if O₃ exposure regimes responsible for growth effects identified in the experimental studies experienced W126 levels and N100 values are similarly experienced under current ambient conditions measured in the U.S., the EPA (as a policy decision implemented by the Administrator) would adopt a secondary O₃ NAAQS, different in form and level than the current 8-h average O₃ NAAQS, to be protective of the most sensitive tree species if the following conditions were to prevail:

- The affected sensitive vegetation is found to be ecologically and/or economically important,
- The existing secondary O₃ NAAQS is unable to control for W126 levels at 7 ppm-hrs, and
- The 3-year 4th Max offers no better control for the N100 values than the W126 at a level of 7 ppm-hrs and below.

The second draft PA (EPA, 2023, page 4-131), in summarizing its available evidence considering the secondary O₃ NAAQS, currently concludes

Thus, consideration of the available information, as discussed above, leads us to conclude that the combined consideration of the body of evidence and the quantitative air quality and exposure analyses, including associated uncertainties, do not call into question the adequacy of the protection provided by the current secondary standard. Rather, this information provides support for the current standard, and thus supports consideration of retaining the current standard, without revision.

More specifically, we first conclude that the evidence continues to support O₃ as the indicator for the secondary NAAQS for photochemical oxidants. Second, we recognize that the evidence supports a standard that protects against cumulative growth impacts, such as those characterized by the W126 index, as well as against high, peak hourly concentrations that are associated with visible foliar injury, as well as reduced tree growth.

The second draft PA (EPA, 2023, pages 4-132 – 4-133) currently concludes

In reaching these conclusions, we have also considered the continued attention given by the CASAC to the W126 index as a potentially suitable metric for a

secondary standard. As discussed in this and other sections above, there are aspects of O₃ air quality not controlled by a cumulative index such as the W126 index, and that contribute risk of vegetation effects. These air quality aspects include occurrences of elevated hourly O₃ concentrations which have demonstrated a role in effects such as reduced growth and visible foliar injury. We note the inability of the W126 index to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. This is illustrated, for example, by the fact that some of the experimental exposures on which the RBL E-R functions are based reflect W126 index values that may currently occur in areas of the U.S. that meet the current standard and yet the experimental exposures include tens of hours at/above 100 ppb (which have been found to contribute to greater RBL for the same W126 index, as discussed in sections 4.3.3.1.2 and 4.3.4.1) while more than a handful of such hours are rare in areas meeting the standard. Thus, we conclude that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. In order to achieve such control, in addition to control of cumulative exposures, to protect against vegetation effects, we conclude that if a secondary standard in the form of the W126 index were to be considered by the Administrator, it should also be accompanied by an additional standard more focused on control of high hourly concentrations.

Based on its analyses in the second draft of the PA (EPA, 2023), the EPA appears to reach the following conclusions for the secondary O₃ NAAQS that appear to be accurate based on the current state of science for the effects of O₃ exposures on vegetation:

- The EPA believes that the evidence supports a standard that protects against (1) cumulative growth impacts, such as those characterized by the W126 index, as well as against (2) high, peak hourly concentrations that are associated with visible foliar injury, as well as reduced tree growth.
- The EPA believes that there are aspects of O₃ air quality not controlled by a cumulative index, such as the W126 index, that contribute to the risk of vegetation effects. These air quality aspects include occurrences of elevated hourly O₃ concentrations, which have demonstrated a role in effects such as reduced growth and visible foliar injury.

Based on its analyses in the second draft of the PA (EPA, 2023), the EPA appears to reach the following conclusions for the secondary O₃ NAAQS that it is suggested requires further modifications on wording to clarify the current state of science more accurately for the effects O₃ exposures on vegetation. The EPA's current conclusion reads as follows:

- The EPA believes that the W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. In its analysis, the EPA notes that some of the experimental exposures on which the RBL E-R functions are based upon W126 index values which are experienced in areas of the U.S. that meet the current

standard. However, some of the experimental exposures include tens of hours at/above 100 ppb, which are not exhibited in current areas meeting the current 8-h standard.

It is suggested that the EPA's current conclusion be modified as follows:

The W126 index is unable to distinguish between O₃ air quality with and without repeated peak hourly concentrations, such as those at or above 100 ppb. In its analysis, the EPA notes that some of the experimental exposures on which the RBL E-R functions are based upon W126 index values which are experienced in areas of the U.S. that meet the current standard. However, some of the experimental exposures include tens of hours at/above 100 ppb, which are not exhibited in areas currently meeting the 8-h standard. However, the experimental treatment W126 levels for the most sensitive tree species and thus, the most vulnerable vegetation, exhibit an infrequent number of hourly average O₃ concentrations ≥ 100 ppb at the 5% PRBL level. Lee et al. (2022) reported for the most sensitive species in their analysis, a biomass loss of 5% at W126 values in the range of 2.5-9.2 ppm-hrs with hourly average O₃ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures.

The EPA's current conclusion reads as follows:

- The EPA believes that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. To achieve such control, in addition to control of cumulative exposures to protect against vegetation effects, the Agency concludes that if a secondary standard in the form of the W126 index were to be considered by the Administrator, the secondary standard should also be accompanied by an additional standard more focused on control of high hourly concentrations.

It is suggested that the current conclusion be modified as follows:

The EPA believes that it cannot be expected that a W126 index-based standard, alone, will always control peak hourly concentrations. To achieve such control, in addition to the control of cumulative exposures to protect against vegetation effects, the Agency concludes that if a secondary standard in the form of the W126 index were to be considered by the Administrator, the secondary standard should include the level of the cumulative W126 exposure index, coupled with the level of the number of elevated hourly average O₃ concentrations, the Administrator best believes would be protective of the most sensitive vegetation.

The EPA's current conclusion reads as follows:

- The EPA believes that the body of evidence does not call into question the adequacy of the protection for vegetation provided by the current secondary standard.

It is suggested that the current conclusion be modified as follows:

The EPA believes that the body of evidence calls into question the adequacy of the protection for vegetation provided by the current secondary standard. The EPA Administrator should consider a cumulative W126 exposure value selected at a level protective of the most sensitive vegetation, coupled with a measure of the number of hourly average O₃ concentrations equal to or greater than 100 ppb (N100) and those levels be established at a W126 value of 7 ppm-hrs accompanied with an N100 value equal to or greater than 1.

The EPA has included the Lee et al. (2022) study in the second draft of the PA (EPA, 2023). The analyses reported by Lee et al. (2022) are an important contribution to the science associated with the effects of O₃ exposures on vegetation. The study's results help bring focus on the appropriate form and level of the secondary O₃ NAAQS. The results of the Lee et al. (2022) analyses have been incorporated into the key tables contained within the second draft of the PA (EPA, 2023). These tables provide the EPA with the necessary information that allow the Agency to recommend to the Administrator the W126 cumulative exposure value, coupled with the range of levels of N100 values, that will protect the most sensitive, vulnerable vegetation from adverse effects associated with O₃ exposures.

5. Acute and Chronic Ozone Effects

The Administrator noted (Federal Register, 2020b - page 87266) that in the Agency's 2015 review of the O₃ NAAQS, that

In the review completed in 2015, the Administrator concluded, in consideration of the then-available health effects information, that an 8-hour averaging time remained appropriate for addressing health effects associated with short-term exposures to ambient air O₃ and that it could effectively limit health effects attributable to both short- and long-term O₃ exposures (80 FR 65348, October 26, 2015).

In its review of the 2015 O₃ NAAQS rulemaking (Federal Register, 2015 – page 65358), the Administrator noted:

In considering estimates of exposures of concern for the 60, 70, and 80 ppb benchmarks within the context of her judgments on adversity, the Administrator notes that, due to interindividual variability in responsiveness, not every occurrence of an exposure of concern will result in an adverse effect. As discussed above (II.B.2.b.i), this point was highlighted by some commenters who opposed revision of the current standard, based on their analysis of effects shown to occur following exposures to 72 ppb O₃. This point was also highlighted by some commenters who advocated for a level of 60 ppb, based on the discussion of O₃-induced inflammation in the proposal. In particular, this latter group of

commenters highlighted discussion from the proposal indicating that “[i]nflammation induced by a single O₃ exposure can resolve entirely but as noted in the ISA (U.S. EPA, 2013a, p. 6-76), ‘continued acute inflammation can evolve into a chronic inflammatory state’” (e.g., ALA et al., p. 48). ***Consistent with these comments, and with her consideration of estimated exposures of concern in the proposal, the Administrator judges that the types of respiratory effects that can occur following exposures of concern, particularly if experienced repeatedly, provide a plausible mode of action by which O₃ may cause other more serious effects. Because of this, as in the proposal, the Administrator is most concerned about protecting against repeated occurrences of exposures of concern*** (emphasis added).

Thus, in the Administrator’s decision in 2015, the Agency believed that both acute and chronic effects could be reduced by reducing the higher hourly average concentrations. As emissions are reduced, the higher part of the distribution of hourly O₃ average concentrations shifts downwards toward the middle part of the hourly average concentration distribution.

For the Administrator’s decision in 2020, the Agency believed that both acute and chronic effects could be reduced by reducing the higher hourly average O₃ concentrations. The Administrator noted in his 2020 (EPA, 2020, page 87300) decision that

Further, in considering an implication of the commenters’ claim that a “long-term standard” is needed in order to provide protection from health effects that may be elicited by long-term exposures to O₃, we note that the impact of standards with short averaging times, such as eight hours, is not limited to reducing short-term exposures. This is because a reduction in magnitude of short-term exposure concentrations (e.g., daily maximum 8-hour concentrations) is also a reduction in exposure to such concentrations over the longer term. For example, a standard, such as the current one, that limits daily maximum 8-hour concentrations to not exceed 70 ppb as a 3-year average of the annual fourth highest value, in addition to limiting the magnitude of concentrations to which a population is exposed in eight hour periods, also limits the frequency to which the population is exposed to such concentrations over the long term. That is, the reduction in frequency of the higher concentrations reduces exposures to such concentrations over the short and long term. Thus, given that, as indicated by the current and established evidence, the O₃ concentrations most likely to contribute to health effects are the higher concentrations, the current standard provides control of exposures to such concentrations over both the short and long term. In light of all of the considerations raised here, we disagree with the commenters (sic) assertion of the need for a long-term O₃ standard.

6. Margin of Safety Considerations

As noted in the Administrator’s Review of the Ozone National Ambient Air Quality Standards document (Federal Register, 2020b – page 87258):

Section 109 (42 U.S.C. 7409) directs the Administrator to propose and promulgate “primary” and “secondary” NAAQS for pollutants for which air quality criteria are issued (42 U.S.C. 7409(a)). Section 109(b)(1) defines primary standards as ones “the attainment and maintenance of which in the judgment of the Administrator, based on such criteria and allowing an adequate margin of safety, are requisite to protect the public health.”¹

The requirement that primary standards provide an adequate margin of safety was intended to address uncertainties associated with inconclusive scientific and technical information available at the time of standard setting (Federal Register, 2020b – page 87258). The margin of safety was also intended to provide a reasonable degree of protection against hazards that research has not yet identified. As noted by the Administrator (Federal Register, 2020b - page 87258):

Thus, in selecting primary standards that include an adequate margin of safety, the Administrator is seeking not only to prevent pollution levels that have been demonstrated to be harmful but also to prevent lower pollutant levels that may pose an unacceptable risk of harm, even if the risk is not precisely identified as to nature or degree. The CAA does not require the Administrator to establish a primary NAAQS at a zero-risk level or at background concentration levels (see *Lead Industries Ass’n v. EPA*, 647 F.2d at 1156 n.51, *Mississippi v. EPA*, 744 F.3d at 1351), but rather at a level that reduces risk sufficiently so as to protect public health with an adequate margin of safety.

In addressing the requirement for an adequate margin of safety, the EPA considers such factors as the nature and severity of the health effects involved, the size of the sensitive population(s), and the kind and degree of uncertainties. The selection of any particular approach to providing an adequate margin of safety is a policy choice left specifically to the Administrator’s judgment. See *Lead Industries Ass’n v. EPA*, 647 F.2d at 1161-62; *Mississippi v. EPA*, 744 F.3d at 1353.

In NAAQS reviews generally, evaluations of how particular primary standards address the requirement to provide an adequate margin of safety include consideration of such factors as the nature and severity of the health effects, the size of the sensitive population(s) at risk, and the kind and degree of the uncertainties present. The Administrator noted that in the 2015 decision, given (1) the consideration of the evidence, (2) exposure and risk information, (3) advice from the CASAC, and (4) public comments, judged that a revised primary standard of 70 ppb, in terms of the 3-year average of the annual 4th highest daily maximum 8-hour average O₃ concentrations, was sufficient to protect public health, including the health of at-risk populations, with an adequate margin of safety (Federal Register, 2020b - page 87285).

As noted above, the decision process for deciding an adequate margin of safety involves consideration of such factors as the nature and severity of the health effects, the size of the sensitive population(s) at risk, and the kind and degree of the uncertainties. One factor that

influences the degree of uncertainty is background O₃. According to a Court decision in 2019, the current rulemaking background O₃ should not directly influence the setting of the level of the NAAQS. The Administrator notes (Federal Register, 2020b - page 87261) that in the decision on subsequent litigation on the 2015 O₃ NAAQS decisions, the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) in its August 19, 2019 decision addressed arguments regarding considerations of background O₃ concentrations, and socioeconomic and energy impacts. Regarding background O₃, the Court rejected the argument that the EPA was required to take background O₃ concentrations into account when setting the NAAQS. The Court found that the text of the Clean Air Act section 109(b) precluded this interpretation because it would mean that if background O₃ levels in any part of the country exceeded the level of O₃ that is requisite to protect public health, the EPA would be obliged to set the standard at the higher nonprotective level. Thus, the Court concluded that the EPA did not act unlawfully or arbitrarily or capriciously in setting the 2015 NAAQS without regard for background O₃.

While background O₃ currently, as per the Court's August 19, 2019 decision, is not a direct consideration in the setting of the level of the O₃ standard, background O₃ is an important consideration for assessing human health effects risks. The risk assessments play an important role in the margin of safety determinations and thus, background O₃, in an indirect manner, can possibly influence the level of the O₃ standard. Background O₃ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total ambient O₃ levels and potentially can influence human health risk assessments associated with margin of safety determinations. The EPA notes that across the ensemble of available modeling studies in the literature, *seasonal mean* daily maximum 8-h background O₃ concentrations are estimated to range from 20–50 ppb (EPA, 2020a - page 1-1). This means that over an entire O₃ season, hourly average background O₃ concentration can be higher than the 20-50 ppb seasonal mean of the daily maximum 8-h range of values. In many cases, mortality and hospital admission risk metrics are based on non-threshold, approximately linear C-R functions, and therefore are sensitive to changes in O₃ along the full range of O₃ concentrations (page 9-30 of the 2014 Health REA (EPA, 2014b)), including the low-level values associated with background O₃. For lung function probabilistic population-based Exposure-Response (E-R) functions risk assessments, the lower concentrations, which are at background O₃ levels, have the potential to contribute to the total risk at low-elevation sites because of the more frequent occurrences than the higher values. As noted above, an adequate margin of safety is a policy choice left specifically to the Administrator's judgment. The greater the contribution of background O₃ to the human health risk assessment, the greater the uncertainty will be to the input into the margin of safety consideration. Thus, because of its importance in affecting the human health risk assessments used in the margin of safety determination, background O₃ in the second draft PA (EPA, 2023) should have been quantified for the 8 cities highlighted in the Agency's modeling analyses (i.e., Atlanta, Boston, Dallas, Detroit, Philadelphia, Phoenix, Sacramento, and St. Louis). However, quantification of background O₃ for the 8 cities was not performed. Therefore, no information was provided in the current O₃ NAAQS reconsideration rulemaking process concerning the contribution of background O₃ to the human health effects risk assessments which provide valuable information on the margin of safety considerations.

7. Future Research

7.1 Health

Areas of future health research are discussed in the second draft PA (EPA, 2023, page 3-115). The authors note that a critical aspect of the Agency's consideration of the evidence and the quantitative risk/exposure estimates is the understanding of O₃ effects below the lowest concentrations studied in controlled human exposure studies, for longer exposures and for different population groups, particularly including people with asthma. The authors highlight areas for future health-related research, model development, and data collection activities to address these uncertainties and limitations in the current scientific evidence. The items identified are as follows:

- 1. An important aspect of risk assessment and characterization to inform decisions regarding the primary standard is our understanding of the exposure-response relationship for O₃-related health effects in at-risk populations. Additional research is needed to more comprehensively assess risk of respiratory effects in at-risk individuals exposed to O₃ in the range of 40 to 80 ppb, and lower, for 6.6 hours while engaged in moderate exertion.**

As noted in the second draft PA (EPA, 2023), the lower hourly average O₃ concentrations shift from the lower values toward the mid-range as emissions are reduced. The upward shifting of the hourly average concentrations from the low-end of the distribution is indicative of background O₃ becoming more and more important in the lower MDA8 concentration range. Thus, suggesting that researchers utilize O₃ concentrations in the background concentration part of the distribution (i.e., 40 ppb) may not provide helpful information for the standard-setting process. I would suggest that additional focus be on the 6.6-hour studies in the 50 ppb range using *variable* exposure methodologies (rather than constant concentration exposure regimes) that resemble current ambient levels. Hazucha et al. (1992) and Adams (2003, 2006a, 2006b) have shown that the higher hourly average O₃ concentrations elicit a greater effect than the middle and lower values. Unless the EPA can provide analyses that constant concentration exposure regimes mimic actual ambient data patterns and these patterns predominate in comparison to the variable hour-by-hour O₃ regimes observed under ambient conditions in the U.S., it is recommended that experimental treatments utilize O₃ exposure regimes experienced under actual ambient conditions.

- 2. Epidemiologic studies assessing the influence of “long-term” or “short-term” O₃ exposures is complicated by a lack of knowledge regarding the exposure history of study populations. Further, existing studies generally focus on either long-term or short-term exposure separately, thereby making it difficult to assess whether a single short-term high-level exposure versus a repeated long-term low-level exposure, or a combination of both short-term high-level and repeated long-term low-level exposures, influence health outcomes of the study subjects. Epidemiologic studies that include exposure measurements across a longer-term assessment period and can simultaneously assess the impact of these various elements of exposure (i.e., magnitude, frequency, durations, and pattern) are needed.**

While it may appear that to reduce acute O₃ health effects requires a short-term standard, such as the 4th highest 8-h daily maximum exposure metric and that a long-term average concentration standard is required to reduce chronic O₃ health effects, such is not necessarily the case. In 2015, the EPA believed that by implementing a control strategy that reduced the higher concentrations that concentrations of concern for “chronic” effects would also be reduced (Federal Register, 2015 – page 65399). The EPA (Federal Register, 2015 – page 65358) commented on how the Agency chose to reduce “chronic” and “acute” O₃ exposures for the protection of human health. The Agency believed that the reduction of the repeated occurrences of exposures of concern would reduce both “chronic” and “acute” health effects. The EPA stated:

...This point was also highlighted by some commenters who advocated for a level of 60 ppb, based on the discussion of O₃-induced inflammation in the proposal. In particular, this latter group of commenters highlighted discussion from the proposal indicating that “[i]nflammation induced by a single O₃ exposure can resolve entirely but, as noted in the ISA (U.S. EPA, 2013a, p. 6-76), ‘continued acute inflammation can evolve into a chronic inflammatory state’” (e.g., ALA et al., p. 48). Consistent with these comments, and with her consideration of estimated exposures of concern in the proposal, the Administrator judges that the types of respiratory effects that can occur following exposures of concern, particularly if experienced repeatedly, provide a plausible mode of action by which O₃ may cause other more serious effects. ***Because of this, as in the proposal, the Administrator is most concerned about protecting against repeated occurrences of exposures of concern*** (emphasis added).

The EPA then commented (Federal Register, 2015 – page 65358) on the reduction of the higher concentrations and how these reductions not just influence the highest MDA8 concentrations, but also those values that are below these highest levels. In other words, by reducing the peak exposures, there is a cascading of the upper end of the distribution of O₃ concentrations down toward the mid-level values. The EPA stated as follows:

...In addition, though the available information does not support the identification of specific benchmarks below 60 ppb that could be appropriate for consideration for at-risk populations, and though CASAC did not recommend consideration of any such benchmarks, the EPA expects that a revised standard with a level of 70 ppb will also reduce the occurrence of exposures to O₃ concentrations at least somewhat below 60 ppb (U.S. EPA, 2014a, Figures 4-9 and 4-10). Thus, even if some members of at-risk populations may experience effects following exposures to O₃ concentrations somewhat below 60 ppb, a revised level of 70 ppb would be expected to reduce the occurrence of such exposures. Therefore, the EPA has considered O₃ exposures that could be relevant for at-risk populations such as children and people with asthma, and does not agree that controlled human exposure studies reporting respiratory effects in healthy adults following exposures to 60 ppb O₃ necessitate a standard level below 70 ppb.

Thus, given EPA's conclusions reached in the 2015 O₃ rulemaking process as summarized above, it is suggested that researchers, as well as policy makers, be aware that a long-term epidemiological study does not have to employ exposure metrics based on long-term average concentrations. As shown earlier, EPA noted in the second draft PA (EPA, 2023) that for sites experiencing emission reductions, exposure metrics that use averaging over longer time periods of hourly O₃ measurements, such as the 6-month (April-September) average of daytime (8am-7pm) O₃ concentrations, show inconsistent trends with only about half of the sites exhibiting decreases in this metric and most other sites exhibiting no trend. Earlier in these comments, it was shown, using the same hourly data, that sites experiencing emission reductions have annual average O₃ concentrations showing increasing trends, while the 4th highest MDA8 values experience decreasing trends. This behavior was associated with the lower hourly average concentrations shifting upward toward the mid-values as emission reductions occurred. Thus, because of this behavior, long-term average exposure metrics appear to have serious limitations for assessing risks associated with O₃ exposures. Both annual average and other long-term average metrics are influenced by the titration of O₃ by NO as NO_x emissions are reduced to protect the public's health and welfare. Simon et al. (2016) suggest that because increases in O₃ occur at low O₃ concentrations at times when NO_x emission reductions occur, it is increasingly important for health studies to evaluate whether the shape of the concentration-response relationship changes at these lower O₃ levels.

If the health studies continue to report that the shape of the concentration-response relationship at lower O₃ levels reflects the same shape as the upper O₃ levels, then risk analyses may continue to show only marginal improvement as emissions are reduced. As discussed in Section 3.2.1, Fig. 3-33 (presented below) (reproduced from Fig. 9-8 on Page 9-32 from the 2014 HREA (EPA, 2014b)) illustrates the percent reduction in exposures and risks after just meeting alternative standards relative to just meeting the 2008 NAAQS of 75 ppb. In this plot, each row represents one of the key analytical results and each column gives the results for 2007 and 2009 for each urban study area. The scales are the same between analyses, and as such, it is informative to examine both the overall patterns of change between alternative standards, and the absolute value of the percent reductions in risk metrics between analyses. The top row is the Exposure > 60 ppb; the second row is the Lung Function Risk (dFEV₁ > 10%); the third row is Mortality; the fourth row is Hospital Admissions. The risks associated with mortality and hospital admissions are much less than the risks associated with Exposure > 60 ppb and Lung Function Risk (dFEV₁ >

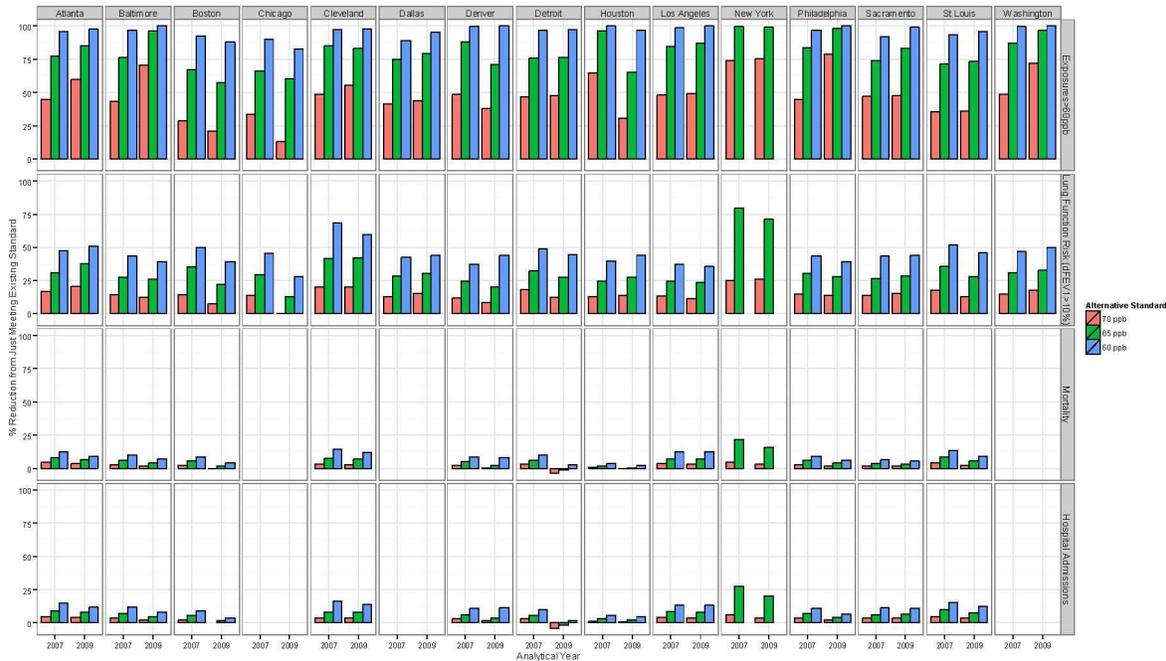


Figure 9-8. Comparison of the Percent Reduction in Key Risk Metrics for Alternative Standard Levels Relative to Just Meeting the Existing 75 ppb Standard.

Figure 3-33 (from Section 3.2.1 in these comments). Source: Figure 9-8 from EPA (2014b).

10%). This is because mortality and hospital admission risk metrics are based on non-threshold, approximately linear C-R functions, and therefore are sensitive to changes in O₃ along the full range of O₃ concentrations (page 9-30 of the 2014 Health REA (EPA, 2014b)). As explained by the Agency (EPA, 2014b), because O₃ in the lower concentration range may shift upward as the result of NO_x emission reductions, this can lead to increases in risk on some days, which can lead to a net increase or decrease in risk over the entire year, depending on whether the days with increased risk exceed days with decreased risk (generally due to a preponderance of days with lower O₃ concentrations).

It is suggested that investigators performing long-term epidemiological studies consider using an exposure metric focused on repeated acute exposures whose effects accumulate over time. One example of such a metric might be the one described by Lefohn, Hazucha, Shadwick, and Adams (2010a). The authors described a sigmoidal weighting scheme for hourly average O₃ concentrations. The weighting scheme addresses the nonlinearity of response (i.e., the greater effect of higher O₃ concentrations over the mid- and low-range values) on an hourly basis. The weighting scheme focused on the use of daily O₃ exposures that were integrated over time. The authors described a W90 exposure index for use in assessing FEV₁ decrements. The scheme is shown in Fig. 7-1 below. The form of the W90 index is $\sum w_i \times C_i$ with weight $w_i = 1/[1 + M \times \exp(-A \times C_i/1000)]$, where $M = 1400$, $A = 90$, and where C_i is the hourly average O₃ concentration in units of ppb. The W90 index has units of ppb-hrs. The weighting scheme of this exposure metric utilizes a mathematical approach that provides greater focus on the higher hourly average O₃ concentrations than the lower values. The W90 exposure metric, while similar

to the W126 index used in assessing vegetation risk, has a slightly different form. The application of the W90 exposure index might result in greater mortality and hospital admission estimated risks than reported previously. In the 2014 Health REA (EPA, 2014b), the mortality and hospital admission risks were based on non-threshold, approximately linear C-R functions.

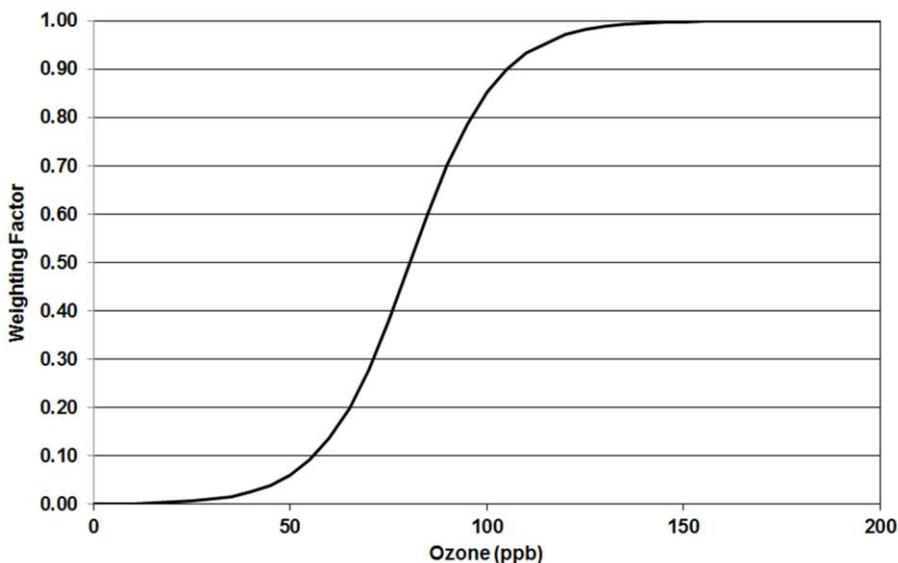


Figure 7-1. The weighting applied to hourly average ozone values for the calculation of the W90 exposure index (see Lefohn et al., 2010a).

7.2 Vegetation

Areas of future vegetation research are discussed in the second draft PA (EPA, 2023, page 4-135).

- 1. Additional controlled exposure studies of effects, such as biomass impacts, that include multiple exposure levels within the lower range of exposures associated with ambient air quality conditions common today, extend over multiple years, and include the collection of detailed O₃ concentration data over the exposure would reduce uncertainty in estimates of effects across multiple-year periods and at the O₃ exposures common today. Also needed is evaluation of such datasets with regard to the role of peak concentrations in combination with that of cumulative seasonal exposures (e.g., as quantified by metrics such as the W126 and SUM06 indices).**

As demonstrated in the information provided in Tables 4A-7 and 4A-8 in the second draft PA (EPA, 2023, pages 4A-21 – 4A-26 and pages 4A-28 – 4A-30), additional experiments are needed using hour-by-hour exposures that mimic ambient exposures. Perhaps the EPA may wish

to consider initiating major research initiatives similar to the National Crop Loss Assessment Network (NCLAN) and tree seedling experiments the Agency funded many years ago. To reduce uncertainty at the lower range of exposures in the development of E-R response functions, high, middle, and low exposures will be required with additional focus on the lower range (i.e., more treatments than previously applied in the crop and tree seedling experiments). Specifically, more attention should be paid to the lower range of O₃ exposures that include a low number or 0 occurrences of hourly average O₃ concentrations \geq 100 ppb. In other words, a better definition of the relationship between vegetation effects, W126 exposure, and N100 values in the lower part of the effects curve is required.

In addition, because of climate change affecting the distribution of hourly average O₃ concentrations (see for example, the American Lung Association's 2022 State of the Air Report, ALA, 2022), future vegetation experiments should focus on the application of realistic ambient hourly average concentration distributions that include treatments exhibiting both high and low hourly average values. In the second draft PA (EPA, 2023), much has been discussed about the elevated hourly average O₃ concentrations used in the vegetation experiments and comparing the levels of these hourly concentrations to ambient levels exhibited in today's environment. It should not be necessarily assumed that O₃ exposures will continue to decrease over time as NO_x and VOC emissions are reduced. As noted by Zanis et al. (2022), further climate change research is suggested on O₃ extremes in selected polluted regions based on Earth System Models because this work would be an asset for air pollution policies as heat waves and air pollution episodes pose a serious threat to human health and may worsen under future climate change depending on various climate change modeling scenarios.

8. References

- Adams, W.C., Savin, W.M., Christo, A.E., 1981. Detection of ozone toxicity during continuous exercise via the effective dose concept. *Journal of Applied Physiology* 51 (2), 415–422.
- Adams, W.C.; Ollison, W.M. 1997. Effects of prolonged simulated ambient ozone dosing patterns on human pulmonary function and symptomatology. Pittsburgh, PA: Air & Waste Management Association.
- Adams, W.C. 2003. Comparison of chamber and face mask 6.6-hour exposure to 0.08 ppm ozone via square-wave and triangular profiles on pulmonary responses. *Inhalation Toxicology* 15: 265-281.
- Adams, W.C. 2006a. Comparison of chamber 6.6-h exposures to 0.04 - 0.08 ppm ozone via square-wave and triangular profiles on pulmonary responses. *Inhalation Toxicology* 18, 127-136.
- Adams, W.C. 2006b. Human pulmonary responses with 30-minute time intervals of exercise and rest when exposed for 8 hours to 0.12 ppm ozone via square-wave and acute triangular profiles. *Inhalation Toxicology* 18, 413-422.

- Akriditis, D., Zanis, P., Pytharoulis, I., Mavrakis, A., Karacostas, Th. 2010. A deep stratospheric intrusion event down to the earth's surface of the megacity of Athens. *Meteorology and Atmospheric Physics* 109, 9-18.
- ALA, 2022. American Lung Association. State of the Air 2022. American Lung Association, Washington, DC. <https://www.lung.org/our-initiatives/healthy-air/sota/>.
- Albers, J.R., Perlwitz, J., Butler, A.H., Birner, T., Kiladis, G.N., Lawrence, Z.D., Manney, G.L., Langford, A.O., Dias, J. 2018. Mechanisms governing interannual variability of stratosphere-to-troposphere ozone transport. *Journal of Geophysical Research: Atmospheres*, 123, 234–260. <https://doi.org/10.1002/2017JD026890>.
- Albers, J.R., Butler, A.H., Langford, A.O., Elsbury, D., Breeden, M.L. 2022. Dynamics of ENSO-driven stratosphere-to-troposphere transport of ozone over North America. *Atmos. Chem. Phys.*, 22, 13035–13048, 2022. <https://doi.org/10.5194/acp-22-13035-2022>.
- Altshuller, A.P, Lefohn, A.S. 1996. Background ozone in the planetary boundary layer over the United States. *J Air Waste Manag Assoc* 46: 134-141. <http://dx.doi.org/10.1080/10473289.1996.10467445>.
- Amiro, B.D., Gillespie, T.J., Thurtell, G.W. 1984. Injury response of *Phaseolus vulgaris* to ozone flux density. *Atmos. Environ.* 18: 1207-1215.
- Ashmore, M.R. 1984. Effects of ozone on vegetation in the United Kingdom. Proceedings of the OECD workshop on ozone; February 29-March 2, 1984, Goteburg, Sweden. Goteburg, Sweden: Swedish Environmental Research Institute.
- Bennett, J.P., Oshima, R.J., Lippert, L.F. 1979. Effects of ozone on Injury and dry matter partitioning in pepper plants. *Environ. Exp. Bot.* 19: 33-39.
- Bergweiler, C.J., Manning, W.J. Chevone, B.I. 2008. Seasonal and diurnal gas exchange differences in ozone-sensitive common milkweed (*Asclepias syriaca* L.) in relation to ozone uptake. *Environ Pollut* 152: 403-415. <http://dx.doi.org/10.1016/j.envpol.2007.06.019>.
- Blanchard, C.L., Shawb, S.L., Edgerton, E.S., Schwab, J.L. 2019. Emission influences on air pollutant concentrations in New York State: I. ozone. *Atmos. Environ.* X3 (2019) 10033: 1-12. <https://doi.org/10.1016/j.aeaoa.2019.100033>.
- Blanchard, C.L., Hidy, G.M. 2019. Ozone response to emission reductions in the southeastern United States. *Atmos. Chem. Phys.*, 18, 8183–8202, 2018. <https://doi.org/10.5194/acp-18-8183-2018>.
- Bowman, H., Turnock, S., Bauer, S.E., Tsigaridis, K., Deushi, M., Oshima, N., O'Connor, F.M., Horowitz, L., Wu, T., Zhang, J., Kubistin, D., Parrish, D.D. 2022. Changes in anthropogenic precursor emissions drive shifts in the ozone seasonal cycle throughout the northern

midlatitude troposphere. *Atmos. Chem. Phys.*, 22, 3507–3524. <https://doi.org/10.5194/acp-22-3507-2022>.

Chen, L.-W.A., Chien, L.-C., Li, Y., Lin, G. 2020. Nonuniform impacts of COVID-19 lockdown on air quality over the United States. *Science of the Total Environment* 745. <https://doi.org/10.1016/j.scitotenv.2020.141105>.

Cooper, O.R., Stohl, A., Hübler, G., Hsie, E.Y., Parrish, D.D., Tuck, A.F., Kiladis, G.N., Oltmans, S.J., Johnson, B.J., Shapiro, M., Moody, J.L., Lefohn, A.S. 2005. Direct transport of mid-latitude stratospheric ozone into the lower troposphere and marine boundary layer of the tropical Pacific Ocean. *Journal of Geophysical Research* 110, D23310, doi:10.1029/2005JD005783.

Cox, LA (2020a). Letter from Louis Anthony Cox, Jr., Chair, Clean Air Scientific Advisory Committee, to Administrator Andrew R. Wheeler. Re: CASAC Review of the EPA's Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards (External Review Draft – October 2019). February 19, 2020. EPA-CASAC-20-003. Office of the Administrator, Science Advisory Board Washington, DC. Available at: [https://yosemite.epa.gov/sab/sabproduct.nsf/264cb1227d55e02c85257402007446a4/4713D217BC07103485258515006359BA/\\$File/EPA-CASAC-20-003.pdf](https://yosemite.epa.gov/sab/sabproduct.nsf/264cb1227d55e02c85257402007446a4/4713D217BC07103485258515006359BA/$File/EPA-CASAC-20-003.pdf).

Cristofanelli, P., Bonasoni, P., Tositti, L., Bonafè, U., Calzolari, F., Evangelisti, F., Sandrini, S., Stohl, A. 2006. A 6-year analysis of stratospheric intrusions and their influence on ozone at Mt. Cimone (2165 m above sea level). *Journal of Geophysical Research* 111, D03306, doi:10.1029/2005JD006553.

Cristofanelli, P., Bracci, A., Sprenger, M., Marinoni, A., Bonafè, U., Calzolari, F., Duchi, R., Laj, P., Pichon, J.M., Roccatò, F., Venzac, H., Vuillermoz, E., Bonasoni, P. 2010. Tropospheric ozone variations at the Nepal Climate Observatory-Pyramid (Himalayas, 5079m.a.s.l.) and influence of deep stratospheric intrusion events. *Atmospheric Chemistry and Physics* 10, 6537–6549. doi:10.5194/acp-10-6537-2010.

Dai, L., Feng, Z., Pan, X., Xu, Y., Li, P., Lefohn, A.S., Harmens, H. Kobayashi, K. 2019. The detoxification by apoplastic antioxidants is insufficient to remove the harmful effects of elevated ozone in tobacco, soybean and poplar. *Environ. Pollut.* 245: 380-388. DOI: <https://doi.org/10.1016/j.envpol.2018.11.030>.

Danielsen, E.F. 1968. Stratospheric-tropospheric exchange based on radioactivity, ozone and potential vorticity. *J. Atmos. Sci.* 25, 502-518.

Danielsen, E.F. 1974; The natural stratosphere of 1974, CIAP Monogr. I, edited by Grobecker A.J., pp 155-122. DOT-TST-75-51. US Dept. of Transp., Washington, D.C.

Danielsen, E.F., Mohnen, V.A. 1977. Project Duststorm report: Ozone transport, in situ measurements and meteorological analyses of tropopause folding. *J. Geophys. Res.* 82, 5867-5877.

- Dantas, G., Siciliano, B., França, B.B., da Silva, C.M., Arbilla, G. 2020 The impact of COVID-19 partial lockdown on the air quality of the city of Rio de Janeiro, Brazil. *Science of the Total Environment* 729 (2020). <https://doi.org/10.1016/j.scitotenv.2020.139085>.
- Davies, T.D., Schuepbach, E. 1994. Episodes of high ozone concentrations at the earth's surface resulting from transport down from the upper troposphere/lower stratosphere: a review and case studies. *Atmospheric Environment* 28, 53-68.
- Davis, D. D., Orendovici, T. 2006. Incidence of ozone symptoms on vegetation within a National Wildlife Refuge in New Jersey, USA. *Environ. Pollut.* 143:555-564.
- Dolwick, P., Akhtar, F., Baker, K., Possiel, N., Simon, H., Tonnesen, G. 2015. Comparison of background ozone estimates over the western United States based on two separate model methodologies. *Atmospheric Environment* 109: 282-296, doi: 10.1016/j.atmosenv.2015.01.005.. <http://dx.doi.org/10.1016/j.atmosenv.2015.01.005>.
- Drechsler-Parks, D.M., Horvath, S.M., Bedi, J.F., 1990. The “effective dose” concept in older adults exposed to ozone. *Experimental Gerontology* 25, 107–115.
- Duncan, B. N., Lamsal, L. N., Thompson, A. M., Yoshida, Y., Lu, Z., Streets, D. G., Hurwitz, M. M., Pickering, K. E. 2016. A space-based, high-resolution view of notable changes in urban NO_x pollution around the world (2005–2014), *J. Geophys. Res. Atmos.*, 121, 976–996. doi:10.1002/2015JD024121.
- Emberson, L., Ashmore, M.R., Cambridge, H.M., Simpson, D., Tuovinen, J.P. 2000a. Modelling stomatal ozone flux across Europe. *Environ Pollut* 109: 403-413. [http://dx.doi.org/10.1016/S0269-7491\(00\)00043-9](http://dx.doi.org/10.1016/S0269-7491(00)00043-9).
- Emberson, L.D., Wieser, G., Ashmore, M.R. 2000b. Modelling of stomatal conductance and ozone flux of Norway spruce: Comparison with field data. *Environ Pollut* 109: 393-402. [http://dx.doi.org/10.1016/S0269-7491\(00\)00042-7](http://dx.doi.org/10.1016/S0269-7491(00)00042-7).
- Emery, C., Jung, J., Downey, N., Johnson, J., Jimenez, M., Yarwood, G., Morris, R. 2012. Regional and global modeling estimates of policy relevant background ozone over the United States. *Atmospheric Environment* 47, 206-217.
- Evans, G, Finkelstein, P., Martin, B., Possiel, N., Graves, M. 1983. Ozone measurements from a network of remote sites. *Journal of the Air Pollution Control Association*, 33, No. 4, 291-296. DOI: 10.1080/00022470.1983.10465575.
- Federal Register. Vol. 73, No. 60 / Thursday, March 27, 2008 / Rules and Regulations. National Ambient Air Quality Standards for Ozone, 40 CFR Part 50 and 58, pp 16436-16514.
- Federal Register. Vol. 75, No. 11 / Tuesday, January 19, 2010 / Proposed Rules. National Ambient Air Quality Standards for Ozone, pp. 2938-3052.

- Federal Register. Vol. 80, No. 206 / Monday, October 26, 2015. National Ambient Air Quality Standards for Ozone, 40 CFR Part 50, 51, 52, 53, and 58, pp 65292-65468.
- Federal Register. Vol. 85, No. 158 / Friday, August 14, 2020a. Draft Review of the Ozone National Ambient Air Quality Standards, 40 CFR Part 50, pp 49830-49917.
- Federal Register. Vol. 85, No. 251 / Friday, December 31, 2020b. Review of the National Ambient Air Quality Standards for Ozone, 40 CFR Part 50, pp. 87256-87351.
- Feng, Z., Büker, P., Pleijel, H., Emberson, L., Karlsson, P.E., Uddling, J. 2017. A unifying explanation for variation in ozone sensitivity among woody plants. *Global Change Biol* 24: 78-84. <http://dx.doi.org/10.1111/gcb.13824>.
- Fiore, A.M., Oberman, J.T., Lin, M.Y., Zhang, L., Clifton, O.E., Jacob, D.J., Naik, V., Horowitz, L.W., Pinto, J.P., Milly, G.P. 2014. Estimating North American background ozone in U.S. surface air with two independent global models: Variability, uncertainties, and recommendations. *Atmos Environ* 96: 284-300. <http://dx.doi.org/10.1016/j.atmosenv.2014.07.045>
- FLAG. 2010. Federal Land Managers' Air Quality Related Values Work Group (FLAG). Phase I Report-Revised (2010). Natural Resource Report NPS/NRPC/NRR—2010/232. National Park Service, U.S. Department of the Interior.
- Folinsbee, L.J., Drinkwater, B.L., Bedi, J.B., Horvath, S.M., 1978. The influence of exercise on the pulmonary function changes due to exposure to low concentrations of ozone. In: Folinsbee, L.J., Wagner, J.A., Borgia, J.F., Drinkwater, B.L., Gliner, J.A., Bedi, J.B. (Eds.), *Environmental Stress: Individual Human Adaptations*. Academic Press, New York, NY, pp. 125–145.
- Folinsbee L.J., McDonnell, W.F., Horstman, D.H. 1988. Pulmonary function and symptom responses after 6.6-hour exposure to 0.12 ppm ozone with moderate exercise. *J Air Waste Manag Assoc* 38: 28-35.
- Fowler, D., Cape, J.N. 1982. Air pollutants in agriculture and horticulture. In: Unsworth MH, Ormrod DP. eds. *Effects of Gaseous Air Pollution in Agriculture and Horticulture*. London: Butterworth Scientific.
- Frey, H.C. 2014. Letter from Dr. H. Christopher Frey, Chair, Clean Air Scientific Advisory Committee to Honorable Gina McCarthy, Administrator, US EPA. Re: CASAC Review of the EPA's Second Draft Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards. June 26, 2014. EPA-CASAC-14-004. Office of the Administrator, Science Advisory Board Washington, DC. Available at: <https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P100JR6F.txt>.

- Goldberg, D.L., Anenberg, S.C., Griffin, D., McLinden, C.A., Lu, Z., Streets, D.G. 2020. Disentangling the Impact of the COVID-19 Lockdowns on Urban NO₂ From Natural Variability. *Geophysical Research Letters*, 47, e2020GL089269. <https://doi.org/10.1029/2020GL089269>.
- Goumenaki, E., González-Fernández, I., Barnes, J. 2021. Ozone uptake at night is more damaging to plants than equivalent day-time flux. *Planta* 253(3):75. doi: 10.1007/s00425-021-03580-w. <https://doi.org/10.1007/s00425-021-03580-w>.
- Grantz, D.A., Zhang, X.J., Massman, W.J., Den Hartog, G., Neumann, H.H., Pederson, J.R. 1995. Effects of stomatal conductance and surface wetness on ozone deposition in field-grown grape. *Atmos Environ* 29: 3189-3198. [http://dx.doi.org/10.1016/1352-2310\(95\)00129-M](http://dx.doi.org/10.1016/1352-2310(95)00129-M).
- Grantz, D.A., Zhang, X.J., Massman, W., Delany, A., Pederson, R. 1997. Ozone deposition to a cotton (*Gossypium hirsutum* L) field: Stomatal and surface wetness effects during the California Ozone Deposition experiment. *Agr Forest Meteorol* 85: 19-31. [http://dx.doi.org/10.1016/S0168-1923\(96\)02396-9](http://dx.doi.org/10.1016/S0168-1923(96)02396-9).
- Grantz, D.A., Vu, H.B., Heath, R.L., Burkey, K.O. 2013. Demonstration of a diel trend in sensitivity of *Gossypium* to ozone: a step toward relating O₃ injury to exposure or flux. *J Exp Bot* 64: 1703-1713. <http://dx.doi.org/10.1093/jxb/ert032>.
- Grulke, N.E., Alonso, R., Nguyen, T., Cascio, C., Dobrowolski, W. 2004. Stomata open at night in pole-sized and mature ponderosa pine: Implications for O₃ exposure metrics. *Tree Physiol* 24: 1001-1010. <http://dx.doi.org/10.1093/treephys/24.9.1001>.
- Guo, J.J., Fiore, A.M., Murray, L.T., Jaffe, D.A., Schnell, J.L., Moore, C.T., Milly, G.P. 2018. Average versus high surface ozone levels over the continental USA: Model bias, background influences, and interannual variability. *Atmos Chem Phys* 18: 12123-12140. <http://dx.doi.org/10.5194/acp-18-12123-2018>.
- Haagenson, P.L., Shapiro, M.A., Middleton, P., Laird, A.R. 1981. A case study relating high ground level ozone to enhanced photochemistry and isentropic transport from the stratosphere. *J. Geophys. Res.* 86, 5231-5237.
- Hazucha, M.J. 1987. Relationship between ozone exposure and pulmonary function changes. *Journal of Applied Physiology* 62, 1671–1680.
- Hazucha, M.J., Folinsbee, L.J., Seal E. 1992. Effects of steady-state and variable ozone concentration profiles on pulmonary function. *Am Rev Respir Dis* 146: 1487-1493.
- Hazucha, M., Lefohn, A.S. 2007. Nonlinearity in Human Health Response to Ozone: Experimental Laboratory Considerations *Atmospheric Environment*. 41: 4559-4570.

- Heath, R.L., Lefohn, A.S., Musselman R.C. 2009. Temporal processes that contribute to nonlinearity in vegetation responses to ozone exposure and dose. *Atmos Environ* **43**: 2919-2928.
- Heck, W.W., Dunning, J.A., Hindawi, I.J. 1966. Ozone: nonlinear relation of dose and injury in plants. *Science* 151, 577–578.
- Heck, W.W.; Tingey, D.T. 1971. Ozone time-concentration model to predict acute foliar injury. In: Englund, H. M.; Beery, W. T., eds. Proceedings of the second international clean air congress; December 1970; Washington. DC. New York, NY: Academic Press; pp. 249-255.
- Henderson, W.R., Reinert, R.A. 1979. Yield response of four fresh market tomato cultivars after acute ozone exposure in the seedling stage. *J. Am. Soc. Hortic. Sci.* 104: 754-759.
- Henderson, R. 2006. Letter from Dr. Rogene Henderson, Chair, Clean Air Scientific Advisory Committee to Honorable Stephen L. Johnson, Administrator, US EPA. Re: CASAC Peer Review of the Agency's 2nd Draft Ozone Staff Paper October 24, 2006. EPA-CASAC-07-001. Office of the Administrator, Science Advisory Board U.S. EPA HQ, Washington DC. Available at: <https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P1000WO7.txt>.
- Hildebrand E., Skelly J. M., and Fredericksen T. S. 1996. Foliar response of ozone-sensitive hardwood tree species from 1991 to 1993 in the Shenandoah National Park, Virginia. *Can J. For. Res.* 26, 658-669.
- Hocking, W.K., Carey-Smith, T., Tarasick, D.W., Argall, P.S., Strong, K., Rochon, Y., Zawadzki, I., Taylor, P.A. 2007. Detection of stratospheric ozone intrusions by wind profiler radars. *Nature* 450, 281–284.
- Hogg, A., Uddling, J., Ellsworth, D., Carroll, M.A., Pressley, S., Lamb, B., Vogel, C. 2007. Stomatal and non-stomatal fluxes of ozone to a northern mixed hardwood forest. *Tellus B Chem Phys Meteorol* 59: 514-525. <http://dx.doi.org/10.1111/j.1600-0889.2007.00269.x>.
- Hogrefe, C., Henderson, B., Tonnesen, G., Mathur, R., Matichuk, R. 2021. Multiscale Modeling of Background Ozone: Research Needs to Inform and Improve Air Quality Management. *EM Magazine*. Air and Waste Management Association, Pittsburgh, PA, 1-6, (2020).
- Hogsett, W.E., Tingey, D.T., Holman, S.R. 1985. A programmable exposure control system for determination of the effects of pollutant exposure regimes on plant growth. *Atmos Environ* **19**: 1135–1145.
- Horstman, D.H., Folinsbee, L.J., Ives, P.J., Abdul-Salaam, S., McDonnell, W.F. 1990. Ozone concentration and pulmonary response relationships for 6.6-hour exposures with five hours of moderate exercise to 0.08, 0.10, and 0.12 ppm. *Am J Respir Crit Care Med* **142**: 1158-1163.

- Huang, X., Ding, A., Gao, J., Zheng, B., Zhou, D., et al. 2020. Enhanced secondary pollution offset reduction of primary emissions during COVID-19 lockdown in China. *National Science Review*, nwaal37, <https://doi.org/10.1093/nsr/nwaa137>.
- Jaffe, D.A., Cooper, O.R., Fiore, A.M., Henderson, B.H., Tonnesen, G.S., Russell, A.G., Henze, D.K., Langford, A.O., Lin, M., Moore, T. 2018. Scientific assessment of background ozone over the US: Implications for air quality management. 6. <http://dx.doi.org/10.1525/elementa.309>.
- Jin, X, Fiore, AM, Murray, LT, Valin, LC, Lamsal, LN, Duncan, B, Folkert Boersma, K, De Smedt, I, Abad, GG, Chance, K and Tonnesen, GS (2017). Evaluating a Space-Based Indicator of Surface Ozone-NO_x-VOC Sensitivity Over Midlatitude Source Regions and Application to Decadal Trends: Space-Based Indicator of O₃ Sensitivity. *Journal of Geophysical Research: Atmospheres*, Volume 122, Issue 19, pp. 10,439-10,461. <https://agupubs.onlinelibrary.wiley.com/doi/full/10.1002/2017JD026720>.
- Junge, C. 1962. Global ozone budget and exchange between stratosphere and troposphere. *Tellus* 14, 363–377.
- Kim, C.S., Alexis, N.E., Rappold, A.G., Kehrl, H., Hazucha, M.J., Lay, J.C., Schmitt, M.T., Case, M., Devlin, R.B., Peden, D.B., Diaz-Sanchez, D. 2011. Lung function and inflammatory responses in healthy young adults exposed to 0.06 ppm ozone for 6.6 hours. *Am. J. Respir. Crit. Care Med.* 183: 1215-1221. <http://dx.doi.org/10.1164/rccm.201011-1813OC>.
- King, J.S., Kubiske, M.E., Pregitzer, K.S., Hendrey, G.R., McDonald, E.P., Giardina, C.P., Quinn, V.S., Karnosky, D.F. 2005. Tropospheric O₃ compromises net primary production in young stands of trembling aspen, paper birch and sugar maple in response to elevated atmospheric CO₂. *New Phytol* 168(3): 623-635.
- Kohut, R. 2007. *Handbook for Assessment of Foliar Ozone Injury on Vegetation in the National 30 Parks: Revised Second Edition*. Kohut, R.
- Köllner, B., Krause, GHM. 2003. Effects of two different ozone exposure regimes on chlorophyll and sucrose content of leaves and yield parameters of sugar beet (*Beta Vulgaris* L.) and rape (*Brassica Napus* L.). *Water Air Soil Poll.* **144**: 317–332. Koplitz, S., Simon, H., Henderson, B., Liljegren, J., Tonnesen, G., Whitehill, A., Wells, B. 2021. Changes in Ozone Chemical Sensitivity in the United States from 2007 to 2016. *ACS Environmental Au* 0, 0, pp <https://doi.org/10.1021/acsenvironau.1c00029>.
- Kulle, T.J., Sauder, L.R., Hebel, J.R., Chatham, M.D., 1985. Ozone response relationships in healthy nonsmokers. *American Review of Respiratory Diseases* 132, 36–41.
- Lamarque, J.-F., Hess, P.G. 1994. Cross-tropopause mass exchange and potential vorticity budget in a simulated tropopause folding. *Journal of Atmospheric Science* 51, 2246-2269.

- Langford, A.O., Aikin, K.C., Eubank, C.S., Williams, E.J. 2009. Stratospheric contribution to high surface ozone in Colorado during springtime. *Geophysical Research Letters* 36, L12801. <http://dx.doi.org/10.1029/2009GL038367>.
- Langford, A.O., Alvarez, R.J, Brioude, J., Fine, R., Gustin, M.S., Lin, M.Y., Marchbanks, R.D., Pierce, R.B., Sandberg, S.P., Senff, C.J., Weickmann, A.M., Williams, E.J. 2017. Entrainment of stratospheric air and Asian pollution by the convective boundary layer in the southwestern US. *J Geophys Res Atmos* 122: 1312-1337. <http://dx.doi.org/10.1002/2016JD025987>.
- Langford et al. 2022. The Fires, Asian, and Stratospheric Transport–Las Vegas Ozone Study (FAST-LVOS). *Atmos. Chem. Phys.*, 22, 1707–1737. <https://doi.org/10.5194/acp-22-1707-2022>.
- Lapina, K., Henze, D.K., Milford, J.B., Huang, M., Lin, M., Fiore, A.M., Carmichael, G., Pfister, G.G., Bowman, K. 2014. Assessment of source contributions to seasonal vegetative exposure to ozone in the US. *J. Geophys. Res. Atmos.* 119(1): 324-340.
- Larsen, R.I., Heagle, A.S., Heck, W.W. 1983. An air quality data analysis system for interrelating effects, standards, and needed source reductions: Part 7. An O₃ -502 leaf injury mathematical model. *J. Air Pollut. Control Assoc.* 33: 198-207.
- Larsen, R. I., Heck, W.W. 1984. An air quality data analysis system for interrelating effects, standards, and needed source reductions: Part 8. An effective mean O₃ crop reduction mathematical model. *J. Air Pollut. Control Assoc.* 34: 1023-1034.
- Laughner, J., Cohen, R. 2019. Direct observation of changing NO_x lifetime in North American cities. *Science*. 366 (6466): 723-727. DOI: 10.1126/science.aax6832.
- Le, T., Wang, Y., Liu, L., Yang, J., Yung, Y.L., Li, G., Seinfeld, J.H. 2020. Unexpected air pollution with marked emission reductions during the COVID-19 outbreak in China. *Science* 10.1126/science.abb7431.
- Lee, E.H., Hogsett, W.E. 1996. Methodology for calculating inputs for ozone secondary standard benefits analysis part II. Office of Air Quality Planning and Standards. Research Triangle Park, NC.
- Lee, E.H., Hogsett, W.E. 1999. Role of concentrations and time of day in developing ozone exposure indices for a secondary standard. *J Air Waste Manage Assoc* 49: 669–681.
- Lee, E.H., Andersen, C.P., Beedlow, P.A., Tingey, D.T., Koike, S., Dubois, J.-J., Kaylor, S.D., Novak, K., Rice, R.B., Neufeld, H.S., Herrick, J.D. 2022. Ozone exposure-response relationships parametrized for sixteen tree species with varying sensitivity in the United States, *Atmospheric Environment* (2022), doi: <https://doi.org/10.1016/j.atmosenv.2022.119191>.

- Lefohn, A.S., Benedict, H.M. 1982. Development of a mathematical index that describes ozone concentration, frequency, and duration. *Atmospheric Environment* 16:2529-2532.
- Lefohn, A.S., Tingey, D. T. 1984. The co-occurrence of potentially phytotoxic concentrations of various gaseous air pollutants. *Atmospheric Environment*. 18(11):2521-2526.
- Lefohn, A. S., Davis, C. E., Jones, C. K., Tingey, D. T., Hogsett, W. E. 1987. Co-Occurrence Patterns of Gaseous Air Pollutant Pairs at Different Minimum Concentrations in the United States. *Atmospheric Environment*. 21(11):2435-2444.
- Lefohn, A.S., Runeckles, V.C. 1987. Establishing standards to protect vegetation - Ozone exposure/dose considerations. *Atmospheric Environment* 21:561-568.
- Lefohn, A.S., Lawrence, J.A., Kohut, R.J. 1988. A comparison of indices that describe the relationship between exposure to ozone and reduction in the yield of agricultural crops. *Atmospheric Environment* 22:1229-1240.
- Lefohn, A.S., Foley, J.K. 1992. NCLAN Results and their Application to the Standard-Setting Process: Protecting Vegetation from Surface Ozone Exposures, *J Air Waste Manag Assoc* 42(8): 1046-1052. <https://www.tandfonline.com/doi/abs/10.1080/10473289.1992.10467049>.
- Lefohn, A.S., Foley, J.K. 1993. Establishing Relevant Ozone Standards to Protect Vegetation and Human Health: Exposure/Dose-Response Considerations, *Air & Waste*, 43:1, 106-112, DOI: 10.1080/1073161X.1993.10467111. <https://doi.org/10.1080/1073161X.1993.10467111>.
- Lefohn, A.S., Jackson, W., Shadwick, D.S., Knudsen, H.P. 1997. Effect of surface ozone exposures on vegetation grown in the southern Appalachian Mountains: Identification of possible areas of concern. *Atmospheric Environment* 31(11): 1695-1708. [https://doi.org/10.1016/S1352-2310\(96\)00258-0](https://doi.org/10.1016/S1352-2310(96)00258-0).
- Lefohn, A.S., Shadwick, D.S., Ziman, S.D. 1998. The difficult challenge of attaining EPA's new ozone standard. *Environmental Science & Technology* 32(11):276A-282A.
- Lefohn, A.S., Oltmans, S.J., Dann, T., Singh, H.B. 2001. Present-day variability of background ozone in the lower troposphere. *J. Geophys. Res.* 106(D9):9945-9958, doi:10.1029/2000JD900793.
- Lefohn, A.S., Hazucha, M.J., Shadwick, D., Adams, W.C. 2010a. An alternative form and level of the human health ozone standard. *Inhalation Toxicology* 22:999-1011.
- Lefohn, A. S., Shadwick, D., Oltmans, S. J. 2010b. Characterizing changes in surface ozone levels in metropolitan and rural areas in the United States for 1980–2008 and 1994–2008. *Atmos. Environ.* 44 (39):5199–5210.
- Lefohn, A.S., Wernli, H., Shadwick, D., Limbach, S., Oltmans, S.J., Shapiro, M. 2011. The importance of stratospheric-tropospheric transport in affecting surface ozone concentrations

in the Western and Northern Tier of the United States. *Atmospheric Environment* 45:4845-4857.

- Lefohn, A.S., Wernli, H., Shadwick, D., Oltmans, S.J., Shapiro, M. 2012. Quantifying the frequency of stratospheric-tropospheric transport affecting enhanced surface ozone concentrations at high- and low-elevation monitoring sites in the United States. *Atmospheric Environment* 62: 646-656.
- Lefohn, A.S., Emery, C., Shadwick, D., Wernli, H., Jung, J., Oltmans, S.J. 2014. Estimates of background surface ozone concentrations in the United States based on model-derived source apportionment. *Atmospheric Environment* 84:275-288.
<http://dx.doi.org/10.1016/j.atmosenv.2013.11.033>.
- Lefohn, A. S., Malley, C. S., Simon, H., Wells, B., Xu, X., Zhang, L., Wang, T. 2017. Responses of human health and vegetation exposure metrics to changes in ozone concentration distributions in the European Union, United States, and China. *Atmospheric Environment*, 152: 123-145. doi:10.1016/j.atmosenv.2016.12.025.
- Lefohn, A.S., Malley, C.S., Smith, L., Wells, B., Hazucha, M., Simon, H., Naik, V., Mills, G., Schultz, M.G., Paoletti, E., De Marco, A., Xu, X., Zhang, L., Wang, T., Neufeld, H.S., Musselman, R.C., Tarasick, T., Brauer, M., Feng, Z., Tang, T., Kobayashi, K., Sicard, P., Solberg, S., and Gerosa, G. 2018. Tropospheric ozone assessment report: global ozone metrics for climate change, human health, and crop/ecosystem research. *Elem Sci Anth*, 6(1):28. DOI: <https://doi.org/10.1525/elementa.279>.
- Lefohn, A.S. 2019a. Comments on the First Draft of the EPA Integrated Scientific Assessment Document (External Review Draft, EPA/600/R-19/093, September 2019) Docket ID No. EPA-HQ-ORD-2018-0274-0034.
- Lefohn, A.S. 2019b. Comments on the First Draft of the EPA Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards (External Review Draft, EPA-452/P-19-002, October 2019) Docket ID No. EPA-HQ-OAR-2018-0279-0027.
- Lefohn, A.S. 2020. Comments on Draft Review of the Ozone National Ambient Air Quality Standards. Docket ID No. EPA-HQ-OAR-2018-0279-0421.
- Lefohn, A.S. 2022. Comments on Draft Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards, External Review Draft, EPA-452/D-22-002, April 2022). Docket ID No. EPA-HQ-OAR-2018-0279-0593.
- Li et al. 2020. Characterizing sources of high surface ozone events in the southwestern US with intensive field measurements and two global models. *Atmos. Chem. Phys.*, 20, 10379–10400. <https://doi.org/10.5194/acp-20-10379-2020>.
- Lin, M., Fiore, A.M., Cooper, O.R., Horowitz, L.W., Langford, A.O., Levy II, H., Johnson, B.J., Naik, V., Oltmans, S.J., Senff, C.J. 2012a. Springtime high surface ozone events over the

western United States: Quantifying the role of stratospheric intrusions. *Journal of Geophysical Research* 117, D00V22, doi:10.1029/2012JD018151.

- Lloyd, K.L., Davis, D.D., Marini, R.P., Decoteau, D.R. 2020. Response of Sensitive and Resistant Snap Bean Genotypes to Nighttime Ozone Concentration. *J. AMER. SOC. HORT. SCI.* 145(6):331–339. 2020. <https://doi.org/10.21273/JASHS04808-19>.
- Lu, X., Hong, J., Zhang, L., Cooper, O.R., Schultz, M.G., Xu, X., Wang, T., Gao, M., Zhao, Y., Zhang, Y. 2018. Severe surface ozone pollution in China: a global perspective. *Environmental Science & Technology Letters* 5 (8), 487-494. DOI: 10.1021/acs.estlett.8b00366.
- Ludwig, F.L., Reiter, E., Shelar, E., Johnson, W.B. 1977. The relation of oxidant levels to precursor emissions and meteorological features: v. I, analysis and findings. Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards; report no. EPA-450/3-77-022a. Available from: NTIS, Springfield, VA; PB-275 001.
- Mahato, S., Pal, S., Ghosh, K. G. 2020. Effect of lockdown amid COVID-19 pandemic on air quality of the megacity Delhi, India. *Science of the Total Environment* 730. <https://doi.org/10.1016/j.scitotenv.2020.139086>.
- Male, L., Preston, E., Neely, G. 1983. Yield response curves of crops exposed to SO₂ time series. *Atmos. Environ.* 17: 1589-1593.
- McDonald-Buller, E.C., Allen, D.T., Brown, N., Jacob, D.J., Jaffe, D., Kolb, C.E., Lefohn, A.S., Oltmans, S., Parrish, D.D., Yarwood, G., Zhang, L. 2011. Establishing Policy Relevant Background (PRB) Ozone Concentrations in the United States. *Environmental Science & Technology.* 45(22):9484-97.
- McDonnell, W.F., Kehrl, H.R., Abdulsalaam, S., Ives, P.J., Folinsbee, L.J., Devlin, R.B., Oneil, J.J., Horstman, D.H. 1991. Respiratory response of humans exposed to low levels of ozone for 6.6 hours. *Arch Environ Health* 46:145–150.
- McGrath, J.M., Betzelberger, A.M., Wang, S., Shook, E., Zhu, X.G., Long, S.P., Ainsworth, E.A. 2015. An analysis of ozone damage to historical maize and soybean yields in the United States. *Proc Natl Acad Sci USA* 112: 14390-14395. <http://dx.doi.org/10.1073/pnas.1509777112>.
- McLaughlin, S.B., Shriner, D.S., McConathy, R.K., Mann, L.K. 1979. The effects of SO₂ dosage kinetics and exposure frequency on photosynthesis and transpiration of kidney beans (*Phaseolus vulgaris* L.). *Environ. Exp. Bot.* 19: 179-191.
- Matyssek, R., Sandermann, H., Wieser, G., Booker, F., Cieslik, S., Musselman, R., Ernst, D. 2008. The challenge of making ozone risk assessment for forest trees more mechanistic [Review]. *Environ Pollut* 156: 567-582. <http://dx.doi.org/10.1016/j.envpol.2008.04.017>.

- Miller, F.J., Schlosser, P.M., Janszen, D.B. 2000. Haber's rule: a special case in a family of curves relating concentration and duration of exposure to a fixed level of response for a given endpoint. *Toxicology* **149** (1): 21-34.
- Mills, G., Pleijel, H., Braun, S., Büker, P., Bermejo, V., et al. 2011. New stomatal flux-based critical levels for ozone effects on vegetation. *Atmos Environ* 45: 5064-5068. <http://dx.doi.org/10.1016/j.atmosenv.2011.06.009>.
- Mills, G., et al. 2018. Tropospheric Ozone Assessment Report: Present-day tropospheric ozone distribution and trends relevant to vegetation. *Elem Sci Anth*, 6: 47. DOI: <https://doi.org/10.1525/elementa.302>.
- Musselman, R.C., Oshima, R.J., Gallavan, R.E. 1983. Significance of pollutant concentration distribution in the response of 'red kidney' beans to ozone. *J Am Soc Hortic Sci* **108**: 347–351.
- Musselman, R.C., Huerta, A.J., McCool, P.M., Oshima, R.J. 1986. Response of beans to simulated ambient and uniform ozone distributions with equal peak concentration. *J Am Soc Hortic Sci* 111: 470–473.
- Musselman, R.C., Younglove, T., McCool, P.M. 1994. Response of *Phaseolus vulgaris* L. to differing ozone regimes having identical total exposure. *Atmos Environ* 28: 2727–2731
- Musselman, R.C., Massman, W.J. 1999. Ozone flux to vegetation and its relationship to plant response and ambient air quality standards. *Atmos Environ* 33: 65-73.
- Musselman, R.C., Minnick, T. 2000. Nocturnal stomatal conductances and ambient air quality standards for ozone. *Atmos. Environ.* 34, 719-733.
- Musselman, R.C., Lefohn, A.S., Massman, W.J., Heath, R.L. 2006. A critical review and analysis of the use of exposure- and flux-based ozone indices for predicting vegetation effects. *Atmos Environ* 40(10): 1869-1888.
- Naranjo, L., "Volatile Trees," NASA.gov, 20 November 2011. Retrieved: 9 November 2019. <https://earthdata.nasa.gov/learn/sensing-our-planet/volatile-trees>.
- Neufeld, H.S., Sullins, A., Sive, B.C., Lefohn, A.S. 2019. Spatial and temporal patterns of ozone at Great Smoky Mountains National Park and implications for plant responses. *Atmos. Environ.*: X 2, 100023. DOI: <https://doi.org/10.1016/j.aeaoa.2019.100023>.
- Nussbaum, S., Geissmann, M., Fuhrer, J. 1995. Ozone exposure–response relationships for mixtures of perennial ryegrass and white clover depend on ozone exposure patterns. *Atmos Environ* **29**: 989–995.

- Ohara, T., Akimoto, H., Kurokawa, J., Horii, N., Yamaji, K., Yan, X., Hayasaka, T. 2007. An Asian emission inventory of anthropogenic emission sources for the period 1980–2020, *Atmos. Chem. Phys.*, 7, 4419–4444, doi:10.5194/acp-7-4419-2007.
- Oksanen, E., Holopainen, T. 2001. Responses of two birch (*Betula pendula* Roth. clones to different ozone profiles with similar AOT40 exposure. *Atmos Environ* 35: 5245–5254.
- Oltmans, S.J., Lefohn, A.S., Harris, J.M., Tarasick, D.W., Thompson, A.M., Wernli, H., Johnson, B.J., Novelli, P.C., Montzka, S.A., Ray, J.D., Patrick, L.C., Sweeney, C., Jefferson, A., Dann, T., Davies, J., Shapiro, M., Holben, B.N. 2010. Enhanced ozone over western North America from biomass burning in Eurasia during April 2008 as seen in surface and profile observations. *Atmospheric Environment*. 44: 4497-4509.
- Ordóñez, C., Brunner, D., Staehelin, J., Hadjinicolaou, P., Pyle, J.A., Jonas, M., Wernli, H., Prevot, A.S.H. 2007. Strong influence of lowermost stratospheric ozone on lower tropospheric background ozone changes over Europe. *Geophysical Research Letters* 34, L07805, doi:10.1029/2006GL029113.
- Oshima, R.J., Braegelmann, P.K., Baldwin, D. W., Van Way, V., Taylor, O.C. 1977a. Reduction of tomato fruit size and yield by ozone. *J. Am. Soc. Hortic. Sci.* 102: 289-293.
- Oshima, R J., Braegelmann, P.K., Baldwin, D.W., Van Way, V., Taylor, O.C. 1977b. Responses of five cultivars of fresh market tomato to ozone: a contrast of cultivar screening with foliar injury and yield. *J. Am. Soc. Hortic. Sci.* 102: 286-289.
- Paoletti, E., Manning, W.J. 2007. Toward a biologically significant and usable standard for ozone that will also protect plants [Review]. *Environ Pollut* 150: 85-95.
<http://dx.doi.org/10.1016/j.envpol.2007.06.037>.
- Patel, H., Talbot, N., Salmond, J., Dirks, K., Xie, S., Davy, P. 2020. Implications for air quality management of changes in air quality during lockdown in Auckland (New Zealand) in response to the 2020 SARS-CoV-2 epidemic. *Science of the Total Env.*
<https://doi.org/10.1016/j.scitotenv.2020.141129>.
- Reed, R.J. 1955. A study of a characteristic type of upper level frontogenesis. *J. Meteorol.* 12, 226-237.
- Reinert, R.A., Nelson, P.V. 1979. Sensitivity and growth of twelve elatior begonia cultivars to ozone. *HortScience* 14: 747-748.
- Rice, J. 2014. Ozone Monitoring Season Analysis. Memorandum to the Ozone NAAQS Review Docket, EPA–HQ–OAR–2008–0699.
- Samet, JM. 2010. Letter from Jonathan Samet, Chair, Clean Air Scientific Advisory Committee, to Administrator Lisa Jackson. Re: CASAC Review of EPA’s Proposed Ozone National Ambient Air Quality Standard (Federal Register, Vol. 75, Nov. 11, January 19, 2010).

February 19, 2010. EPA-CASAC-10-007. Office of the Administrator, Science Advisory Board U.S. EPA HQ, Washington DC. Available at:
<https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockkey=P10072T1.txt>.

Samet, JM. 2011. Letter from Jonathan Samet, Chair, Clean Air Scientific Advisory Committee, to Administrator Lisa Jackson. Re: CASAC Response to Charge Questions on the Reconsideration of the 2008 Ozone National Ambient Air Quality Standards. March 30, 2011. EPA-CASAC-11-004. Available at:
[https://yosemite.epa.gov/sab/sabproduct.nsf/368203f97a15308a852574ba005bbd01/F08BEB48C1139E2A8525785E006909AC/\\$File/EPA-CASAC-11-004-unsigned+.pdf](https://yosemite.epa.gov/sab/sabproduct.nsf/368203f97a15308a852574ba005bbd01/F08BEB48C1139E2A8525785E006909AC/$File/EPA-CASAC-11-004-unsigned+.pdf).

Schelegle, E.S., Walby, W.F., Adams, W.C. 2007. Time course of ozone-induced changes in breathing pattern in healthy exercising humans. *J Appl Physiol*. 102:688–697.

Schelegle, E.S., Morales, C.A., Walby, W.F., Marion, S., Allen, R.P. 2009. 6.6-hour inhalation of ozone concentrations from 60 to 87 ppb in healthy humans. *Am. J. Respir. Crit. Care Med*. 180:265–272.

Schuepbach, E., Davies, T.D., Massacand, A.C. 1999. An unusual springtime ozone episode at high elevation in the Swiss Alps: Contributions both from cross-tropopause exchange and from the boundary layer. *Atmospheric Environment* 33, 1735-1744.

Schultz, M.G. et al. 2017. Tropospheric Ozone Assessment Report: Database and Metrics Data of Global Surface Ozone Observations. *Elementa* 5(58). DOI:
<https://doi.org/10.1525/elementa.244>

Shapiro, M.A. 1980. Turbulent mixing within tropopause folds as a mechanism for the exchange of chemical constituents between the stratosphere and troposphere. *J. Atmos. Sci.* 37, 994-1004.

Sicard, P., Serra, R., Rossello, P. 2016. Spatiotemporal trends in ground-level ozone concentrations and metrics in France over the time period 1999–2012, *Environ. Res.*, 149, 122-144. <http://dx.doi.org/10.1016/j.envres.2016.05.014>.

Sicard, P., De Marco, A., Agathokleous, E., et al. 2020. Amplified ozone pollution in cities during the COVID-19 lockdown, *Science of the Total Environment*,
<https://doi.org/10.1016/j.scitotenv.2020.139542>.

Silverman, F., Folinsbee, L.J., Barnard, J.W., Shephard, R.J. 1976. Pulmonary Function Changes in Ozone -- Interaction of Concentration and Ventilation. *J Appl Physiol* 41/6:859-864.

Simon, H., Reff, A., Wells, B., Xing, J., Frank, N. 2015. Ozone trends across the United States over a period of decreasing NO_x and VOC emissions. *Environ Sci Technol* 49: 186-195.
[dx.doi.org/10.1021/es504514z](https://doi.org/10.1021/es504514z).

- Simon, H., Wells, B., Baker, K.R., Hubbell, B. 2016. Assessing temporal and spatial patterns of observed and predicted ozone in multiple urban areas. *Environmental Health Perspectives* **124** (9): 1443-1452.
- Skelly, J.M. 2000. Tropospheric ozone and its importance to forests and natural plant communities of the northeastern United States. *Northeastern Naturalist*. 7, 221-236.
- Škerlak, B., Sprenger, M., Wernli, H. 2014. A global climatology of stratosphere–troposphere exchange using the ERA-Interim data set from 1979 to 2011. *Atmospheric Chemistry and Physics*, 14, 913–937, doi:10.5194/acp-14-913-2014.
- Škerlak, B., Sprenger, M., Pfahl, S., Tyrlis, E., Wernli, H. 2015. Tropopause folds in ERA-Interim: Global climatology and relation to extreme weather events. *J Geophys Res Atmos* 120: 4860-4877. <http://dx.doi.org/10.1002/2014JD022787>.
- Škerlak, B., Pfahl, S., Sprenger, M., Wernli, H. 2019. A numerical process study on the rapid transport of stratospheric air down to the surface over western North America and the Tibetan Plateau. *Atmospheric Chemistry and Physics*, 19, 6535–6549, <https://doi.org/10.5194/acp-19-6535-2019>.
- Smith, G. 2012. Ambient ozone injury to forest plants in Northeast and North Central USA: 37 years of biomonitoring. *Environ Monit Assess*(184): 4049-4065.
- Smith, G.C., Morin, R.S., McCaskill, G.L. 2012. Ozone injury to forests across the Northeast and North Central United States, 1994-2010. General Technical Report NRS-103. United States Department of Agriculture, US Forest Service, Northern Research Station.
- Sommer, L., Hersher, R., Huo, J., Benincasa, R. 2020. Traffic is way down because of lockdown, but air pollution? Not so much. National Public Radio. <https://www.npr.org/sections/health-shots/2020/05/19/854760999/traffic-is-way-down-due-to-lockdowns-but-air-pollution-not-so-much>.
- Stohl, A., Spichtinger-Rakowsky, N., Bonasoni, P., Feldmann, H., Memmesheimer, M., Scheel, H.E., Trickl, T., Hübener, S. 2000. The influence of stratospheric intrusions on alpine ozone concentrations. *Atmospheric Environment* 34, 1323-1354.
- Tonneijck, A.E.G. 1984. Effects of peroxyacetyl nitrate (PAN) and ozone on some plant species. In: Proceedings of the OECD workshop on ozone; Goteborg, Sweden. Goteborg, Sweden: Swedish Environ. Research Institute.
- Trainer, M., Parrish, D.D., Buhr, M.P., Norton, R.B., Fehsenfeld, F.C., et al. 1993. Correlation of ozone with NO_y in photochemically aged air. *J Geophys Res* 98: 2917-2925.
- Turnipseed, A.A., Burns, S.P., Moore, D.J.P., Hu, J., Guenther, A.B., Monson, R.K. 2009. Controls over ozone deposition to a high elevation subalpine forest. *Agr Forest Meteorol* 149: 1447-1459. <http://dx.doi.org/10.1016/j.agrformet.2009.04.001>.

- Uddling, J., Teclaw, R.M., Pregitzer, K.S., Ellsworth, D.S. 2009. Leaf and canopy conductance in aspen and aspen-birch forests under free-air enrichment of carbon dioxide and ozone. *Tree Physiol* 29: 1367-1380. <http://dx.doi.org/10.1093/treephys/tpp070>.
- U.S. Court of Appeals FOR THE DISTRICT OF COLUMBIA CIRCUIT Argued December 18, 2018 Decided August 23, 2019 No. 15-1385.
- U.S. EPA. 1978. Air quality criteria for ozone and other photochemical oxidants [EPA Report]. (EPA/600/8-78/004). Washington, DC.
<https://nepis.epa.gov/exe/ZyPURL.cgi?Dockey=200089CW.txt>.
- U.S. EPA. 1986. Air quality criteria for ozone and other photochemical oxidants [EPA Report]. (EPA-600/8-84-020aF - EPA-600/8-84-020eF). Research Triangle Park, NC.
<https://ntrl.ntis.gov/NTRL/dashboard/searchResults.xhtml?searchQuery=PB87142949>.
- U.S. EPA. 1992. Summary of Selected New Information on Effects of Ozone on Health and Vegetation: Supplement to 1986 Air Quality Criteria for Ozone and Other Photochemical Oxidants. Report No. EPA/600/8-88/105F. Research Triangle Park, NC: Environmental Protection Agency. Available from: NTIS, Springfield, VA, PB92-235670.
- U.S. EPA. 1996a. Air quality criteria for ozone and related photochemical oxidants [EPA Report]. (EPA/600/P-93/004AF). Research Triangle Park, NC.
- U.S. EPA. 1996b. Review of the national ambient air quality standards for ozone: Policy assessment of scientific and technical information: OAQPS staff paper [EPA Report]. (EPA/452/R-96/007). Research Triangle Park, NC.
https://www.epa.gov/sites/production/files/2020-07/documents/1996_o3sp_final2.pdf.
- U.S. EPA. 2006. Air Quality Criteria for Ozone and Related Photochemical Oxidants (2006 Final). EPA/600/R-05/004aF-cF. Washington, DC: Environmental Protection Agency. Available at: <https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=149923>.
- U.S. EPA. 2007. Review of the national ambient air quality standards for ozone: Policy assessment of scientific and technical information: OAQPS staff paper [EPA Report]. (EPA/452/R-07/007). Research Triangle Park, NC.
https://www3.epa.gov/ttn/naaqs/standards/ozone/data/2007_07_ozone_staff_paper.pdf.
- U.S. EPA. 2013. Integrated Science Assessment of Ozone and Related Photochemical Oxidants (Final Report). EPA/600/R-10/076F. Research Triangle Park, NC: Environmental Protection Agency. Available at: http://www.epa.gov/ttn/naaqs/standards/ozone/s_o3_2008_isa.html.
- U.S. EPA. 2014a. Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards. Final Report. EPA-452/R-14-006. Research Triangle Park, NC: Office of Air Quality Planning and Standards. August.

- U.S. EPA. 2014b. Health Risk and Exposure Assessment for Ozone. Final Report. EPA/452/R-14-004a. Research Triangle Park, NC: Environmental Protection Agency. Available at: <https://www.epa.gov/naaqs/ozone-o3-standards-risk-and-exposure-assessments-current-review>.
- U.S. EPA. 2014c. Welfare Risk and Exposure Assessment for Ozone. Final Report. EPA-452/P-14-005a. Research Triangle Park, NC: Environmental Protection Agency. Available at <https://www.epa.gov/naaqs/ozone-o3-standards-risk-and-exposure-assessments-current-review>.
- U.S. EPA. 2014d. Health Risk and Exposure Assessment for Ozone Final Report Chapters 7-9 Appendices. EPA-452/R-14-004e. Research Triangle Park, NC: Environmental Protection Agency. Available at <https://www.epa.gov/naaqs/ozone-o3-standards-risk-and-exposure-assessments-review-completed-2015>.
- U.S. EPA. 2015. Implementation of the 2015 Primary Ozone NAAQS: Issues Associated with Background Ozone White Paper for Discussion. U.S. Environmental Protection Agency. Research Triangle Park, NC. U.S. EPA. <https://www.epa.gov/sites/production/files/2016-03/documents/whitepaper-bgo3-final.pdf>.
- U.S. EPA. 2018. Technical Assistance Document for the Reporting of Daily Air Quality – the Air Quality Index (AQI). EPA 454/B-18-007. Research Triangle Park, NC: Environmental Protection Agency. Available at <https://www3.epa.gov/airnow/aqi-technical-assistance-document-sept2018.pdf>.
- U.S. EPA, 2019c: A Look Back: Ozone in 2018. <https://epa.maps.arcgis.com/apps/Cascade/index.html?appid=9bec4031ba6f4887a9f332a8f058b198>
- U.S. EPA. 2020a. Integrated Science Assessment of Ozone and Related Photochemical Oxidants. EPA/600/R-20/012. April. Research Triangle Park, NC: Environmental Protection Agency. April. Available at: <https://www.epa.gov/naaqs/ozone-o3-standards-integrated-science-assessments-current-review>.
- U.S. EPA. 2020b. Policy Assessment for the Review of the Ozone National Ambient Air Quality Standards. EPA-452/R-20-001. May. Available at <https://www.epa.gov/naaqs/ozone-o3-standards-policy-assessments-current-review>.
- U.S. EPA. 2022. Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards (External Review Draft). EPA-452/D-22-002. Research Triangle Park, NC: Office of Air Quality Planning and Standards. April. Available at <https://www.epa.gov/naaqs/ozone-o3-standards-policy-assessments-current-review>.
- U.S. EPA. 2023. Policy Assessment (PA) for the Reconsideration of the Ozone National Ambient Air Quality Standards, External Review Draft Version 2. EPA-452/P-23-002.

- Research Triangle Park, NC: Office of Air Quality Planning and Standards. March. Available at <https://www.epa.gov/naaqs/ozone-o3-standards-policy-assessments-current-review>.
- Wang, X., Zheng, O., Feng, Z., Xie, J., Feng, Z., Ouyang, Z., Manning, W.J. 2008. Comparison of a diurnal vs steady-state ozone exposure profile on growth and yield of oilseed rape (*Brassica napus* L.) in open-top chambers in the Yangtze Delta, China. *Environmental Pollution* 156 (2008) 449-453.
- Wang, L., Pang, J., Feng, Z., Zhu, J., Kazuhiko, K. 2015. Diurnal variation of apoplastic ascorbate in winter wheat leaves in relation to ozone detoxification. *Environ Pollut* **207**: 413-419.
- Wang, M., Aaron, C.P., Madrigano, J., et al. 2019. Association between long-term exposure to ambient air pollution and change in quantitatively assessed emphysema and lung function. *JAMA*. 2019;322(6):546-556. doi:10.1001/jama.2019.10255.
- Wang, T., Xue, L., Feng, Z., Dai, J., Zhang, Y., Tan, Y. 2022. Ground-level ozone pollution in China: a synthesis of recent findings on influencing factors and impacts. *Environ. Res. Lett.* 17 (2022) 063003. <https://doi.org/10.1088/1748-9326/ac69fe>.
- Wells, B. 2015. Memorandum to Ozone NAAQS Review Docket (EPA-HQ-OAR-2008-0699). Expanded Comparison of Ozone Metrics Considered in the Current NAAQS Review. September 28, 2015. Docket Document Identifier EPA-HQ-OAR-2008-0699-0163. Available at: <https://www.regulations.gov/contentStreamer?documentId=EPA-HQ-OAR-2008-0699-4325&contentType=pdf>.
- Wells, B. 2020. Memorandum to Ozone NAAQS Review Docket (EPA-HQ-OAR-2018-0279). Additional Analyses of Ozone Metrics Related to Consideration of the Ozone Secondary Standard. December 2020. Docket Document Identifier EPA-HQ-OAR-2018-0279-0557.
- Wells, B., Dolwick, P., Eder, B., Evangelista, M., Foley, K., Mannshardt, E., Misenis, C., Weishampel, A. 2021. Improved estimation of trends in U.S. ozone concentrations adjusted for interannual variability in meteorological conditions. *Atmospheric Environment*. 248 (2021) 118234. <https://doi.org/10.1016/j.atmosenv.2021.118234>.
- Wernli, H., Davies, H.C. 1997. A Lagrangian-based analysis of extratropical cyclones. I: The method and some applications. *Quart. J. Roy. Meteor. Soc.* 123, 467-489.
- Wernli, H., Bourqui, M. 2002. A Lagrangian “one-year climatology” of (deep) cross-tropopause exchange in the extratropical northern hemisphere. *J. Geophys. Res.* 107(D2), 4021, doi:10.1029/2001JD000812.
- Wu, R., Agathokleous, E., Feng, Z. 2021. Novel ozone flux metrics incorporating the detoxification process in the apoplast: An application to Chinese winter wheat. *Science of the Total Environment* 767 (2021) 144588.

- Xu, X., et al. 2020. Long-term changes of regional ozone in China: implications for human health and ecosystem impacts. *Elem Sci Anth*, 8: 13. DOI: <https://doi.org/10.1525/elementa.409>.
- Xue, L., Wang, T., Louie, P.K.K., Luk, C.W.Y., Blake, D.R., Xu, Z. 2014. Increasing External Effects Negate Local Efforts to Control Ozone Air Pollution: A Case Study of Hong Kong and Implications for Other Chinese Cities, *Environ. Sci. Technol.*, 48, 10769–10775, [dx.doi.org/10.1021/es503278g](https://doi.org/10.1021/es503278g).
- Yun, S-C, Laurence, J.A. 1999. The response of sensitive and tolerant clones of *Populus tremuloides* to dynamic ozone exposure under controlled environmental conditions. *New Phytologist* 143, 305–313.
- Zanis, P., Akritidis, D., Turnock, S., Naik, V., Szopa, S., Georgoulas, A.K., Bauer, S.E., Deushi, M., Horowitz, L.W., Keeble, J., Le Sager, P., O'Connor, F.M., Oshima, N., Tsigaridis, K., van Noije, T. 2022. Climate change penalty and benefit on surface ozone: a global perspective based on CMIP6 earth system models. *Environ. Res. Lett.* 17 024014. <https://doi.org/10.1088/1748-9326/ac4a34>.
- Zhang, L, Jacob, D.J., Downey, N.V., Wood, D.A., Blewitt, D., Carouge, C.C., van Donkelaar, A., Jones, D.B.A., Murray, L.T., Wang, Y. 2011. Improved estimate of the policy-relevant background ozone in the United States using the GEOS-Chem global model with $1/2^\circ \times 2/3^\circ$ horizontal resolution over North America. *Atmospheric Environment* 45, 6769-6776.