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

Docket ID No. EPA-HQ-OAR-2018-0279

Allen S. Lefohn, Ph.D.
A.S.L. & Associates, LLC
302 North Last Chance Gulch
Suite 410
Helena, Montana 59601

May 30, 2022

**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 Draft Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards. His comments are submitted into the Ozone Docket (No. EPA-HQ-OAR-2018-0279) for the purpose of providing scientific clarification for the ozone ($O_3$) rulemaking activity. Previously, Dr. Lefohn has provided comments in the Ozone Docket on 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).

## About the Author

**Dr. Allen S. Lefohn** is President and Founder of A.S.L. & Associates, LLC (http://www.asl-associates.com/) in 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 (2000-2022). 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 extremely important in better understanding the variability in $O_3$ 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 $O_3$ on vegetation and human health, (2) investigating biological mechanisms that define the nonlinearity (i.e.,

weighting of the higher concentrations more than the mid- and low-level values) response to $O_3$ for both human health and vegetation, (3) understanding the importance of background $O_3$ in relation to ambient concentrations, and (4) integrating results from the EPA's air quality database for (a) characterizing co-occurrence patterns of criteria air pollutants under ambient conditions, (b) characterizing $O_3$ 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 that are important for the $O_3$ 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 $O_3$ rulemaking for the 70 ppb $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ NAAQS. Dr. Lefohn presented testimony in March 2015 to the House Committee on Science, Space, and Technology about background $O_3$. In 2015, Dr. Lefohn was a co-guest editor for the *Atmospheric Environment* special issue: Observations and source attribution of $O_3$ 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.elementascience.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 46 years and continues to perform his worldwide research from this location.

## Executive Summary

1. Two key fundamental principles are important in the $O_3$ rulemaking activity. The first fundamental principle is **Higher Hourly Average $O_3$ 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 $O_3$ peak responses were observed in stepwise and triangular (smooth increases and decreases in concentration) exposures 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) $O_3$ effects in plants are cumulative; (2) higher $O_3$ concentrations appear to be more important than lower concentrations in eliciting a response; (3) plant sensitivity to $O_3$ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the $O_3$ 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 $O_3$ concentrations in the Los Angeles area occurred, provide independent confirmation of the experimental studies for the greater importance of the higher hourly average $O_3$ 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 $O_3$. (Section 2).

4. In implementing its Air Quality Index (AQI) reported across the U.S., EPA recognizes the importance of the higher $O_3$ concentration. Ozone pollutant specific sensitive groups are separated by 8-h daily maximum $O_3$ concentrations as shown below in Fig. ES-1. The higher the $O_3$ 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 notes (Federal Register, 2020 – 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 $O_3$ and that it could effectively limit health effects attributable to both short- and long-term $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ standards. (Section 5).

6. The second key fundamental principle is that **Daily Maximum Hourly Averaged $O_3$ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide**. As $O_3$ precursor emissions are reduced at specific locations across the U.S. to attain the current $O_3$ NAAQS, not all hourly average $O_3$ concentrations at these locations shift downward. In fact, at many locations, the lower hourly average $O_3$ concentrations shift upward as described in the next two items. 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).

7. As emissions are reduced to meet the $O_3$ NAAQS, a compression of the higher and lower hourly average $O_3$ 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 $O_3$ concentrations (Fig. ES-2). The site in 1985 appeared to be influenced by NO titration of $O_3$ 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 $O_3$ 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 $O_3$ by NO as reduction in $NO_x$ emissions occur (Lefohn et al., 1998; EPA, 2014b; Simon et al., 2015; Lefohn et al., 2017, 2018). The reduction of $O_3$ 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).

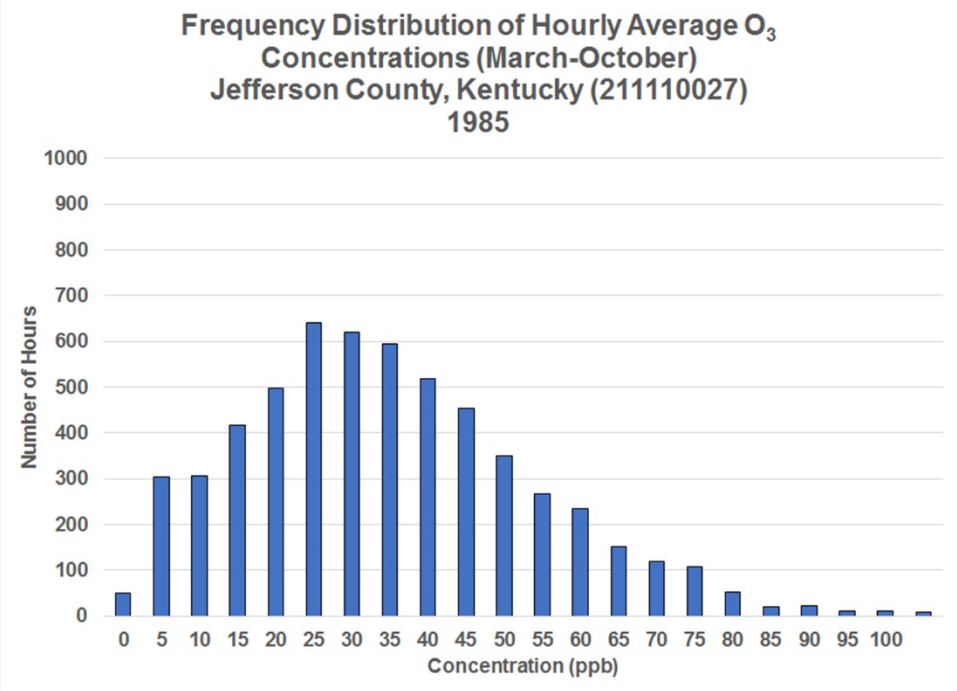



**Figure ES-2. Frequency distribution of the hourly average $O_3$ concentrations in 1985 for an urban-influenced site in Jefferson County (KY) (AQS ID 211110027) monitoring station. Source of data is from the EPA's AQS database.**

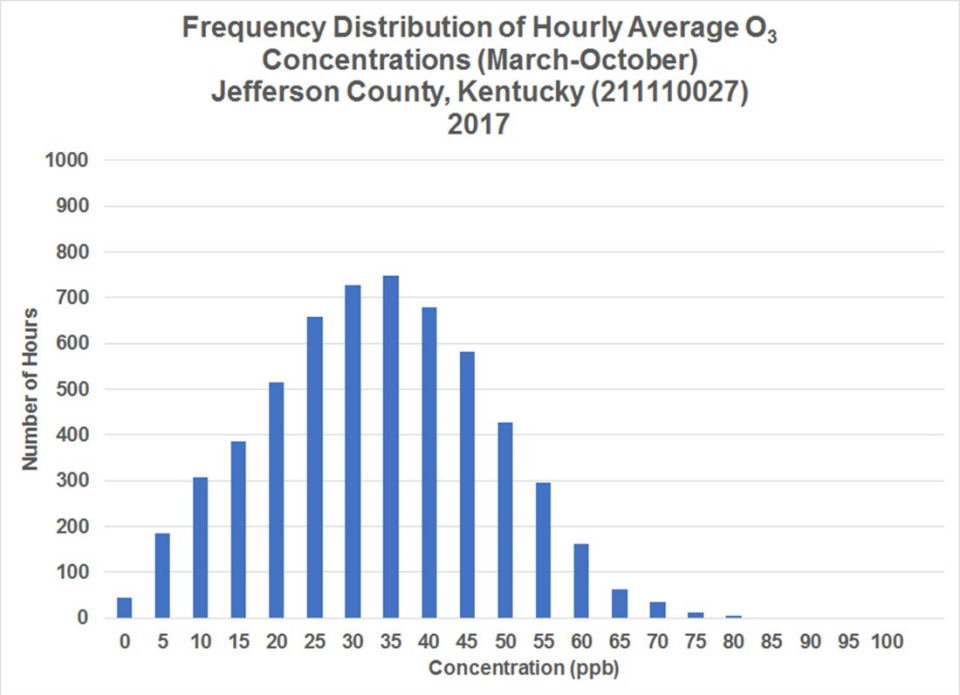



**Figure ES-3. Frequency distribution of the hourly average $O_3$ concentrations in 2017 for an urban-influenced site in Jefferson County (KY) (AQS ID 211110027) monitoring station. Source of data is from the EPA's AQS database.**

8. 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 $O_3$ concentrations as emissions were reduced. During the 2020 COVID-19 lockdown, scientists characterized these changes by applying different $O_3$ 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* $O_3$ concentrations during lockdown; those who applied metrics focused on the lower end of the distribution reported *increasing* $O_3$ concentrations. The lockdown that occurred during the spring in 2020 in the U.S. resulted in large emission reductions of $O_3$ 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. $O_3$ 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 $O_3$ monitoring sites in the U.S. As noted, some of the researchers who

characterized $O_3$ monitoring sites around the world during lockdown applied metrics focused on the lower end of the distribution. Some of these investigators reported that increasing $O_3$ concentrations appeared to be related to the reduction of NO titration of $O_3$. The result of this "natural experiment" provides additional evidence that emission reductions result in increases in the less biologically important hourly average concentrations. The results of the "natural experiment" during the COVID-19 lockdown support the second fundamental principle that **Daily Maximum Hourly Averaged $O_3$ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide**. (Section 3).

9. The compression of the distribution of hourly average $O_3$ concentrations results in *annual average or median concentration* values ***increasing*** at some sites. For example, in Fig. ES-4, the annual 4$^{th}$ highest daily maximum 8-h concentration for 2000-2018 illustrates the trend patterns in $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ levels (Lefohn et al., 2001, 2011, 2012, 2014). When nonparametric statistics are applied, no trend has been observed at Yellowstone National Park using the annual 4$^{th}$ highest daily maximum 8-h average concentration metric. (Section 3).

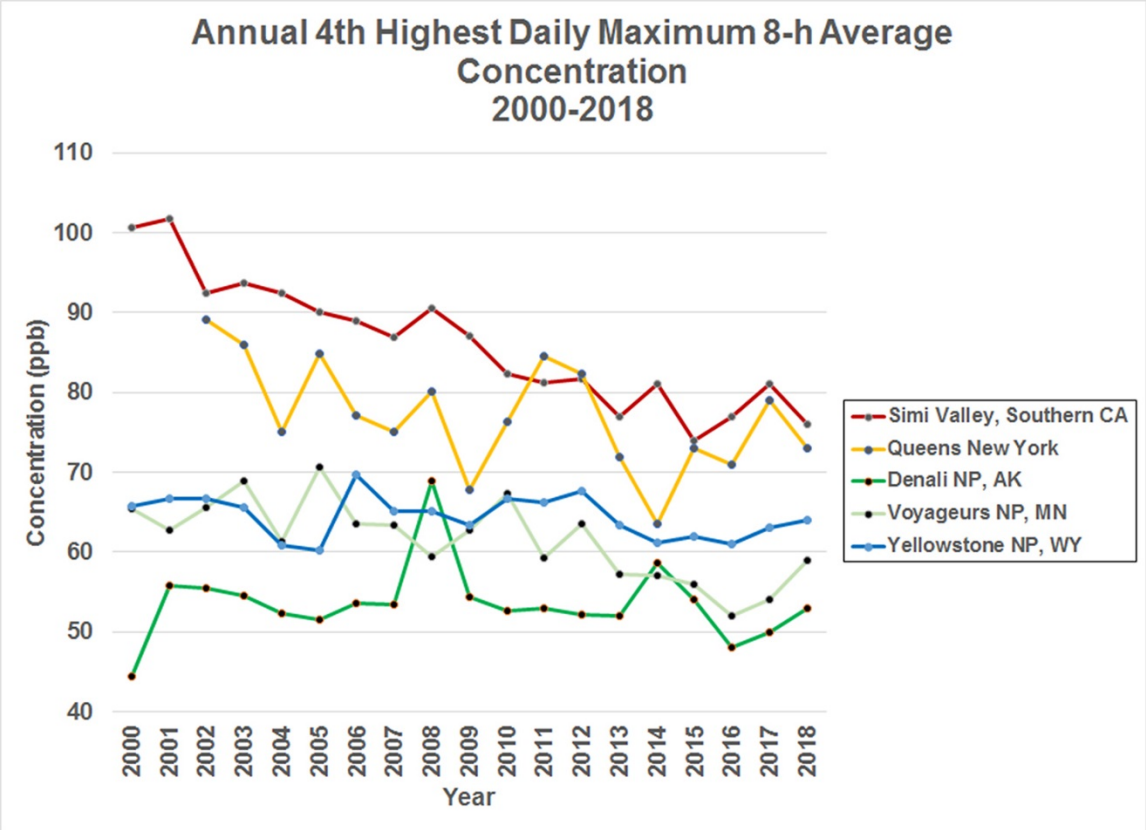



**Figure ES-4. The annual 4th highest daily maximum 8-h average $O_3$ concentration for the period 2000-2018 for Simi Valley, CA (061112002), Queens New York, NY (360810124, Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011). Source of data is from the EPA's AQS database.**

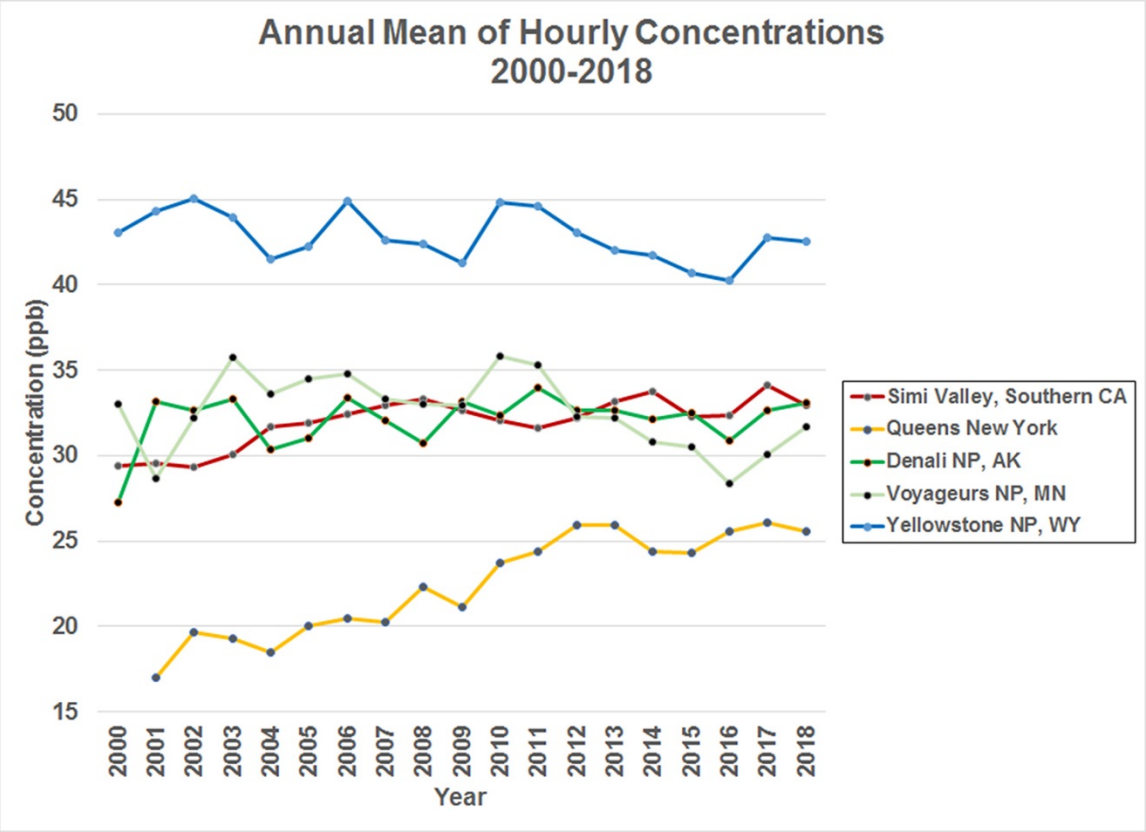



**Figure ES-5. The annual mean of the hourly average $O_3$ concentrations for the period 2000-2018 for Simi Valley, CA (061112002), Queens New York, NY (360810124, Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011). Source of data is from the EPA's AQS database.**

10. One does not have to select necessarily an annual average or seasonal average concentration metric to estimate "chronic" (i.e., long-term) human health effects from $O_3$ exposures. Based on empirical data, annual average concentrations increase as emissions are reduced. The biologically important higher hourly average $O_3$ concentrations are reduced as emissions are reduced. In its review of the 2015 $O_3$ 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 $O_3$ and that it could effectively limit health effects attributable to both short- and long-term $O_3$ exposures. (Section 5).

11. 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). Samet (2010), in summarizing CASAC's comments on the reconsideration of the $O_3$ 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 $O_3$ on human health and the effects of chronic $O_3$ exposures on

welfare, namely that vegetation effects were more dependent on the cumulative exposure to, and uptake of, $O_3$ 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 $O_3$ 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. In August 2019, the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) addressed arguments regarding the adequacy of the EPA Administrator's 2015 decision to use the current form of the $O_3$ human health standard instead of the W126 metric as the secondary $O_3$ NAAQS. One aspect of the Court's August 2019 decision was to question EPA's decision to use a 3-year average of the W126 index rather than a 1-year annual W126 value to protect vegetation. The Court ruled that it lacked any basis to assess the reasonableness of EPA's actions to use the current form of the $O_3$ NAAQS instead of the W126 because EPA never explained why it was reasonable to focus on a 3-year average of the W126 index instead of an annual W126 value. In the 2020 Review of the Ozone National Ambient Air Quality Standards document (Federal Register, 2020), 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." In its 2020 decision, 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 $O_3$ 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 $O_3$ average concentrations $\geq$ 100 ppb. For this additional aspect, the draft PA (EPA, 2022, 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, an important discussion is provided about the comparison between the elevated $O_3$ concentrations (i.e., hourly average values $\geq$ 100 ppb) observed across the U.S. today and the $O_3$ 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 $O_3$ NAAQS for the prevention of "unusually damaging years." (Section 4).

12. As discussed in the draft PA (EPA, 2022), Lefohn et al. (1997) documented the frequency of occurrence of elevated $O_3$ concentrations (i.e., number of hourly $O_3$ average concentrations $\geq$ 100 ppb - referred to as N100) in the tree seedling experiments for some of the 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 $O_3$ 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 $O_3$ 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. Lee et al. (2022) classified 16 tree species (11 of these species are described in the draft PA) according to their sensitivity, based on biomass loss response functions to protect from a 5% biomass loss. The seedling $O_3$ 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 exposures. 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 $O_3$ exposures, especially for the sensitive species. The draft PA (EPA, 2022) does not include the results of Lee et al. (2022) because the authors' paper was not published at the time the draft PA was prepared. The Lee et al. (2022) study is a potentially important contribution that supplements the current information contained within the draft PA on the appropriate form and level of the secondary $O_3$ NAAQS. It is recommended that EPA include the authors' findings when the Agency revises the current draft PA. 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. As discussed in Section 4 in these comments, the weighting scheme for the W126 metric includes all hourly average $O_3$ concentrations, including hourly average $O_3$ concentrations ≥ 100 ppb. It is not a necessary requirement that hourly average $O_3$ concentrations ≥ 100 ppb must be present for vegetation damage to occur. The draft PA (EPA, 2022) notes that of the 1105 monitoring sites with valid design values for 2018-2020, almost 74% do not experience N100 values for the 2018-2020 period. As reported in the literature, not all the treatments in the tree seedling and crop experiments experienced elevated hourly averaged $O_3$ concentrations at and above 100 ppb. At the more moderate experimental treatments, the exposure regimes contained infrequent occurrences of hourly average $O_3$ concentrations ≥ 100 ppb (see for example, Table 4A-6 in the draft PA (EPA, 2022, pages 4A-17 – 4A-21). The EPA appears to be

focused on in the PA more on the exposure regimes experienced in the highest treatments in the tree seedling and crop experiments than on the more moderate exposure regimes occurring in the lower treatments. The more sensitive species experienced biomass and yield losses under exposures experiencing infrequent occurrences of hourly average $O_3$ concentrations ≥ 100 ppb. As has been mentioned previously, 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 draft PA (EPA, 2022, page 4D-10) comments on the variation of the annual W126 index from the 3-year (2018-2020) W126 index. Figs. 4D-6 and 4D-7 in the draft PA (EPA, 2022) illustrate the variation of the annual W126 values from the 2018-2020 average W126 metric values. Both figures show that at the 7 ppm-hrs level for the annual W126 index, the deviation can be large relative to the absolute value of the W126 index. In addition to the deviation analyses used by the EPA for supporting the use of the 3-year versus the annual W126 index, the draft PA (EPA, 2022, page 4-52) refers to 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 $O_3$-related biomass impacts estimated using the established E-R functions for aspen. The draft PA notes that the EPA's analyses of the King et al. (2005) study did 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 multi-year average. The draft PA (EPA, 2022, page 4A-26) notes that datasets of tree growth that investigate the impact of $O_3$ across multiple-year periods, such as that available for aspen in the King et al. (2005) study, are not prevalent. Based on the paucity of data sets, it is suggested that caution be used in drawing a broad interpretation of EPA's use of the King et al. (2005) data. It may be premature to use the conclusions derived from the EPA's analyses of data from the King et al. (2005) study to support the Agency's decision to use a 3-year average of the W126 index. Thus, based on the evidence associated with the deviation analyses and the paucity of datasets available to analyze data similar to those published by King et al. (2005), strong evidence does not appear to be provided in the draft PA (EPA, 2022) to support the use of the 3-year average of the W126 metric instead of the use of an annual W126 value. (Section 4).

13. In the 2020 decision (Federal Register, 2020), the EPA (1) preferred to use the W126, as well as peak hourly concentrations of interest, for assessing vegetation risk associated with $O_3$ exposures and (2) apply the current 8-h form of the $O_3$ NAAQS to control for those W126 values and peak hourly concentrations associated with vegetation effects. As noted in Item 12 above, Lefohn et al. (1997) identified at the 10% level, for the most sensitive tree seedlings in the Southern Appalachian area, a 92-day 24-h W126 value of 5.9 ppm-hrs coupled with an N100 value of 6. The Forest County Potawatomi Community established the $O_3$ thresholds for vegetation to be a 3-year average of the 3-month (June, July, and August) 24-h cumulative W126 value of 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. Lee et al. (2022) reported for the most sensitive species in their analysis for a biomass loss of 5% a 92-day 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. 4-6 (Fig. 4-9 in EPA

(2022, page 4-63)), the right side of the figure shows that the range of values for the current form of the $O_3$ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the $O_3$ NAAQS of 0.070 ppm (i.e., 70 ppb), numerous occurrences greater than 7 ppm-hrs are evident and these occurrences are not geographically isolated to the western part of the US. 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 $O_3$ NAAQS. The W126 metric and the 8-h $O_3$ NAAQS each behave differently at levels of 7 ppm-hrs and below. Focusing on the N100 values, the draft PA 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. At a level of the 3-year $4^{th}$ Max $\leq$ 70 ppb, 9% of the sites experience N100 values > 0 occurrences. In summary, at a W126 level of 7 ppm-hrs (1) there does not appear to be a strong relationship between the annual W126 and the current form of the 3-year 8-h $O_3$ NAAQS and (2) the use of the current 8-h form of the 3-year $4^{th}$ Max $\leq$ 70 ppb to control for the N100 values does not appear to be any more efficient than applying the W126 exposure index to control for the N100 values. Thus, the use of the $O_3$ NAAQS to limit annual W126 values to 7 ppm-hrs (and below) and N100 values to infrequent occurrences for sensitive tree species that are ecologically important and widespread across the U.S. appears to be questionable. The use of a more precise instrument (i.e., the annual W126 index) rather than the use of a blunt tool (i.e., the current 8-h form of the 3-year $4^{th}$ Max standard) is required to protect vegetation across the U.S. (Section 4).

14. The Court of Appeals for the District of Columbia Circuit (DC Circuit) in its August 19, 2019, decision addressed arguments regarding considerations of background $O_3$ concentrations, and socioeconomic and energy impacts. Regarding background $O_3$, the Court rejected the argument that the EPA was required to take background $O_3$ 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 $O_3$ levels in any part of the country exceeded the level of $O_3$ 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 $O_3$. (Section 3). While it is believed that background $O_3$ currently is not a consideration in the setting of the level of the $O_3$ standard, background $O_3$ 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 $O_3$ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total ambient $O_3$ levels and potentially can influence those human health risk assessments associated with the margin of safety determinations for the setting of the primary $O_3$ NAAQS. (Section 3).

15. 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 $O_3$ exposures occur in the spring independent of emission reductions. These observations have important ramifications for assessing the validity of background $O_3$ modeling estimates. (Section 3).

16. 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 $O_3$ 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 $O_3$ modeling estimates. (Section 3).

17. In the ISA (EPA, 2020a, page 1-53), EPA states that background $O_3$ 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 $O_3$ 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 $O_3$ concentrations, which contribute to the estimated human health risk assessment. (Section 3).

18. The authors note that the term **US background (USB)** is used to assess background $O_3$. Background $O_3$ 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 draft PA. 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 $O_3$*** (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 $O_3$ modeling results presented in the draft PA (EPA, 2022). (Section 3).

19. 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 $O_3$ concentrations can be comparable to or greater than spring concentrations. The ISA did not resolve the conflicting conclusions about when seasonal mean background $O_3$ 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 $O_3$

exposures occur in the spring independent of emission reductions. At some National Park Service (NPS) sites, the highest $O_3$ 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 $O_3$ 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 $O_3$ monitoring seasons in 32 states and the District of Columbia was appropriate. The Agency indicated that ambient $O_3$ 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 $O_3$ 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 $O_3$ exposures. The pattern when the highest exposures occur is important for validating model performance estimating background $O_3$ concentrations. There continues to be strong evidence, as supported in the literature, that background $O_3$ across the U.S. is highest at many sites during the springtime (including into the month of June) and background $O_3$ is an important contributor at many high-elevation sites throughout the year. (Section 3).

20. The USB modeling results described in the draft PA (EPA, 2022) 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 draft PA (EPA, 2022, page 2-64) indicates that natural and U.S. anthropogenic $O_3$ contributions peak during the traditional $O_3$ season (May through September), while long-range intercontinental transport of international $O_3$ (i.e., contributions from China, India, etc.) peaks in the spring (February through May). Reviewing Fig. ES-6, the total of the natural and international components shows for the West that March-August appears to be the period of highest total background $O_3$; 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 $O_3$ 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 draft PA (EPA, 2022) 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 draft PA. The draft PA (EPA, 2022) noted that bias adjustment was not performed in the modeling described in the 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 $O_3$ 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 draft PA (EPA, 2022, page 2-66), the background $O_3$ 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 $O_3$ may explain some of the discrepancy between the most current EPA background $O_3$ modeling results described in the draft PA (EPA, 2022) and those reported in the 2014 PA (EPA, 2014a).

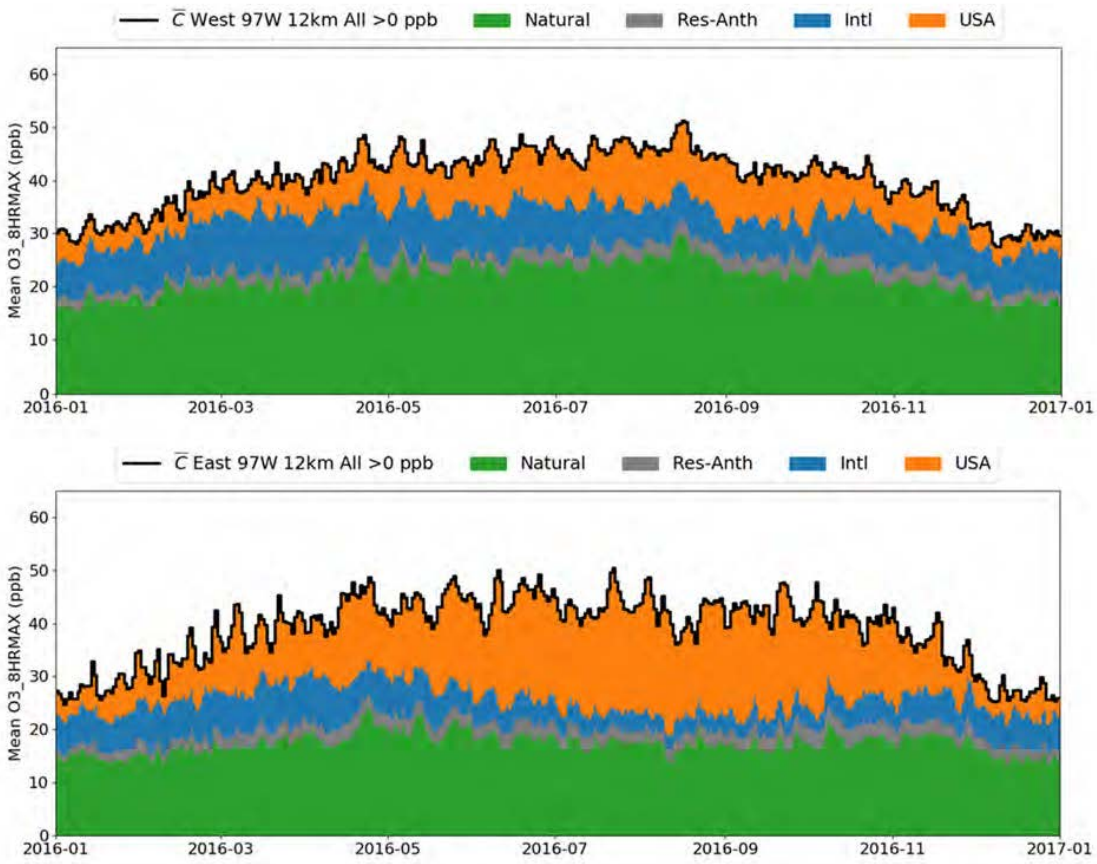



**Figure ES-6. Annual time series of regional average predicted MDA8 total $O_3$ concentration and contributions of each source (see legend) for the West (top), and the East (bottom). Natural is global natural sources, Intl is international anthropogenic sources, USA is U.S. anthropogenic sources, and Res-Anth is the residual anthropogenic. (Source: EPA, 2020b Fig. 2-23, page 2-49).**

21. Empirical data indicate that as emission reductions occur across the U.S., the higher MDA8 concentrations shift at many $O_3$ 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 $O_3$ concentrations. In addition, as emission reductions

occur, background $O_3$ concentrations increase their percentage in the observed total $O_3$ 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 $O_3$. The patterns derived from empirical data showing the (1) compression of the distribution of hourly average $O_3$ 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 $O_3$ levels. Background $O_3$ is an important component of the margin of safety determinations. (Section 3).

22. 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 $O_3$ 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 4$^{th}$ highest daily maximum 8-h concentration exposure metric, shows that some sites have not experienced increasing trends over the period 2000-2014. (Section 3).

23. Depending upon the specific monitoring sites, background $O_3$ contributes varying amounts to the higher hourly average $O_3$ concentrations. For example, the high-elevation Yellowstone National Park site in Wyoming is dominated by background $O_3$ throughout the year with minor anthropogenic contributions (Lefohn et al., 2014). In Fig. ES-7 below, the relative comparison of background $O_3$ levels (noted by blue) to anthropogenic (noted by red) within each concentration level shows that background contributes greater than 80% across all ranges of concentrations. In comparison, Fig. ES-8 illustrates that for the Los Angeles area, a site heavily influenced by anthropogenic emissions, background $O_3$ contributes less than 40% at the higher hourly average concentrations. (Section 3).

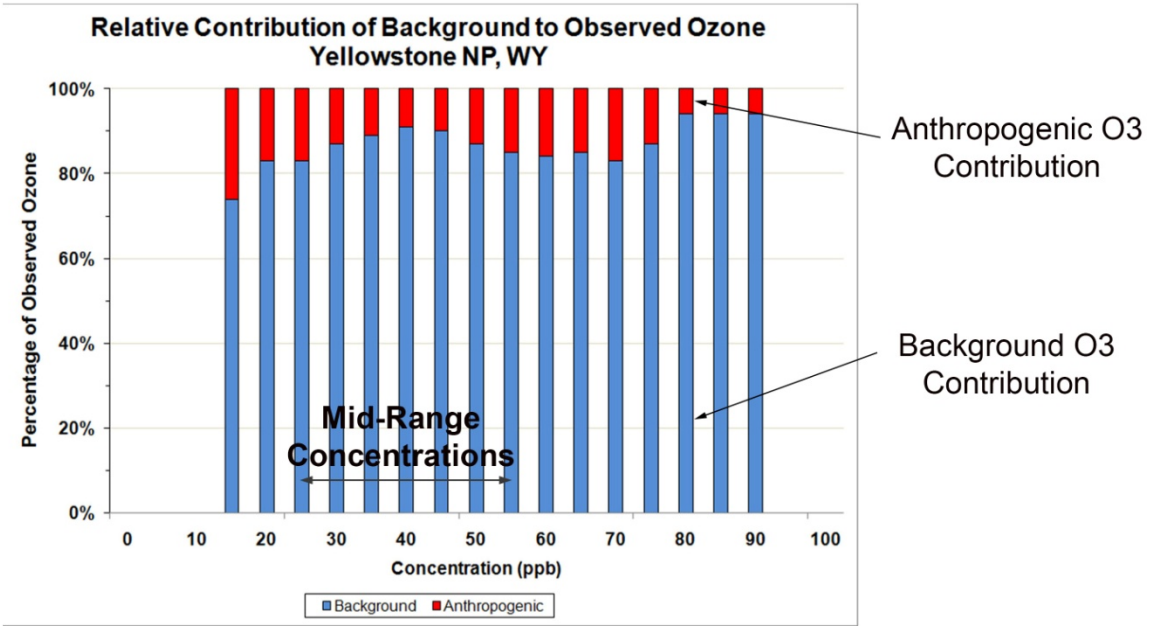


**Figure ES-7. Average relative contributions of current hourly background (blue) and anthropogenic $O_3$ (red) for Yellowstone NP (WY) (AQS ID 560391011) in 2006. (Source: Lefohn et al., 2014).**

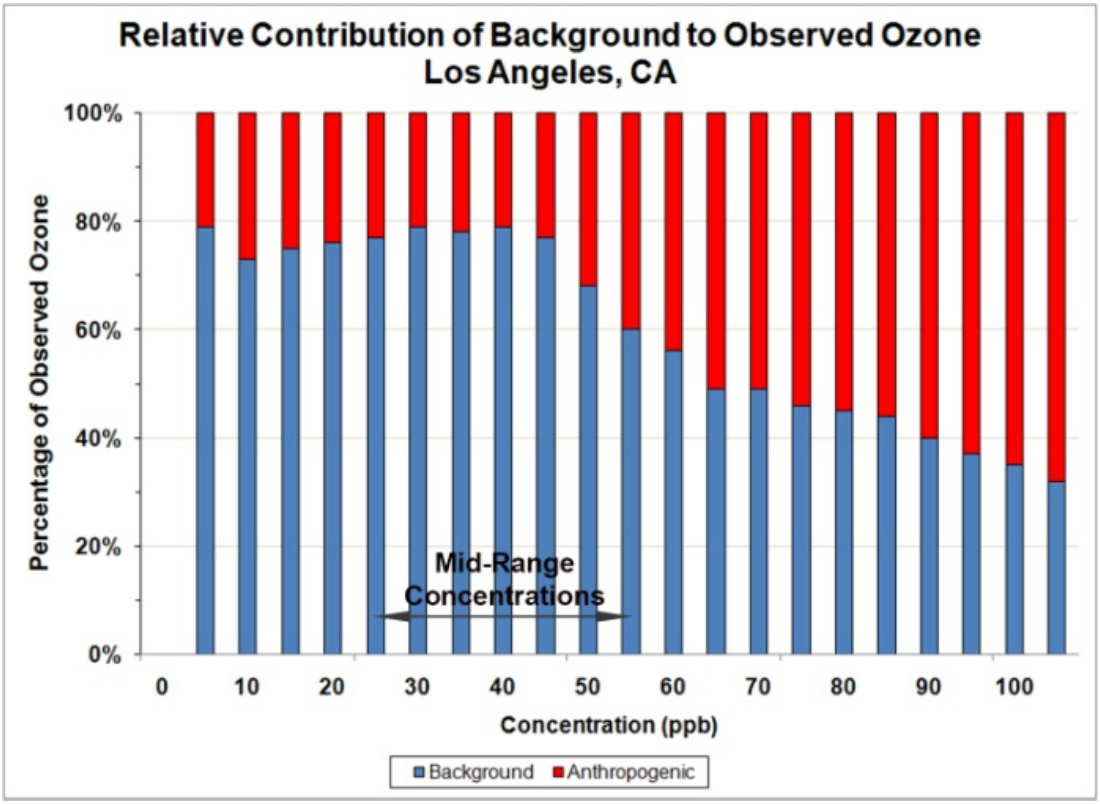



**Figure ES-8. Average relative contributions of current hourly background (blue) and anthropogenic $O_3$ (red) for the Los Angeles (CA) (AQS ID 060719004) area in 2006. (Source: Lefohn et al., 2014).**

24. In the draft PA (EPA, 2022), 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 $O_3$ 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 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 draft PA (EPA, 2022) risk assessment. In Figs. ES-9 through ES-15, total observed $O_3$ 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 draft PA. In the figures, gaps (i.e., the difference between the observed total $O_3$ (noted by the black line) and $USB_{AB}$

concentrations (noted by the green line)) occur, indicating the apparent influence of anthropogenic sources. (Section 3).

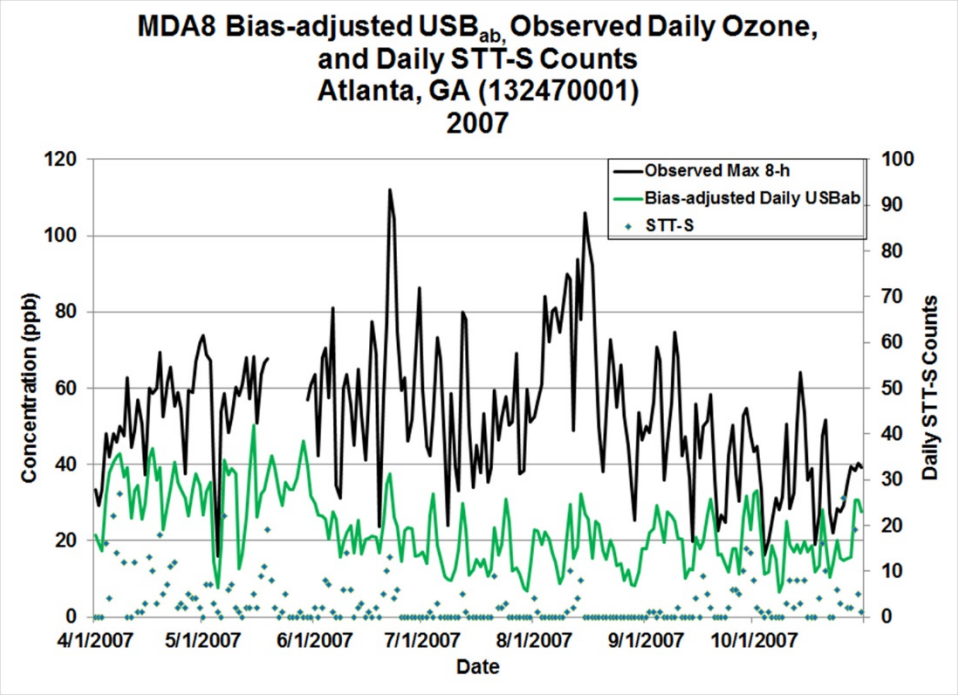



**Figure ES-9. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Atlanta, Georgia (AQS ID 132470001) for April-October 2007. The daily stratospheric-tropospheric transport to surface (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 how the STT-S values are estimated.**

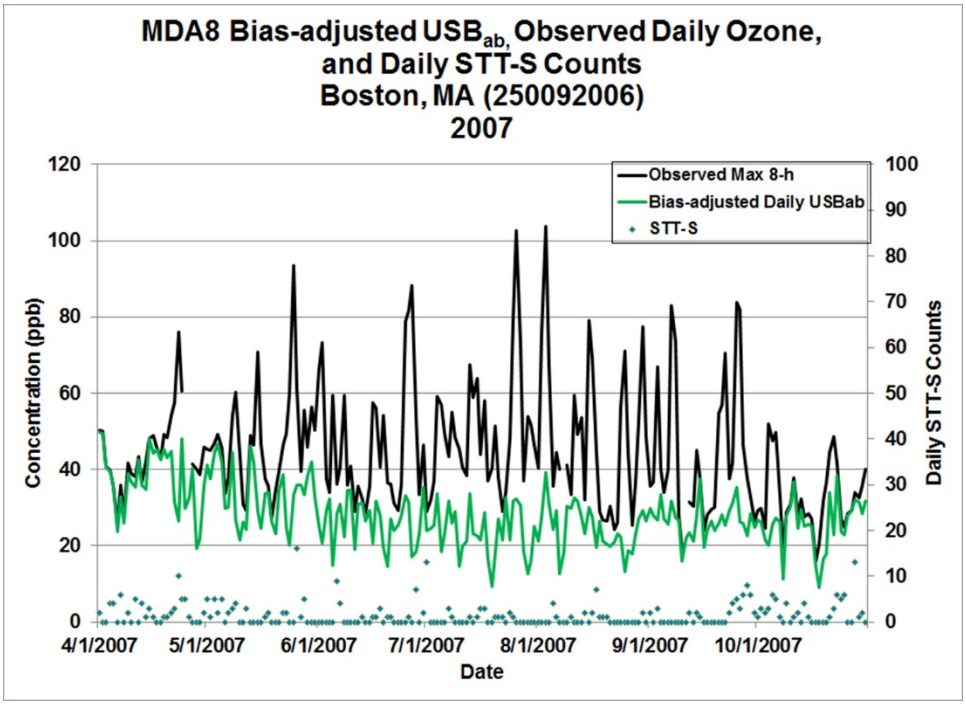



**Figure ES-10. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Boston, Massachusetts (AQS ID 250092006) for April-October 2007. The daily stratospheric-tropospheric transport to 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 how the STT-S values are estimated.**

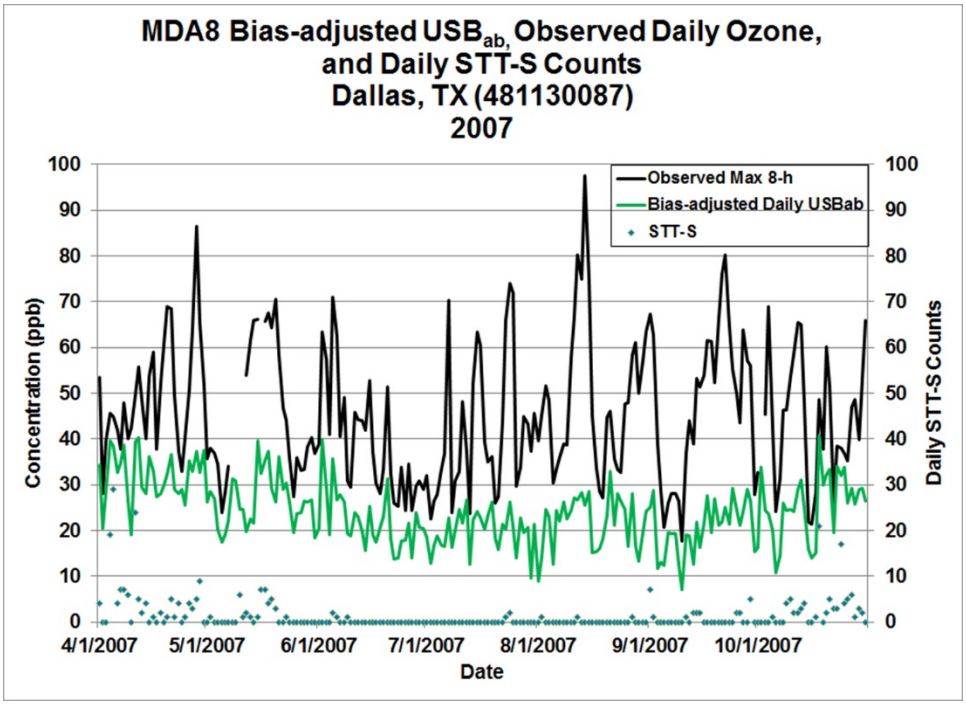



**Figure ES-11. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Dallas, Texas (AQS ID 481130087) for April-October 2007. The daily stratospheric-tropospheric transport to surface (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 how the STT-S values are estimated.**

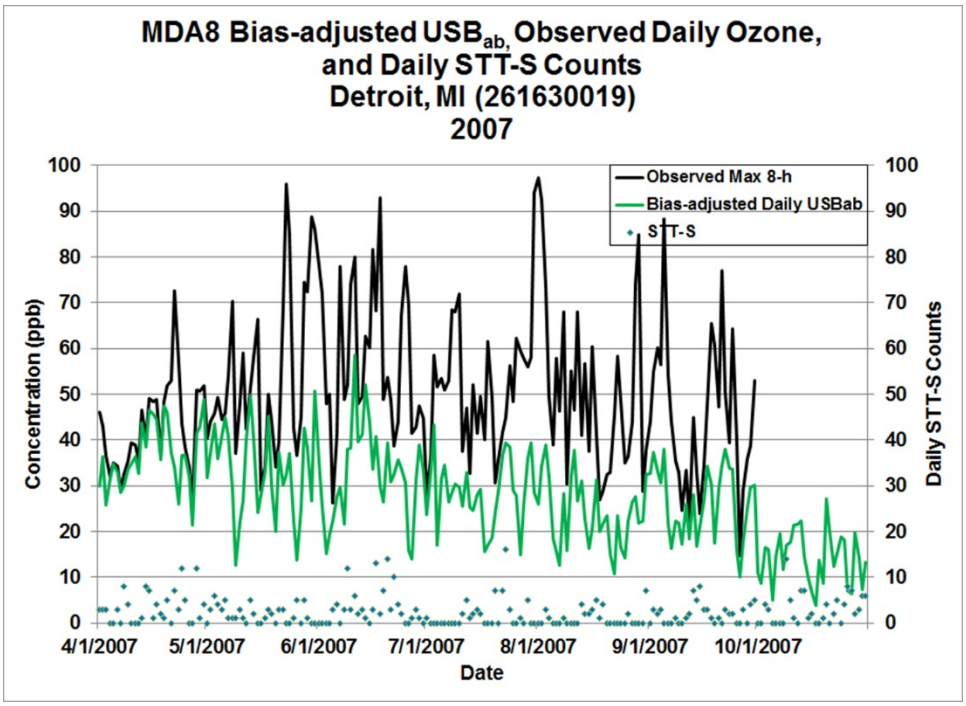



**Figure ES-12. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Detroit, Michigan (AQS ID 261630019) for April-October 2007. The daily stratospheric-tropospheric transport to surface (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 how the STT-S values are estimated.**

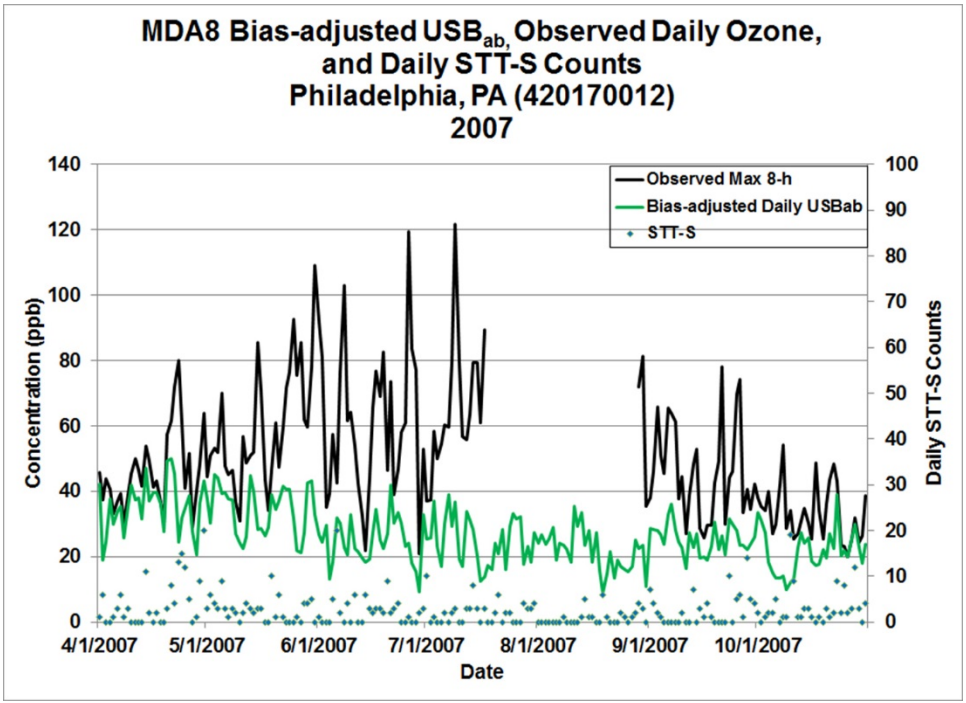



**Figure ES-13. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Philadelphia, Pennsylvania (AQS ID 420170012) for April-October 2007. The daily stratospheric-tropospheric transport 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 $O_3$ 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.**

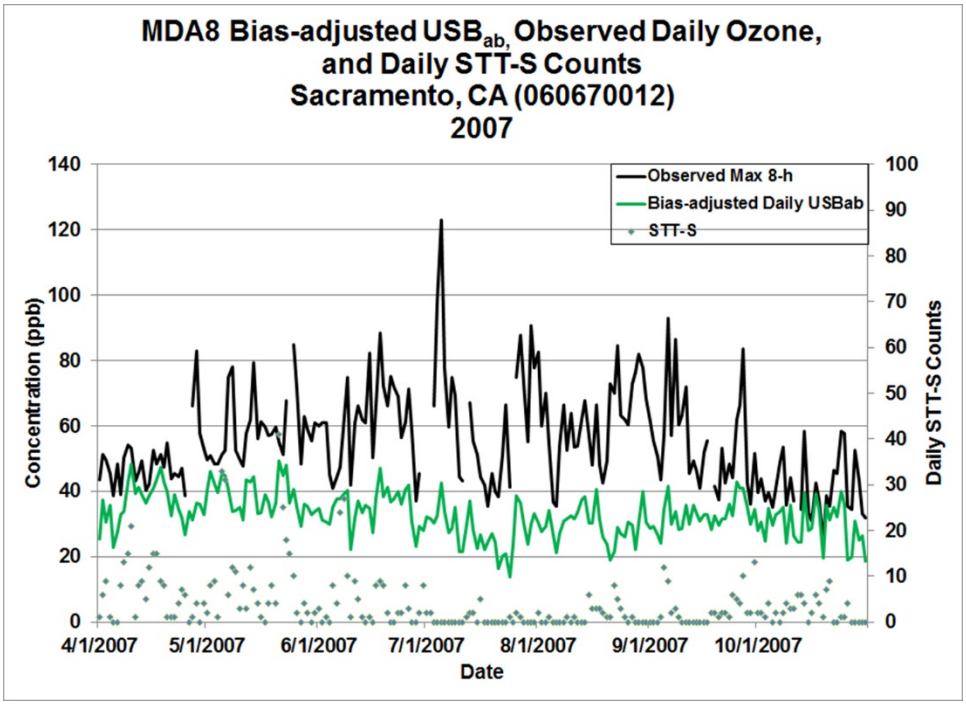



**Figure ES-14. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Sacramento, California (AQS ID 060670012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

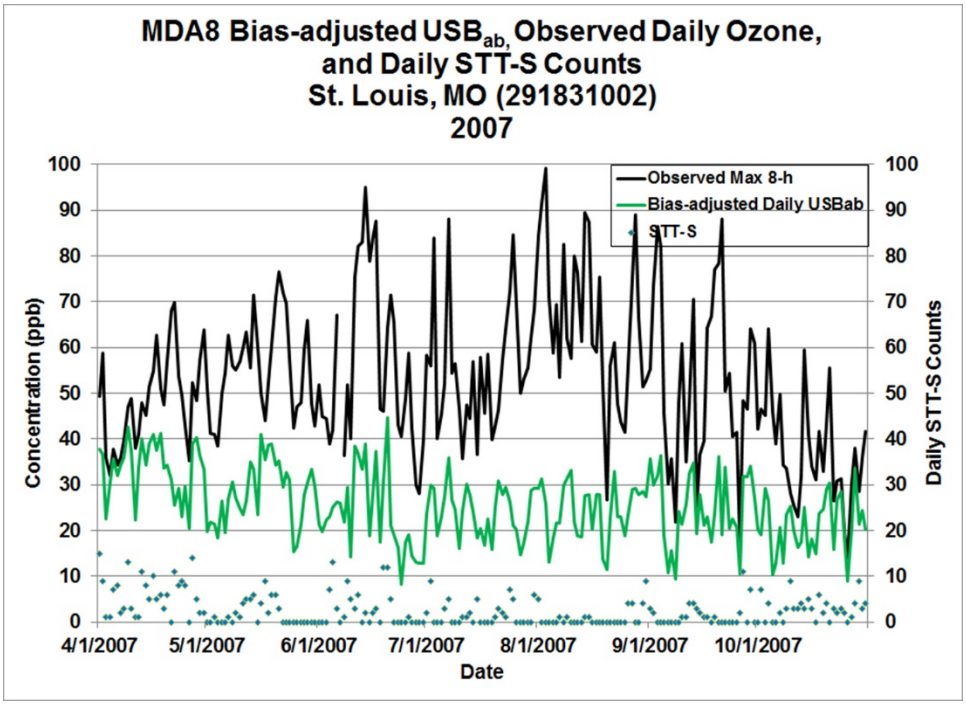



**Figure ES-15. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in St. Louis, Missouri (AQS ID 291831002) for April-October 2007. The daily stratospheric-tropospheric transport to surface (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 how the STT-S values are estimated.**

## 1. Introductory Comments

The Draft Policy Assessment for the Reconsideration of the Ozone National Ambient Air Quality Standards (PA) (EPA, 2022) document was prepared as a part of the current reconsideration of the 2020 final decision on the national ambient air quality standards (NAAQS) for ozone ($O_3$). When final, the PA is intended to "bridge the gap" between 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 $O_3$ NAAQS, completed in 2015, established the current primary and secondary standards (80 FR 65291, October 26, 2015). In December 2020, following a review of the scientific evidence associated with the current ozone standards, the EPA issued its final decision to retain the existing standards without revision. In the decision on subsequent litigation on the 2015 decisions, the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) upheld the 2015 primary standard but 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 draft PA (EPA, 2022, 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, specific attention is provided to the adequacy of the 3-year average of the 4$^{th}$ 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 $O_3$ concentrations, over either an annual or a 3-year period to protect vegetation. In addition, discussion is provided on the use of an annual W126 index versus the use of a 3-year average of the W126 metric.

### 1.1 Purpose of the Policy Assessment (PA) Document

As indicated in the draft PA (EPA, 2022, 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 $O_3$ 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 $O_3$ 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 will focus on the evidence described in the 2020 ISA, 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 will focus 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.

In the comments provided in this review of the draft PA (EPA, 2022), careful attention is focused on the technical aspects that the Agency has summarized in the draft PA (EPA, 2022) from the 2020 ISA (EPA, 2020a), 2020 PA (EPA, 2020b), and the 2020 $O_3$ NAAQS findings (Federal Register, 2020). It is important that the most current scientific results be accessible to the EPA Administrator so that he can reach his decision based on the best available science.

## 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 $O_3$ standards in the United States. The first fundamental principle is

> **Higher Hourly Average $O_3$ 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 $O_3$ 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 $O_3$ 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* developed under laboratory or empirical conditions. Lefohn et al. (2018), in the 24-coauthor international Tropospheric Ozone Assessment Report (TOAR), discussed the rationale for the selection of specific exposure

metrics for assessing human health and vegetation. The first fundamental principle provides guidance to policymakers and researchers on which part of the distribution of hourly average $O_3$ concentrations are biologically relevant for assessing human health and vegetation effects.

As noted in the Administrator's Review of the Ozone National Ambient Air Quality Standards document (Federal Register, 2020 – page 87267), the EPA Administrator, in her 2015 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 $O_3$ 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 $O_3$ exposures, and, in particular, that the effects reported in the controlled human exposure studies are due solely to $O_3$ 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 $O_3$ or higher (i.e., decreased lung function, increased airway inflammation, increased respiratory symptoms, airway hyperresponsiveness, and decreased lung host defense); (2) exposures to $O_3$ concentrations somewhat above 70 ppb have been shown to both decrease lung function and to result in respiratory symptoms; and (3) exposures to $O_3$ 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 $O_3$ exposure concentration documented to result in the widest range of respiratory effects (i.e., 80 ppb), and below the lowest $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ concentrations appear to be more important than lower concentrations in eliciting a response; plant sensitivity to $O_3$ varies with time of day and plant development stage; [and] quantifying exposure with indices that cumulate hourly $O_3$ 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 $O_3$ concentrations, coupled with the cumulative nature of the effects of $O_3$ 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 draft PA (EPA, 2022, page 4-7) notes:

> [8] Both the 2020 and 2013 ISAs reference the longstanding recognition of the risk posed to vegetation of peak hourly $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022, page 4-7), 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, 2020). Further comment on the Administrator's use of the number of hourly average $O_3$ concentrations $\geq$ 100 ppb (N100), as well as the number of days when hourly average $O_3$ concentrations are $\geq$ 100 ppb (D100), will be discussed 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 $O_3$ concentrations for assessing human health and vegetation effects.

## 2.1 Human Health

As noted, the draft PA (2022) is developed for consideration by the EPA Administrator in reaching a decision on the reconsideration of the December 2020 decision to retain the existing $O_3$ NAAQS. For the 2015 $O_3$ rulemaking, the EPA noted that higher $O_3$ 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 $O_3$ concentrations. The work utilized ambient-type elevated concentrations and compared them to constant concentration exposures with the same concentration × time product to assess respiratory effects. Commenting on these earlier results, Lefohn, Hazucha, Shadwick, and Adams (Lefohn et al., 2010) concluded that higher $O_3$ concentrations are important in $FEV_1$ 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 $O_3$ 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 $O_3$ concentrations and their influence on the form of the human health $O_3$ standard are as follows:

**Hazucha et al. (1992)**

> Several recent studies have suggested that in estimating exposure dose ($O_3$ concentration [C] × exposure time [T] × ventilation [V]), $O_3$ concentration needs to be weighted more heavily than either ventilation or duration of exposure in the estimates. Our observations demonstrate that the product of $\overline{C}$ (mean) × T × V is not a sufficient index of exposure.

**Adams (2003, 2006a)**

> These results support previous evidence that $O_3$ concentration has a greater singular effect in the total inhaled $O_3$ dose than do $V_E$ and exposure duration.

**Hazucha and Lefohn (2007)**

> As indicated in Hazucha et al. (1992), several recent studies have suggested that in estimating exposure dose ($O_3$ concentration [C] × exposure time [T] × ventilation [V]), $O_3$ concentration needs to be weighted more heavily than either ventilation or duration of exposure in the estimates. Our observations demonstrate that the product of $\overline{C}$ (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 $O_3$-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 $FEV_1$. 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 (Silverman et al., 1976), the assumption supported by a great majority of human studies published in the subsequent 30 years. 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 $O_3$ concentration, ventilation, and exposure duration on lung function response, Drechsler-Parks et al. (1990) found that in acute exposure protocols, $O_3$ 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 ($O_3$) hourly average concentrations elicit a greater effect on hour-by-hour physiologic response (i.e., forced expiratory volume in 1 s [$FEV_1$]) 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 $O_3$ and spirometric lung function decrements is not linear. In attempting to derive the $O_3$ 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 $O_3$ standard does not appear to be consistent for predicting the hour-by-hour pattern of $FEV_1$ responses obtained following exposures to realistic concentration profiles. Because of nonlinearity, the elevated hourly average concentrations influence the observed instantaneous human health effect (i.e., $FEV_1$ decrement). Alternative forms of the $O_3$ 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 (2010)**

Controlled human laboratory studies have shown that there is a disproportionately greater pulmonary function response from higher hourly average ozone ($O_3$) concentrations than from lower hourly average values and thus, a nonlinear relationship exists between $O_3$ dose and pulmonary function ($FEV_1$) response. We have reanalyzed data from five controlled human response to $O_3$ 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 $O_3$ exposures based on the frequency of breathing (fB) endpoint. In a subsequent paper, Schelegle et al. (2009) applied this concept to include an $FEV_1$ endpoint. Based on VAR/STW(i.e., variable/stepwise) $FEV_1$ response pattern, we have used a similar approach and identified three $FEV_1$ phases associated with exposure to VAR/STW $O_3$ concentrations: (i) a 2 to 3-h initial "induction phase" in response, (ii) followed by the onset of a statistically significant $FEV_1$ nonlinear "response phase," and (iii) a final "reversal phase" (i.e., change in direction of the slope of the $FEV_1$ decrement towards baseline as the hourly average $O_3$ 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. 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 $O_3$ exposures compared to the lower hourly average values and a nonlinear relationship between $O_3$ dose and $FEV_1$ 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., 2010). 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, 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 responses which depended on the applied hourly ozone concentration pattern.

The Draft PA (EPA, 2022, 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 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 $FEV_1$ 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 $FEV_1$ 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. The reason that the $FEV_1$ responses were similar was because there was "recovery" occurring in the variable exposure regime versus the cumulative response based on the square-wave exposure. Following the peak $FEV_1$ response, usually 1 to 2 h after the peak hourly $O_3$ concentration, lung function improved despite continuing $O_3$ exposure. Results from the 80 ppb variable profiles applied in Adams (2003, 2006a) show a recovery as $O_3$ concentrations decline to 50 ppb. Similarly, results from Hazucha et al. (1992) and Adams (2006b) show reversal of $FEV_1$ response as the concentration drops from 60 to 0 ppb. The relatively rapid recovery found 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 $O_3$ hourly average concentrations decrease towards 50 ppb and lower.

The observation that greater instantaneous $FEV_1$ decrements occurred 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 $O_3$ standard does not appear to be consistent for predicting the hour-by-hour pattern of $FEV_1$

responses obtained following exposures to realistic concentration profiles. Because of nonlinearity, the elevated hourly average concentrations influence the observed instantaneous human health effect (i.e., $FEV_1$ decrement). Hazucha and Lefohn (2007) suggested that alternative forms of the $O_3$ 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.

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 4$^{th}$ hour and then began to decrease. The result was that the three $FEV_1$ phases described in Lefohn et al. (2010) (Lefohn, Hazucha, Shadwick, and Adams), associated with exposure to the variable/stepwise $O_3$ 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 $FEV_1$ nonlinear "response phase," and the final phase was the "reversal phase" (i.e., change in direction of the slope of the $FEV_1$ decrement towards baseline as the hourly average $O_3$ concentration was decreased). Note that although the "reversal phase" occurred, the $FEV_1$ 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 $FEV_1$ responses have been reported at 6.6 hours regardless of the exposure protocol (i.e., constant versus stepwise)." However, as noted above, compared to the constant 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 and appears to indicate that the 8-h average form of the $O_3$ standard may need to be revisited sometime in the future.

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 ($O_3$) concentrations than from lower hourly average values and thus, a nonlinear relationship exists between $O_3$ dose and pulmonary function ($FEV_1$) 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 $O_3$ exposure. For understanding cumulative $O_3$ effects on $FEV_1$, 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 $O_3$ activity, as well as the 2019 rulemaking activity (Federal Register, 2020), one controlled human exposure experiment published was the key study that the EPA focused on for the selection of the primary $O_3$ 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 $O_3$ standard. Professor Milan Hazucha and I designed the hour-by-hour exposure regimes for the Schelegle et al. (2009). However, before discussing why this single controlled human exposure experiment outweighed the other studies, I would like to

describe the Agency's rationale in the 2015 $O_3$ 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 $O_3$ standard to protect human health.

When evaluating the epidemiological and controlled human exposure study results, the higher $O_3$ concentrations were an important consideration (Federal Register (2015 – page 65354). The EPA (Federal Register, 2015 – page 65343) agreed with the conclusions of the 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 $O_3$ concentrations. Specifically, the Agency recognized that the effects reported in controlled human exposure studies were due solely to $O_3$ 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 $O_3$-induced decreases in lung function and inflammation in healthy, exercising adults. The draft PA (EPA, 2022, page 3-93) notes that epidemiologic studies provide limited insight regarding exposure concentrations associated with health outcomes that might be expected under air quality conditions that meet the current standard.

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 $O_3$ 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 $O_3$ 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, 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 $O_3$ concentrations in the lower portions of ambient distributions. 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 $O_3$ concentrations greater than the mid- and low-level values resulted in small differences in the estimates of mortality and morbidity risks as a theoretical 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) but the low end of the distribution of concentrations

shifts upward (i.e., increasing mortality), ***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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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.

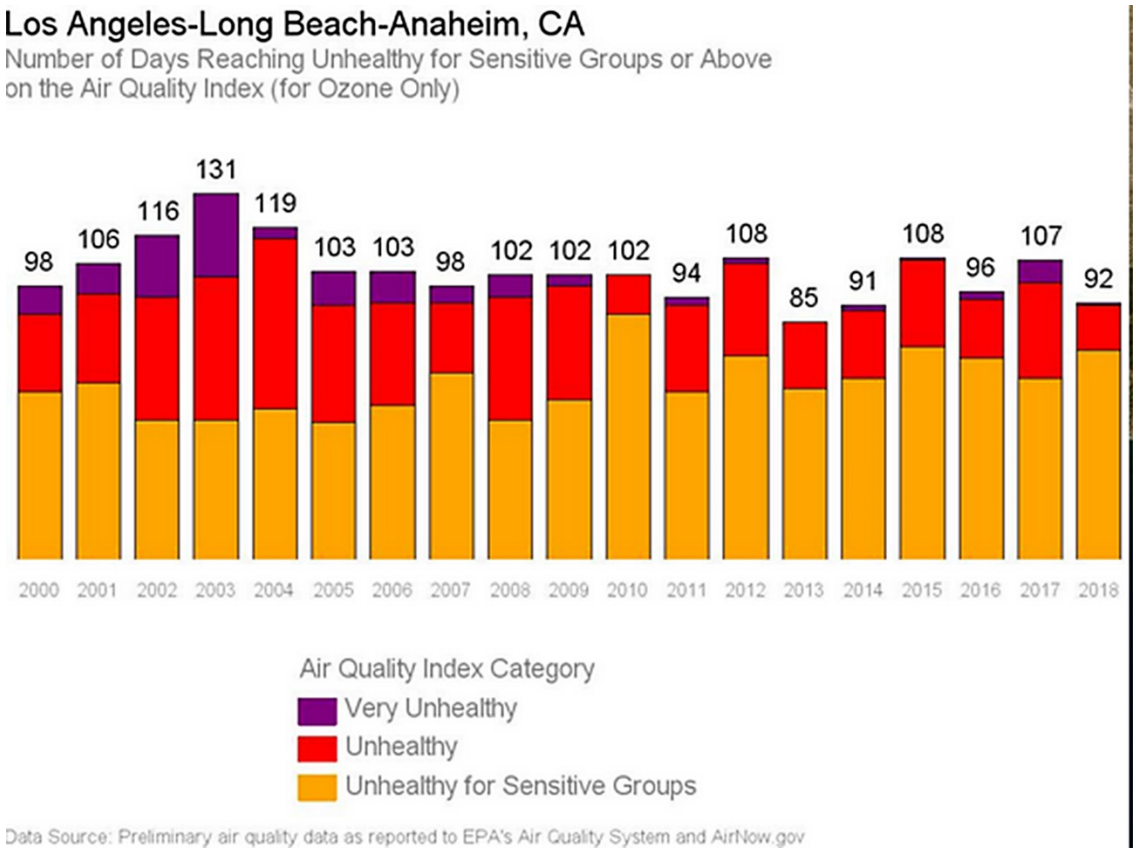



**Figure 2-3. Number of days reaching unhealthy levels for sensitive groups or above on the Air Quality Index for Los Angeles-Long Beach -Anaheim, CA. Source: EPA, 2019c: A Look Back: Ozone in 2018.**
**https://epa.maps.arcgis.com/apps/Cascade/index.html?appid=9bec4031ba6f4887a9f332a8f058b198**

## 2.2 Vegetation

For vegetation, the EPA ISA (2013, 2020a) concluded that (1) $O_3$ effects in plants are cumulative; (2) higher $O_3$ concentrations appear to be more important than lower concentrations in eliciting a response; (3) plant sensitivity to $O_3$ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the $O_3$ 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 $O_3$ concentrations in the Los Angeles area occurred, provides independent confirmation of the experimental studies described below for the greater importance of the higher hourly average $O_3$ 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 $O_3$ exposures between 1980 and 2000 were related to improvements in tree conditions. The frequency of midrange hourly average $O_3$ concentrations was little changed over this period. EPA suggested

that it was the reduction in the higher hourly average $O_3$ concentrations that was responsible for the improvement in tree health. EPA notes these results in the draft PA (EPA, 2022, page 4-88). 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 on 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 $O_3$ 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 $O_3$ concentrations in relation to the mid-range and lower concentrations. For more moderate $O_3$ 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 $O_3$ 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 $O_3$ 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 Laurence, 1999; Lee and Hogsett, 1999; Oksanen and Holopaninen, 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.

The interest in identifying $O_3$ exposure regimes for eliciting adverse effects began earlier with vegetation than with human health researchers. As indicated above, while controlled human health clinical study results regarding the importance of the higher hourly average $O_3$ concentrations were reported in the early 1990s (Hazucha et al., 1992), vegetation researchers reported in the 1960s that higher $O_3$ concentrations were an important factor for assessing vegetation $O_3$ effects. High $O_3$ 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 $O_3$ 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, I held 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 $O_3$ 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 $O_3$ 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. In 1983, Musselman et al. (1983) published experimental evidence supporting the hypothesis. Hogsett et al. (1985), applying exposure regimes designed by me, provided additional support to the Musselman et al. (1983) findings about the importance of the higher hourly average $O_3$ 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 $O_3$ 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 Holopaninen, 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 $O_3$ concentrations. In other words, by reducing the highest hourly average $O_3$ 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) $O_3$ effects in plants are cumulative; (2) higher $O_3$ concentrations appear to be more important than lower concentrations in eliciting a response; (3) plant sensitivity to $O_3$ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the $O_3$ 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 $O_3$ 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 $O_3$ and plant tissues is driven mainly by three distinct processes: changes in external $O_3$ concentration, $O_3$ uptake, and $O_3$ 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 $O_3$ stomatal uptake and plant response. The diurnal pattern of detoxification does not necessarily match the diurnal patterns of external $O_3$ concentration and $O_3$ 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 (nmol $m^{-2}$ $s^{-1}$). 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*** 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 $O_3$ 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 $O_3$ 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 $O_3$ concentrations (0.02 to 0.15 ppm) for 8 hr/day for 1 to 7 days. In beans, foliar injury developed when the internal $O_3$ flux exceeded 115 ~moles/$m^2$ within 1 hr (Bennett, 1979). However, a single 3-hr exposure at about half the $O_3$ concentration (0.27 compared to 0.49 ppm) required approximately 64 percent greater internal $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ and $SO_2$ 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. Severa1 lines of evidence suggest that higher concentrations have a greater influence in determining the impact of $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ in open-top exposure chambers showed that significant yield reductions occurred when the maximum $O_3$ 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 $O_3$ exposure profiles which had the equivalent total $O_3$ 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 $SO_2$ 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, 2020). As mentioned above, EPA (1986) noted the greater importance of concentrations compared to exposure duration. The total $O_3$ 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 both the seasonal average (e.g., M7 and M12) and the SUM00 exposure metrics are still used in experimental studies for assessing vegetation risk, as EPA (1986, 1996a, 2006, 2013) noted in its literature reviews, both the seasonal average and SUM00 $O_3$ 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 $O_3$ 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 $O_3$ concentrations rather than the mid- and lower-level values. Fig. 2-1 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 $O_3$ 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.

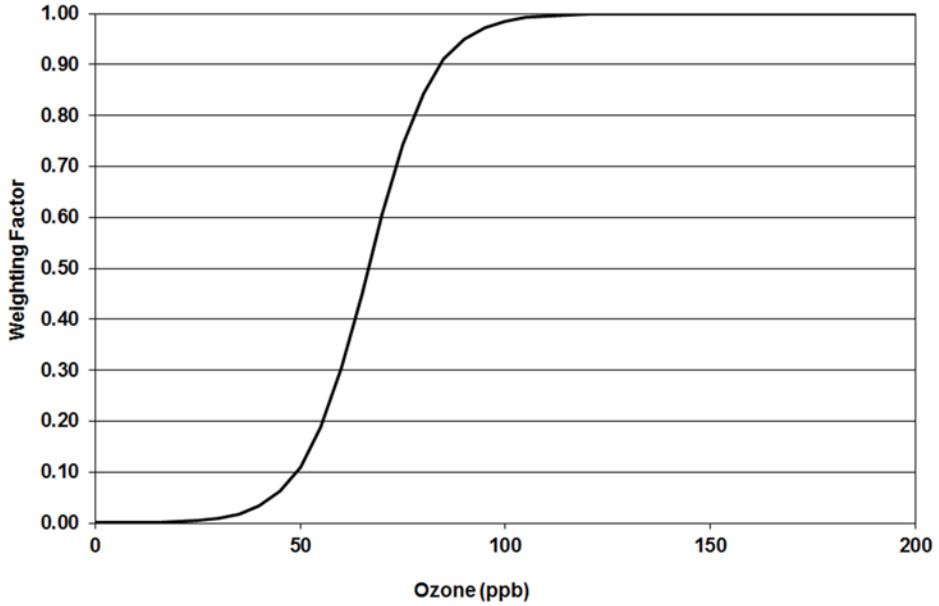



**Figure 2-1. The weighting applied to hourly average $O_3$ values for the calculation of the W126 exposure index (see Lefohn and Runeckles, 1987 and Lefohn et al., 1988 for further details).**

The 2013 ISA (EPA, 2013) noted that at the San Bernardino site, located near Los Angeles, reductions in ambient $O_3$ exposures between 1980 and 2000 were related to improvements in tree conditions. The frequency of midrange hourly average $O_3$ concentrations was little changed over this period. EPA (2013) suggested it was the reduction in the higher hourly average $O_3$ concentrations that was responsible for the improvement in tree health.

As indicated above, based on a thorough review of the vegetation literature, (1) $O_3$ effects in plants are cumulative; (2) higher $O_3$ concentrations are more important than lower concentrations in eliciting a response; (3) plant sensitivity to $O_3$ varies with time of day and plant development stage; and (4) quantifying exposure with indices that accumulate the $O_3$ 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 $O_3$ current reconsideration of the $O_3$ NAAQS (EPA, 2022), 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 added the frequency of hourly or daily $O_3$ average concentrations ≥ 100 ppb. However, as discussed in Section 4, one does not need the occurrence of hourly average $O_3$ concentrations ≥ 100 ppb for vegetation visible 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 for the W126 metric, where the stronger weighting for the highest $O_3$ concentrations resulted in areas with the highest W126 values standing out from those with 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 $O_3$ 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 $O_3$ 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×T=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 the 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 concentrations 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 biological important than the lower and mid values, is lost. 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, 2020), 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., see TOAR paper by 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 defensible**. In many cases, it may not be possible to identify a biologically relevant exposure metric based on modeling results. Prior to selecting and running a model for assessing human health or vegetation effects/risks, it is important to identify an exposure metric for the model that is justified based on biological principles.

The impacts of surface $O_3$ on human health and vegetation have prompted precursor emission reductions in the United States. As emissions change, the distribution of hourly $O_3$ 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 $O_3$ using the 4$^{th}$ 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 draft PA (EPA, 2022, page 2-20) notes that trends analyses show that metrics impacted by averaging longer time periods of hourly $O_3$ measurements, such as the 6-month (April-September) average of daytime (8am-7pm) $O_3$ 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) 4$^{th}$ 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 $O_3$ 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 opposite direction (i.e., the metrics associated with the highest concentrations exhibited decreases over time, while the mean and median metrics exhibited increases). Table 2-1 below provides a summary of the relationship between (1) two of the exposure metrics focused on the highest concentrations (i.e., the annual 4$^{th}$ highest daily maximum 8-h average concentration (4$^{th}$ 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 concentrations.

**Table 2-1. 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 $O_3$ concentrations for the 12-h

period from 08:00 to 19:59 local time April−September), which focus on the midrange of the $O_3$ hourly average concentration distribution, did not appear to adequately describe the magnitude and frequency of high $O_3$ 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 $O_3$ 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 $O_3$ control strategies in the United States and around the world. 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 $O_3$ 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 $O_3$ standards by reducing the higher hourly average $O_3$ 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 $O_3$ occurs with the result that lower hourly average $O_3$ 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 $O_3$ 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 $O_3$ concentrations for reducing 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, we know that based on the scientific literature about the importance of the higher hourly average concentrations discussed in this section, 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 $O_3$ 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 $O_3$ concentrations to protect human health and welfare. Section 3 discusses the second fundamental principle which is described as

**Daily Maximum Hourly Averaged $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ concentrations would move downwards toward the middle hourly average $O_3$ values, while the lower concentrations would move upwards toward the middle values.

The observation of the lower hourly average $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 (e.g., Lefohn et al., 1998; EPA, 2014b, 2020b, 2022; Simon et al., 2015; Lefohn et al., 2017, 2018). The shift is associated with less titration of $O_3$ 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 $O_3$ concentrations as emissions are reduced. We explore the changes in the frequency of both high and low levels of $O_3$ hourly average concentrations. Following the description of the changes in the distribution patterns, we then explore when during the year the highest hourly average $O_3$ 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 $O_3$ levels. Although the U.S. Court of Appeals for the District of Columbia Circuit (DC Circuit) in August 2019 ruled that background $O_3$ should not directly influence the setting of the level of the $O_3$ NAAQS, as described in Section 3.2, background $O_3$ 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 $O_3$ concentrations. Emission reductions will achieve the shifting of the higher hourly average $O_3$ 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 $O_3$ levels. This phenomenon is discussed in later subsections of Section 3. The distribution of background $O_3$ 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 $O_3$ hourly average concentrations may appear to be Gaussian-like with only the amplitude varying at each site. Thus, daily maximum hourly averaged $O_3$ 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 $O_3$ concentrations at both the high end and low end of the distribution. The "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 $O_3$ 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 4$^{th}$ 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; 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 $O_3$ of the lower hourly average concentrations as reduction in $NO_x$ emissions occurs (Lefohn et al., 1998; EPA, 2014b, 2020b, 2022; Simon et al., 2015; Lefohn et al., 2017, 2018).

To attain the $O_3$ standards, a strategy of reducing $NO_x$ emissions has been implemented to reduce the higher hourly average $O_3$ concentrations. As a result of the reduction of NO titration of $O_3$ by $NO_x$ reductions, the lower hourly average concentrations at many locations are shifted upward as the more biologically important higher hourly average $O_3$ 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 $O_3$ precursors in the United States on $O_3$ concentrations. Using daily 8-h average concentrations, the authors reported that decreasing $O_3$ trends generally occurred in the summer, in less urbanized areas, and at the upper end of the $O_3$ distribution (i.e., the higher 8-h concentrations). Conversely, increasing $O_3$ trends generally occurred in the winter, in more urbanized areas, and at the lower end of the $O_3$ distribution. As noted in EPA (2022, page 2-20):

> Simon et al. (2015) found that, similar to results presented in this section for DVs and annual 4$^{th}$ high MDA8 concentrations, the 95$^{th}$ 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 5$^{th}$ 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 $O_3$ values with extremely high and extremely low concentrations becoming less common. As a result, $O_3$ metrics impacted by high hourly $O_3$ concentrations, such as the annual 4$^{th}$ highest MDA8 value, decreased at most US sites across this period. Concurrently, metrics that are impacted by averaging longer time periods of hourly $O_3$ measurements, such as the 6-month (April-September) average of daytime (8am-7pm) $O_3$ 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 4$^{th}$ 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-98 (3C.7.2 Distribution of Hourly $O_3$ Concentrations) of the PA (EPA, 2020b, 2022), the Agency has updated its 2014 modeling analyses. It appears that these figures illustrated in the 2019 PA (EPA, 2020a) and those figures shown in the draft PA (EPA, 2022) are identical. EPA (2022, page 3C-81) notes that

> The precursor reductions used to estimate spatial and temporal patterns of $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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-$O_3$ 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 $O_3$ production chemistry in large population centers across the United States, as well as how $O_3$ chemistry in these areas may evolve in the future.

The draft PA (EPA, 2022) 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 $O_3$ concentrations for 2015-2017 at monitor locations in each urban area. For each hour of the day, the rectangular box represents the $25^{th}$ and $75^{th}$ 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 $75^{th}$ percentile minus the $25^{th}$ percentile) from the box, and dots which represent outlier values. Black boxplots represent observed hourly $O_3$ concentrations, while blue boxplots represent hourly $O_3$ concentrations adjusted to meet the current standard of 70 ppb. Red boxplots represent hourly $O_3$ concentrations adjusted for the 75 ppb scenario, and green boxplots represent hourly $O_3$ 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 $O_3$ 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 $O_3$ 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 PA (EPA, 2022) 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 PA (page 3C-101) as follows:

> The hourly plots show similar patterns in most of the urban study areas. $O_3$ 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 $O_3$ 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 $O_3$ concentrations while other study areas show decreases in $O_3$ concentrations for the 75 ppb scenario.
>
> In some urban study areas $O_3$ concentrations on the mid-range days, represented by the $25^{th} – 75^{th}$ percentile boxes, remained fairly constant (e.g. Boston) while in other urban study areas $O_3$ on mid-range days decreased (e.g. Atlanta). Although daytime $O_3$ 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 $O_3$ titration and thus increases $O_3$ concentrations. Nighttime increases in $O_3$ 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 $O_3$ 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 $O_3$ and increases in nighttime $O_3$ leading to a flatter diurnal $O_3$ pattern (e.g., Sacramento in Fig. 3C-73).

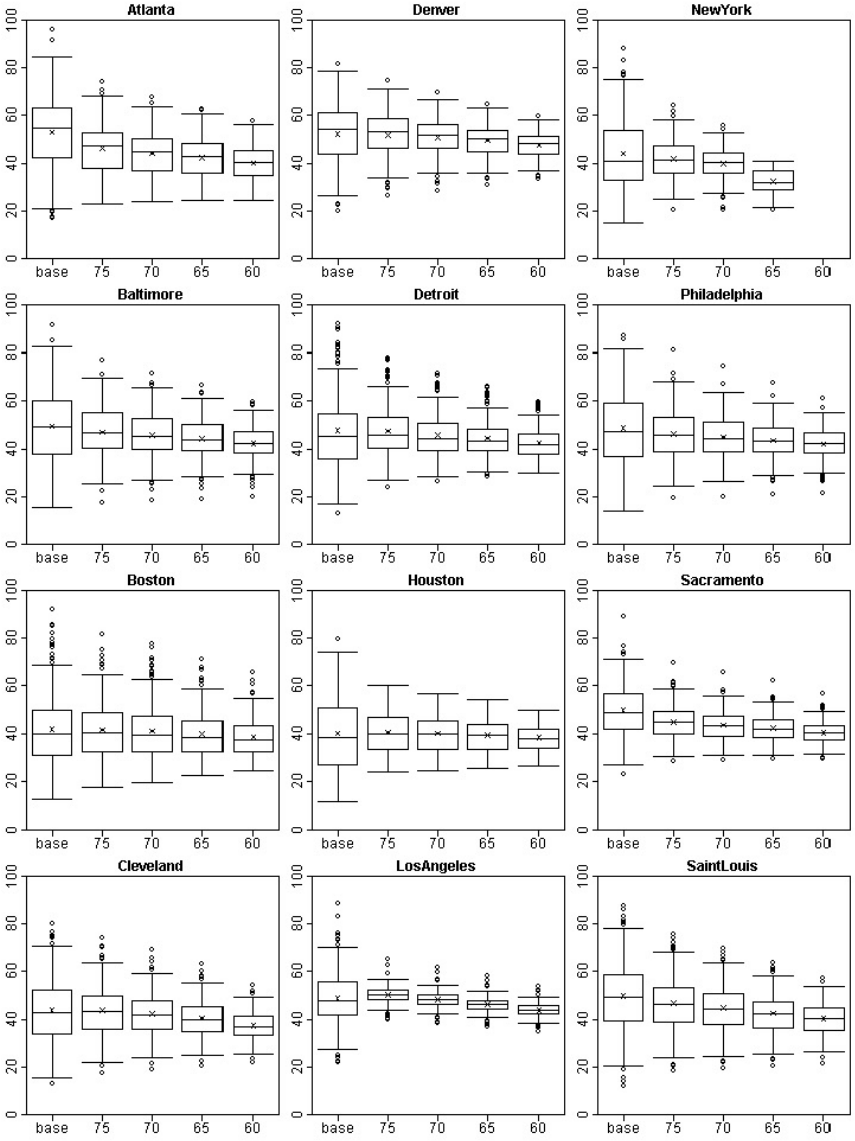



**Figure 4-9. Distributions of Composite Monitor Daily Maximum 8-hr Average Values for the 12 Urban Study Areas in the Epidemiology-based Risk Assessment. Plots depict values based on ambient measurements (base), and values obtained with the HDDM adjustment methodology when just meeting the 75, 70, 65 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).**

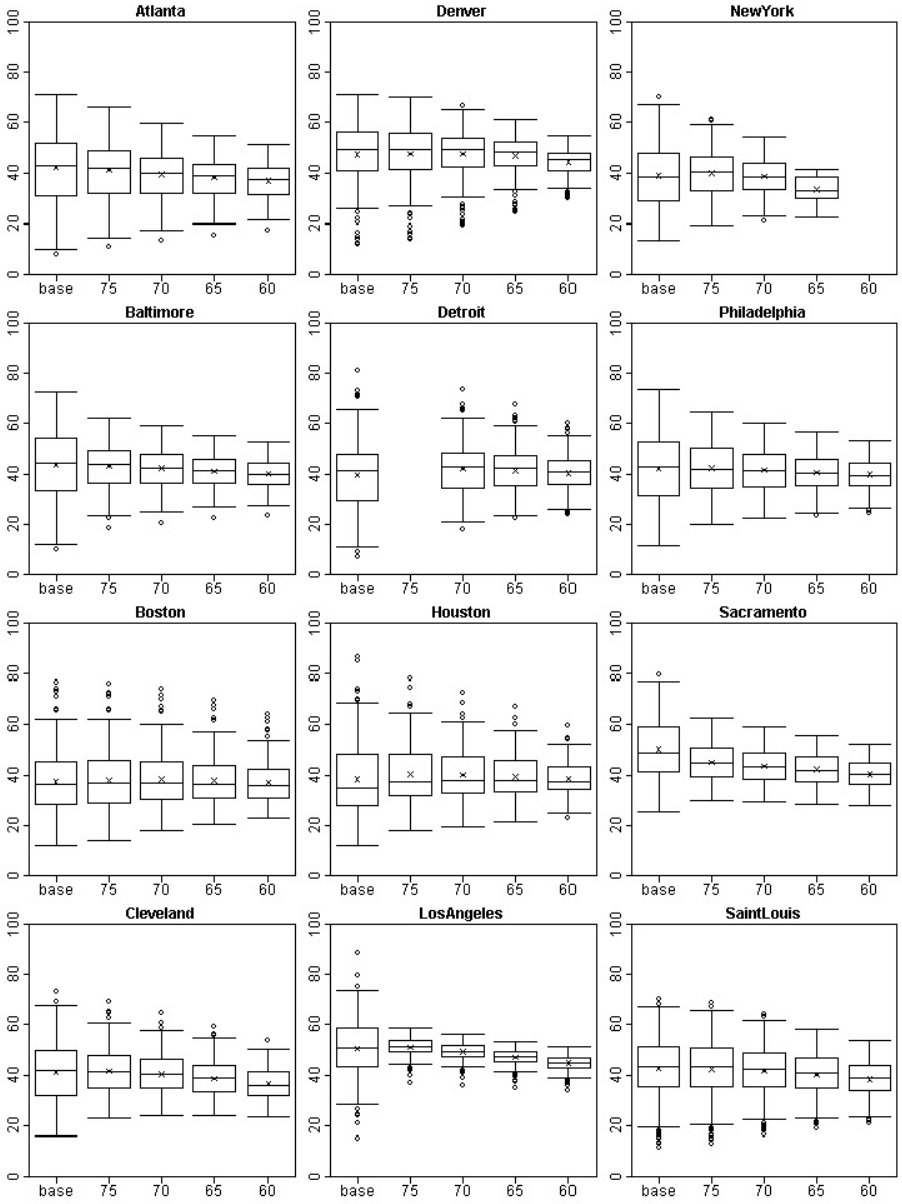



**Figure 4-10. Distributions of Composite Monitor Daily Maximum 8-hr Average Values for the 12 Urban Study Areas in the Epidemiology-based Risk Assessment. Plots depict values based on ambient measurements (base), and values obtained with the HDDM adjustment methodology when just meeting the 75, 70, 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).**

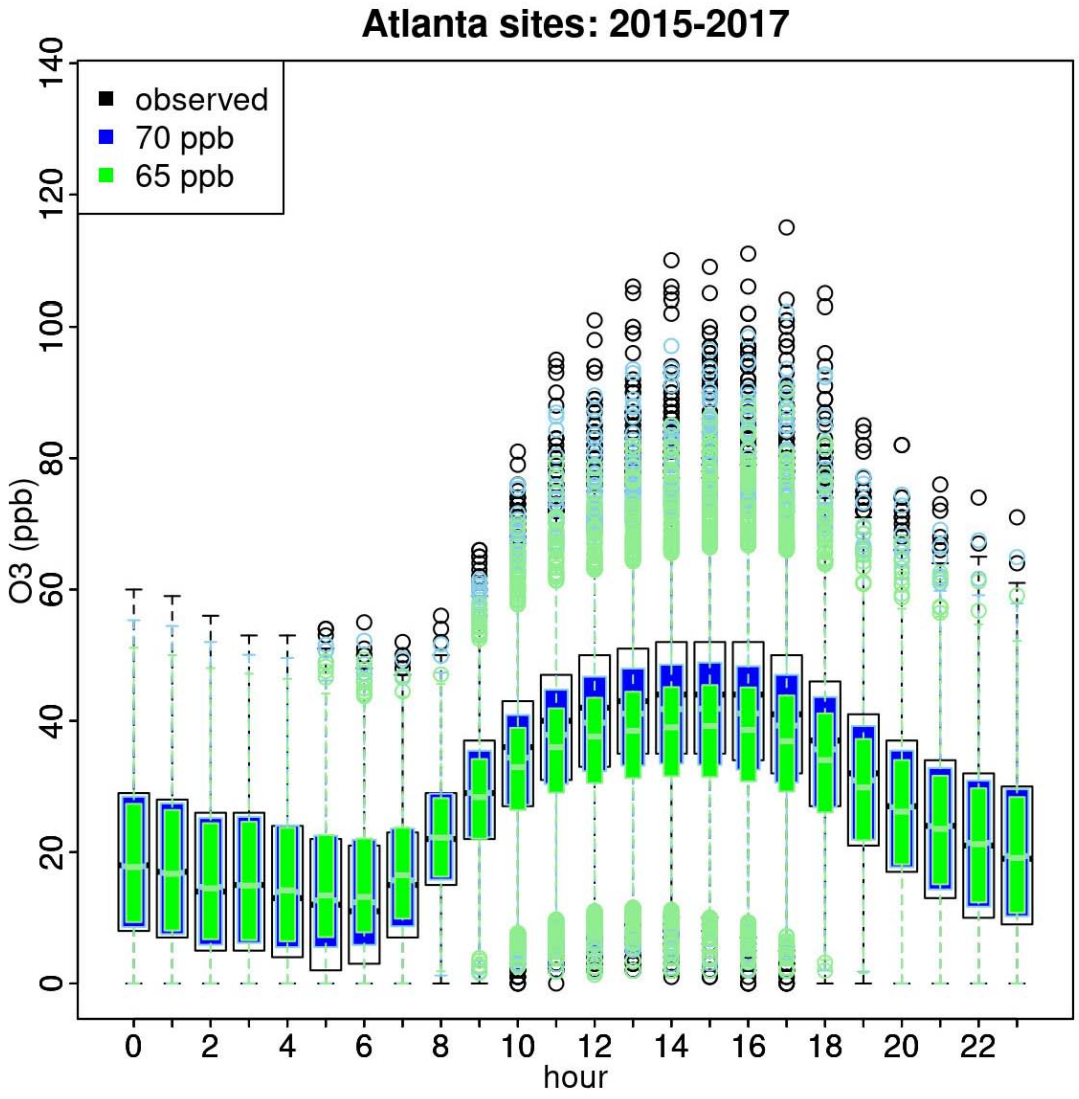



**Figure 3C-67. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Atlanta.**

**Figure 3-3. Figure 3C-67 from EPA (2022).**

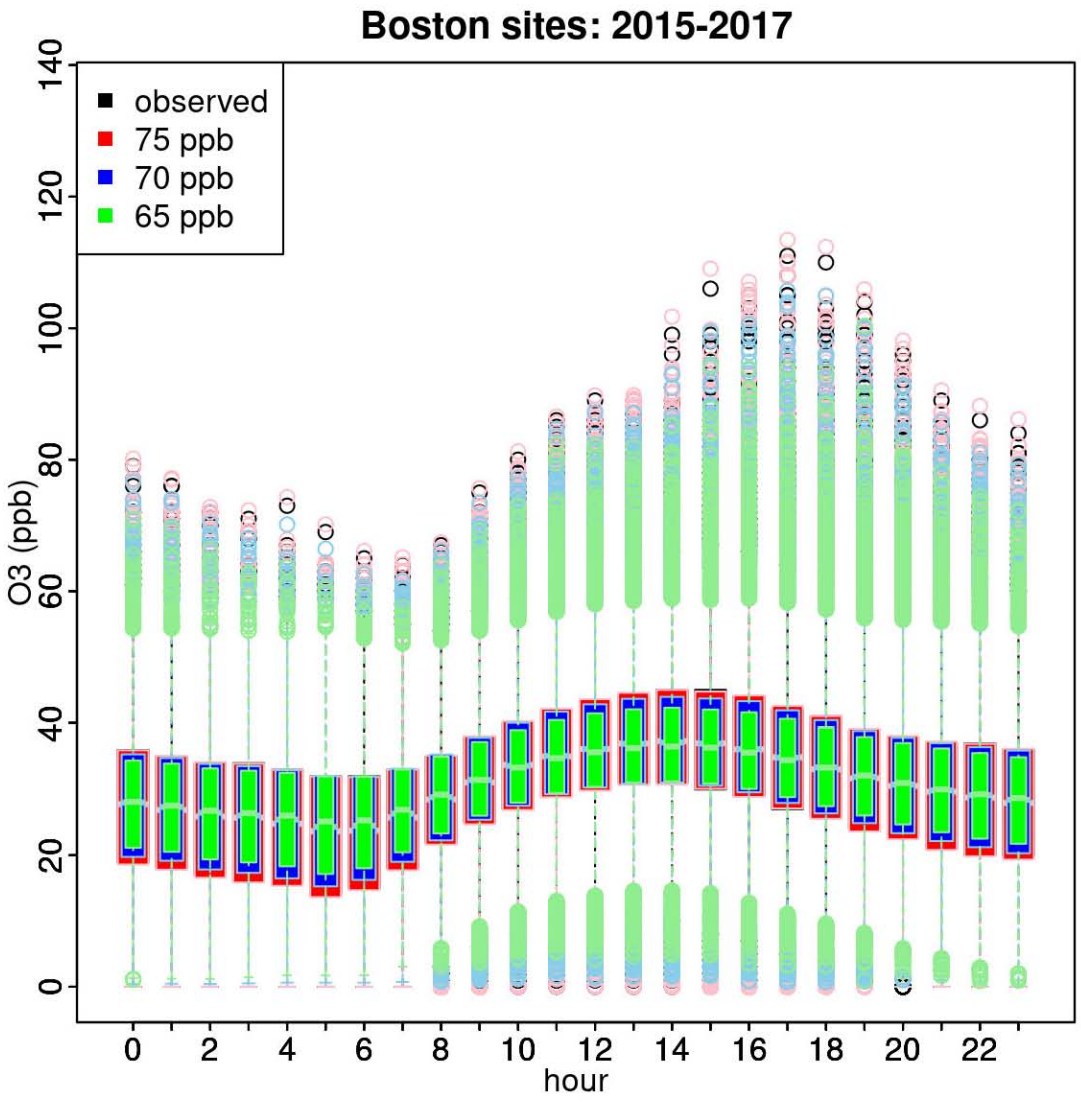



**Figure 3C-68. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Boston.**

**Figure 3-4. Figure 3C-68 from EPA (2022).**

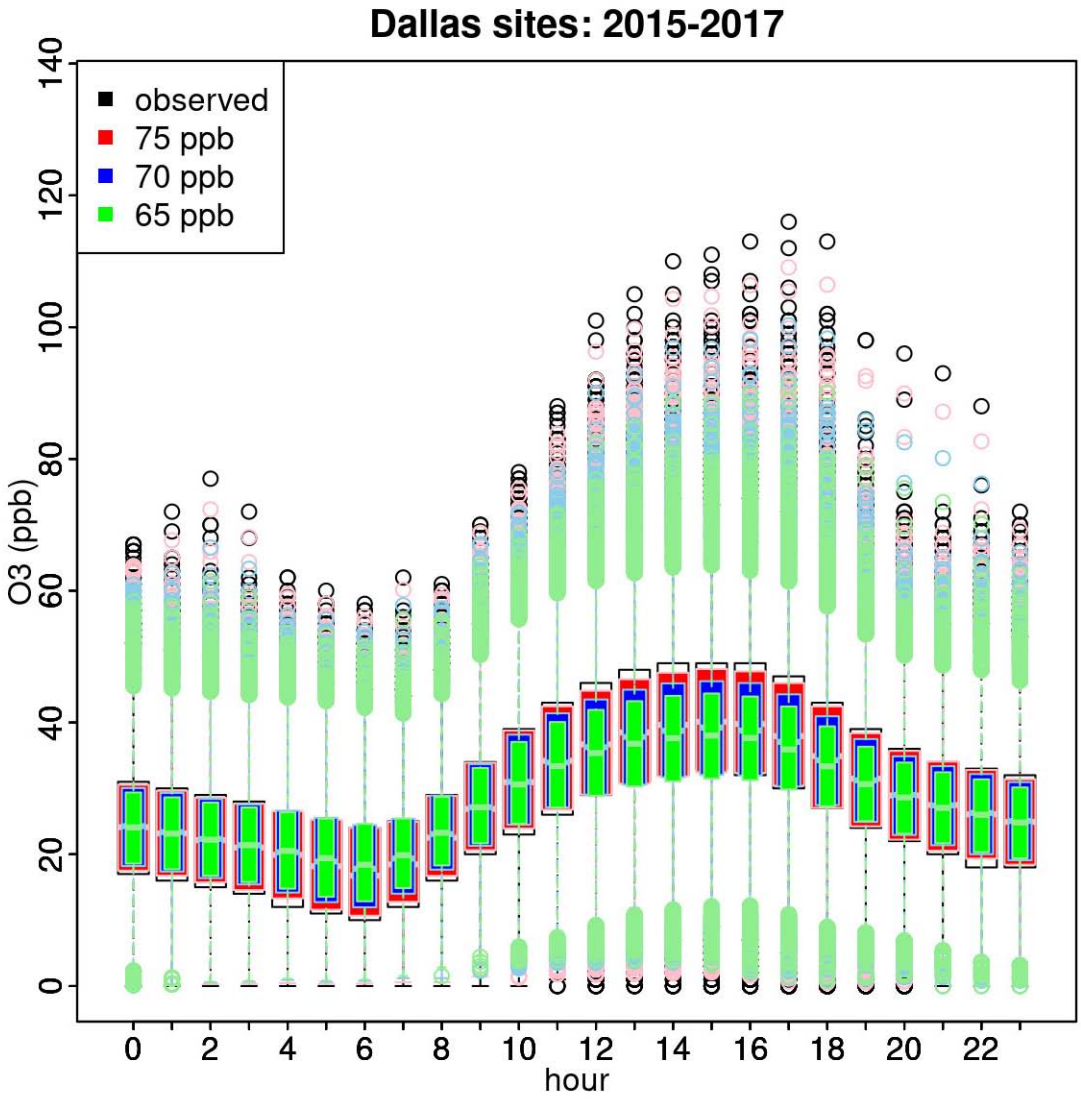



**Figure 3C-69. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Dallas.**

**Figure 3-5. Figure 3C-69 from EPA (2022).**

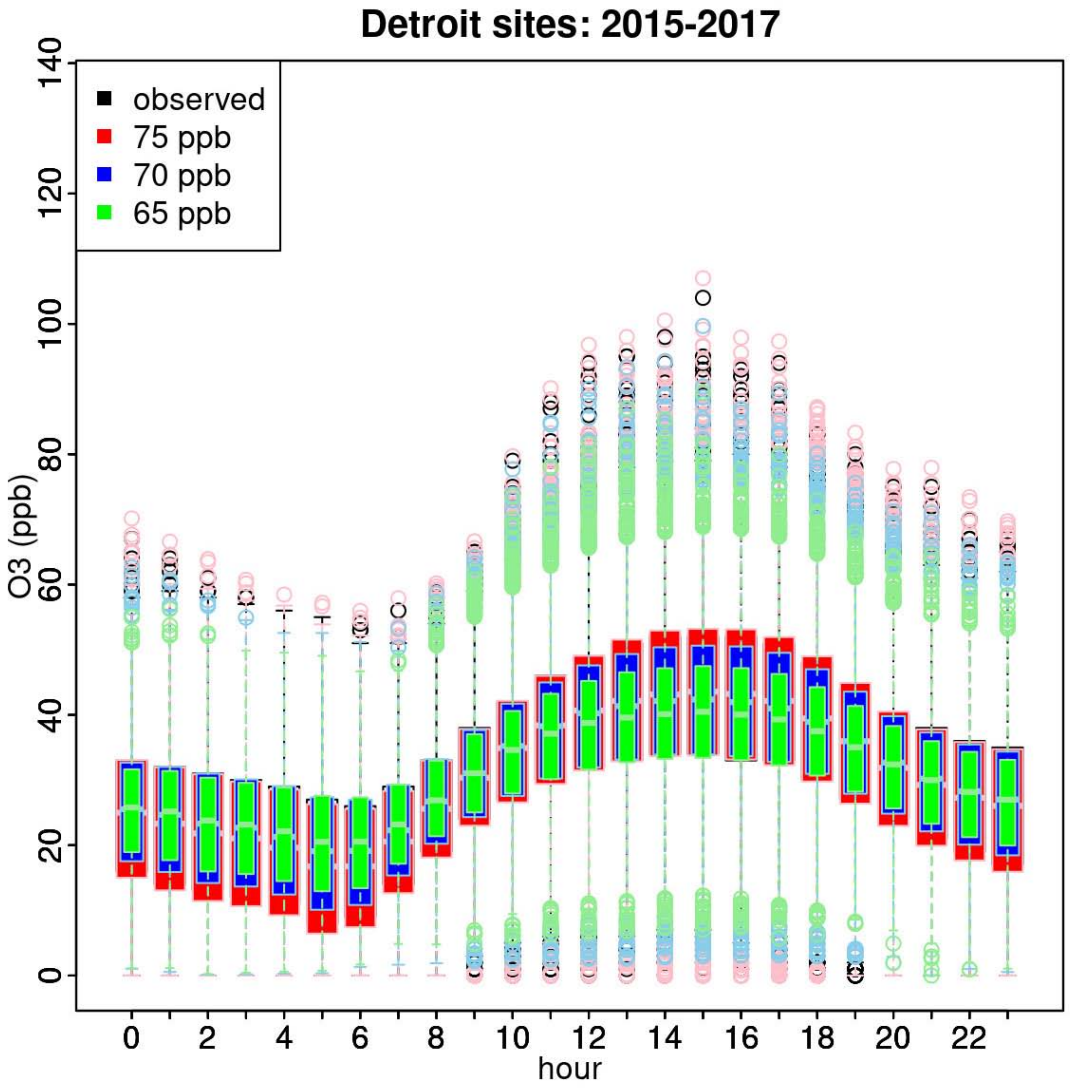



**Figure 3C-70. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Detroit.**

**Figure 3-6. Figure 3C-70 from EPA (2022).**

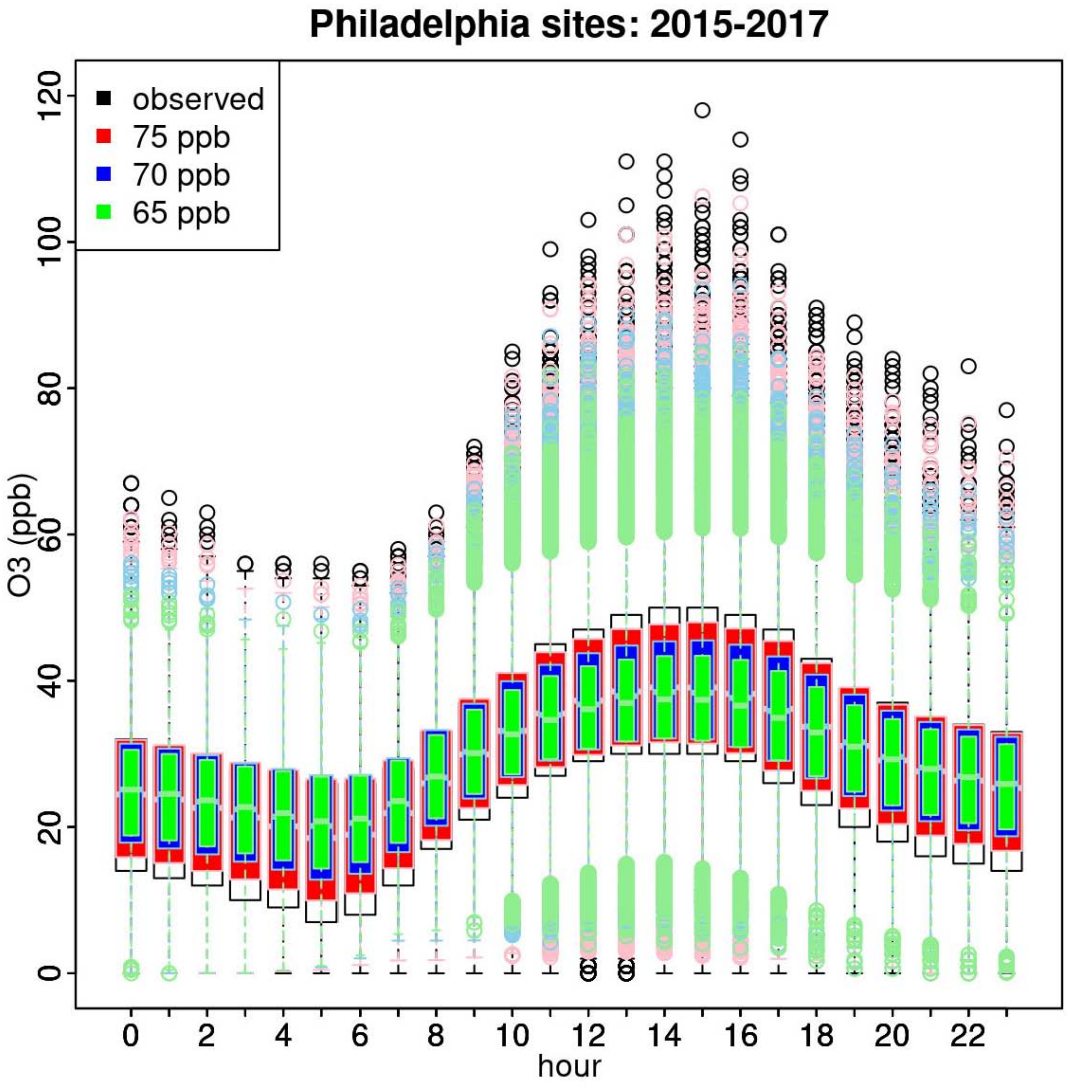



**Figure 3C-71. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Philadelphia.**

**Figure 3-7. Figure 3C-71 from EPA (2022).**

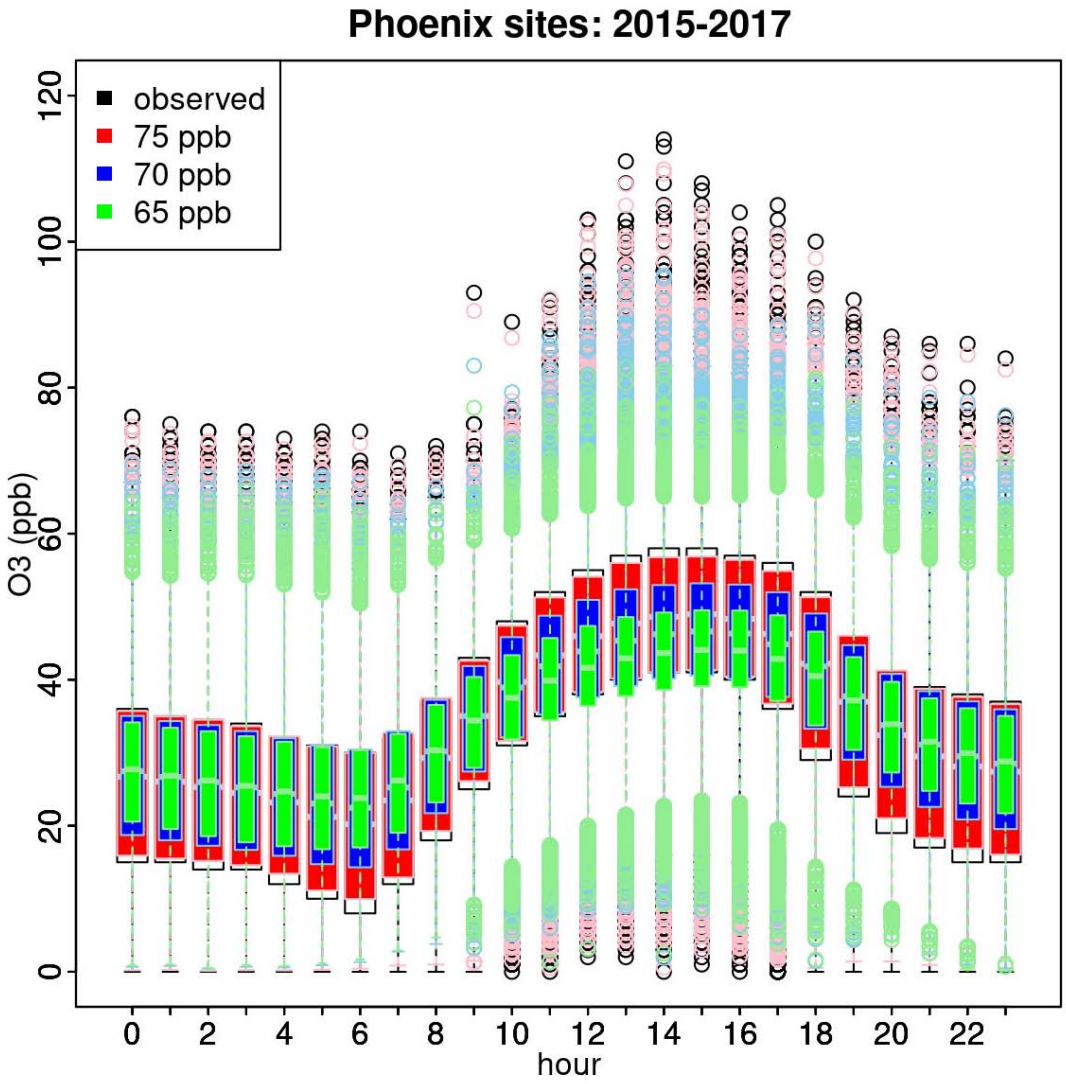



**Figure 3C-72. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Phoenix.**

**Figure 3-8. Figure 3C-72 from EPA (2022).**

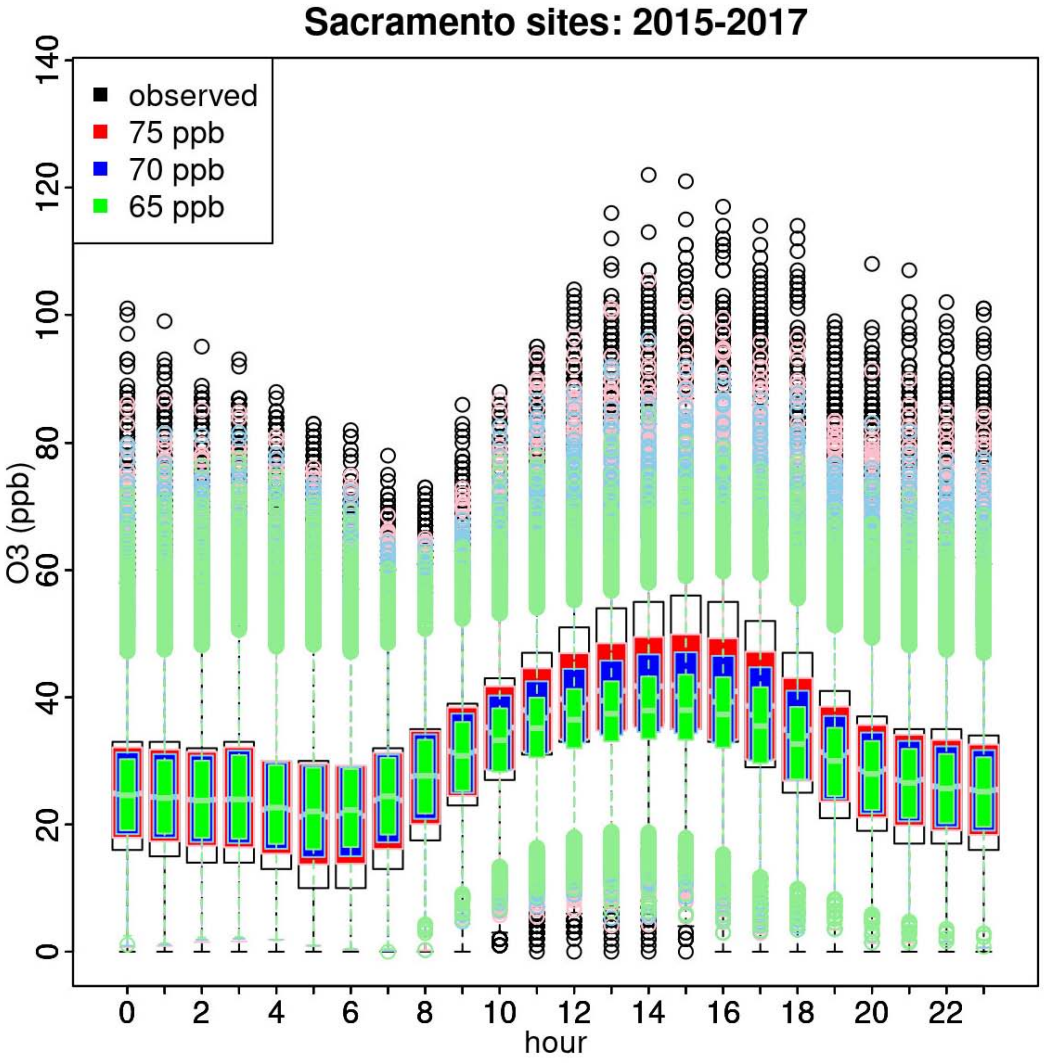



**Figure 3C-73. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in Sacramento.**

**Figure 3-9. Figure 3C-73 from EPA (2022).**

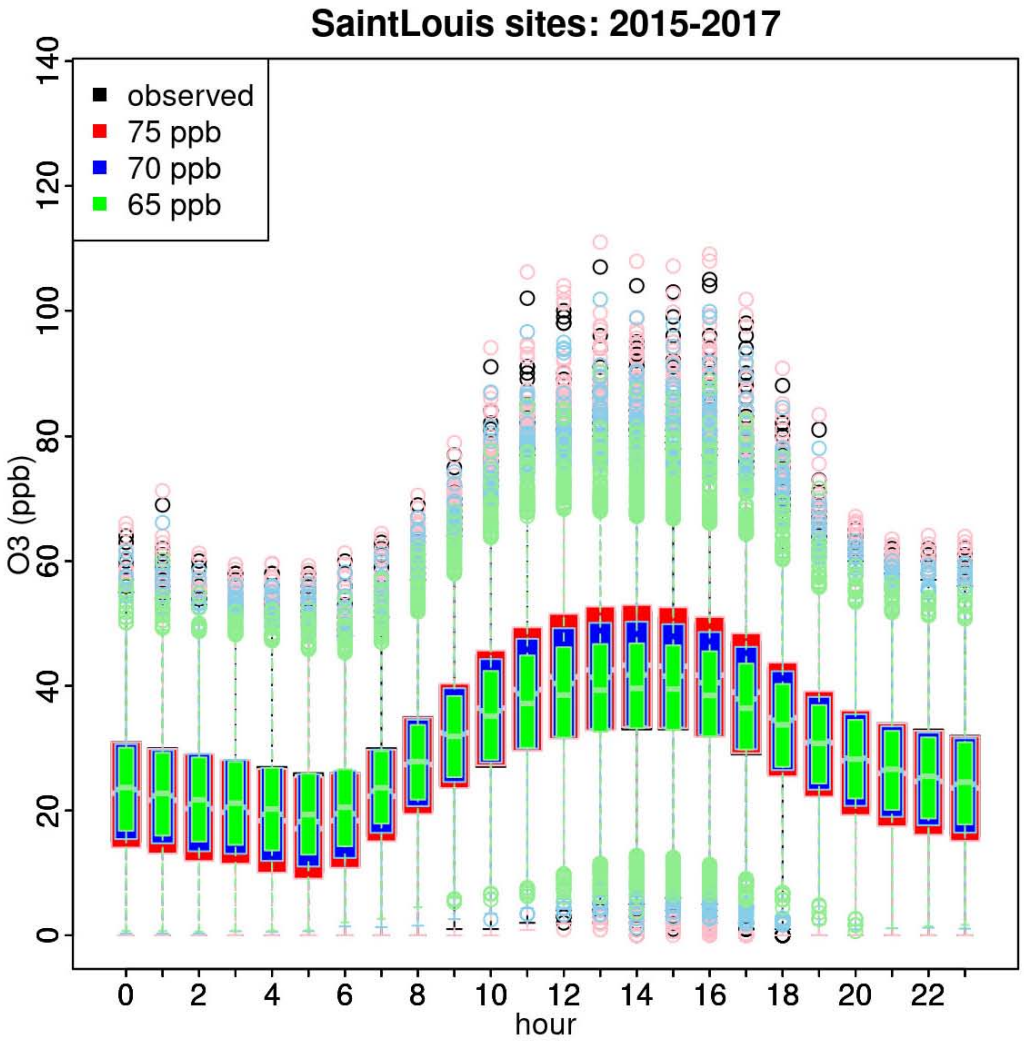



**Figure 3C-74. Diurnal distribution of hourly $O_3$ concentrations at monitoring sites in St. Louis.**

**Figure 3-10. Figure 3C-74 from EPA (2022).**

The Draft PA (EPA, 2022, 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 $O_3$ 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 EPA (2022, page 3C-110) concludes

> In general, the density scatter plots show that the HDDM adjustment procedure predicts increases in MDA8 $O_3$ at low ambient air concentrations and decreases in MDA8 $O_3$ at high concentrations (Figure 3C-83 through Figure 3C-90). The vast majority of the increases in MDA8 $O_3$ 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 $O_3$ 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 $O_3$ concentrations are reduced in urban study areas with large domain-wide reductions in U.S. anthropogenic $NO_x$ emissions.

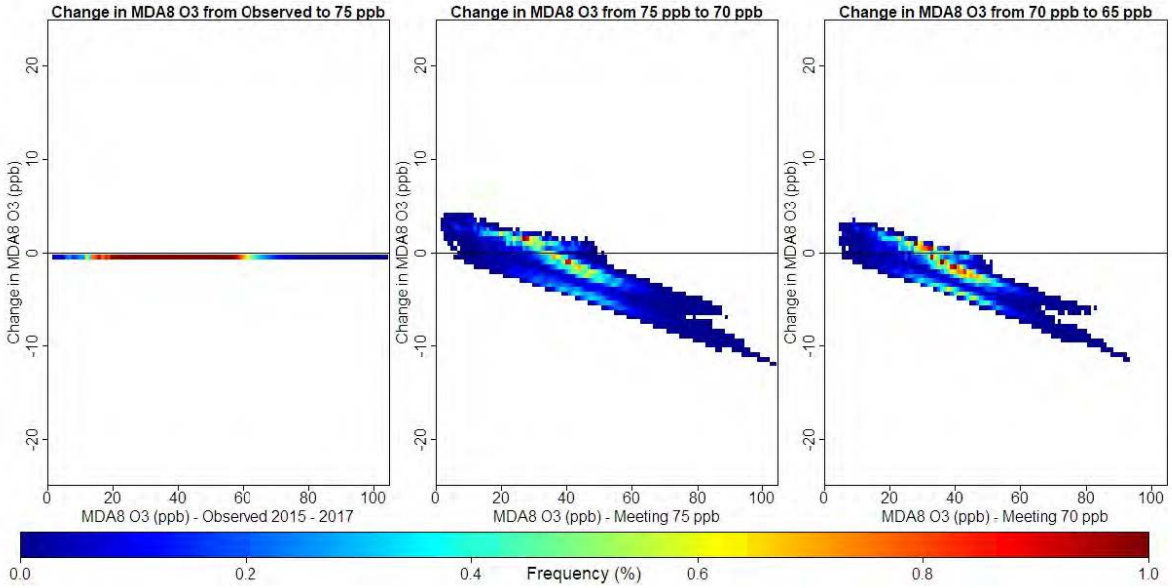



**Figure 3C-83. Changes in MDA8 $O_3$ based on HDDM adjustments in the Atlanta study area.**

**Figure 3-11. Figure 3C-83 from EPA (2022).**

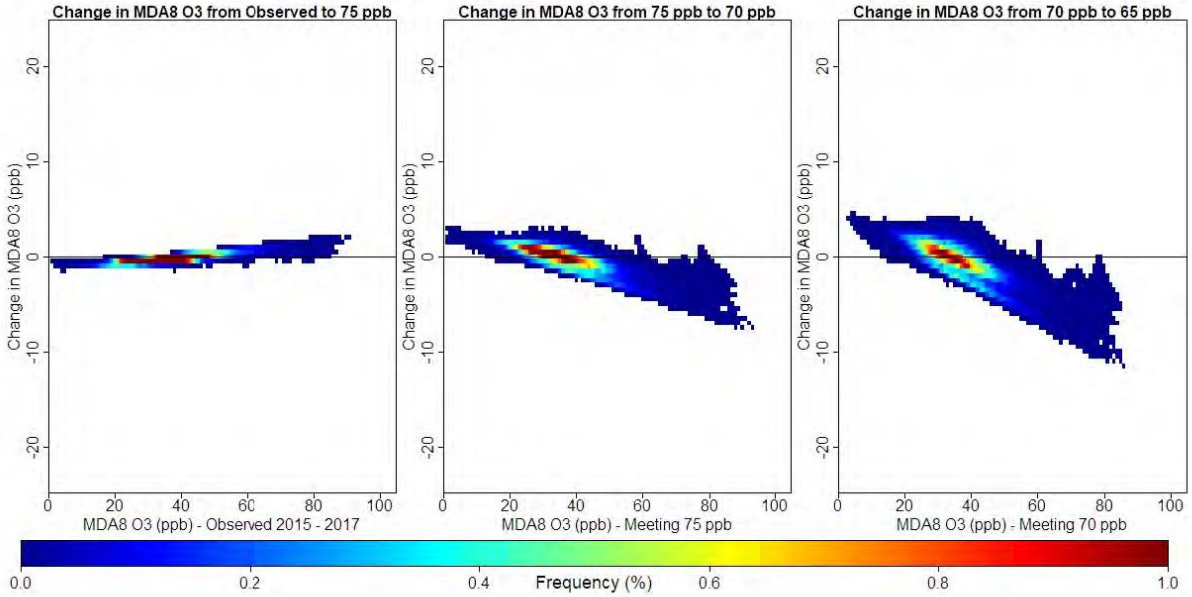



**Figure 3C-84. Changes in MDA8 $O_3$ based on HDDM adjustments in the Boston study area.**

**Figure 3-12. Figure 3C-84 from EPA (2022).**

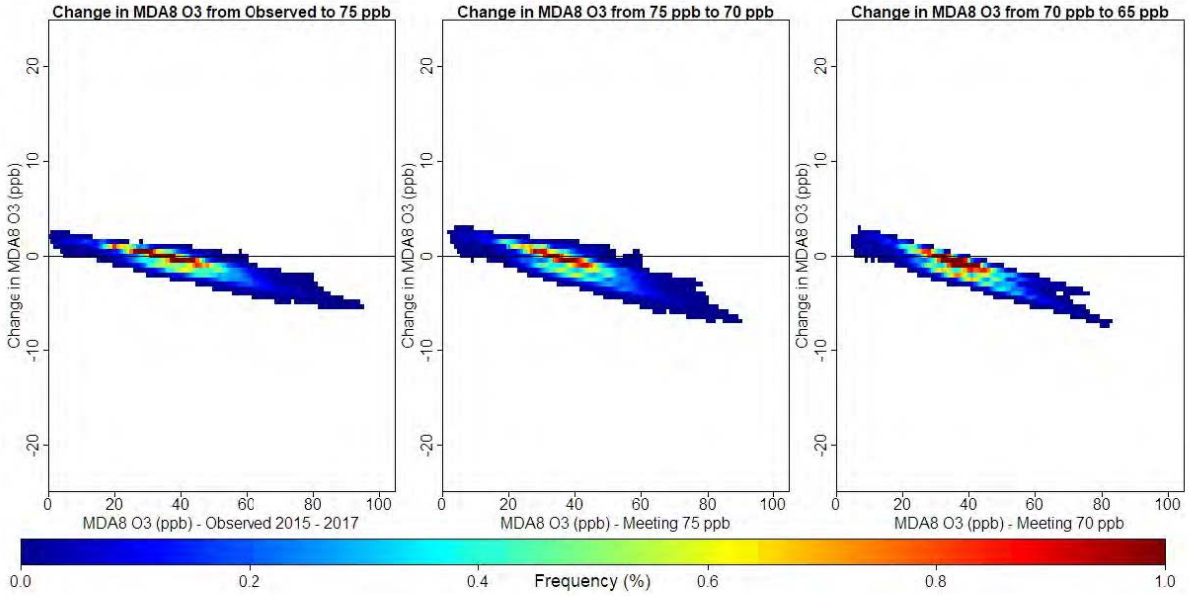



**Figure 3C-85. Changes in MDA8 $O_3$ based on HDDM adjustments in the Dallas study area.**

**Figure 3-13. Figure 3C-85 from EPA (2022).**

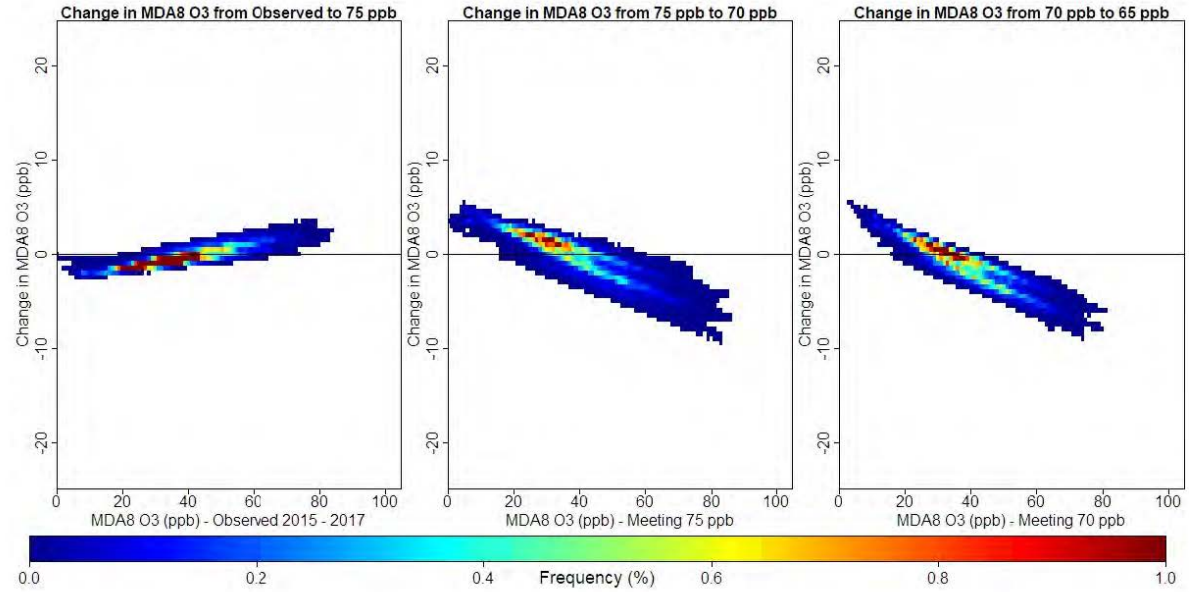



**Figure 3C-86. Changes in MDA8 $O_3$ based on HDDM adjustments in the Detroit study area.**

**Figure 3-14. Figure 3C-86 from EPA (2022).**

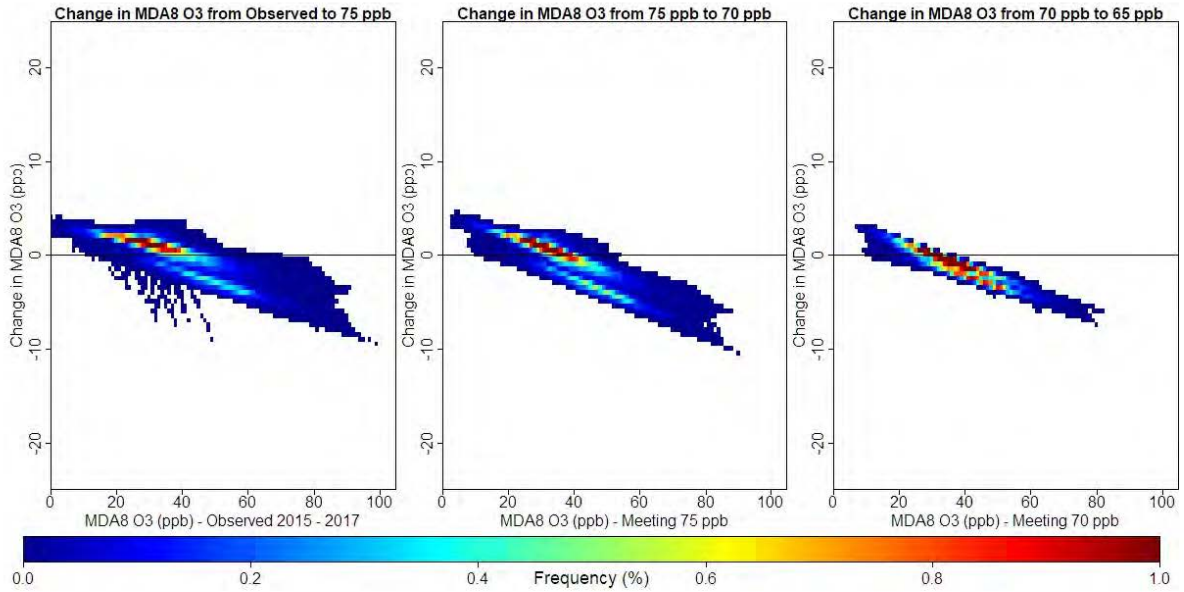



**Figure 3C-87. Changes in MDA8 $O_3$ based on HDDM adjustments in the Philadelphia study area.**

**Figure 3-15. Figure 3C-87 from EPA (2022).**

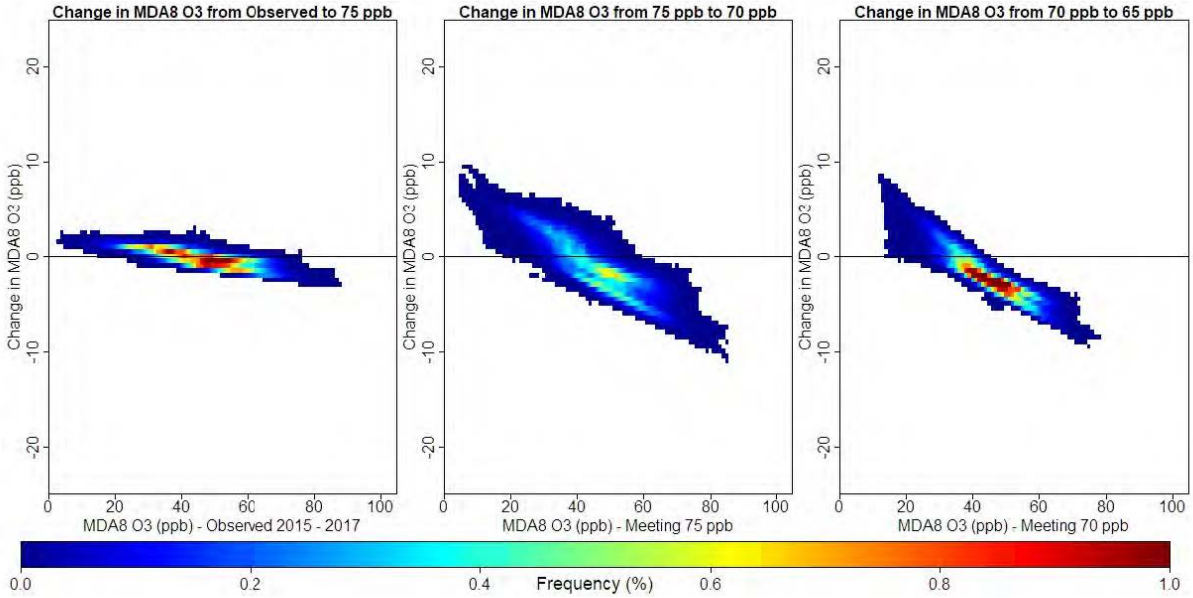



**Figure 3C-88. Changes in MDA8 $O_3$ based on HDDM adjustments in the Phoenix study area.**

**Figure 3-16. Figure 3C-88 from EPA (2022).**

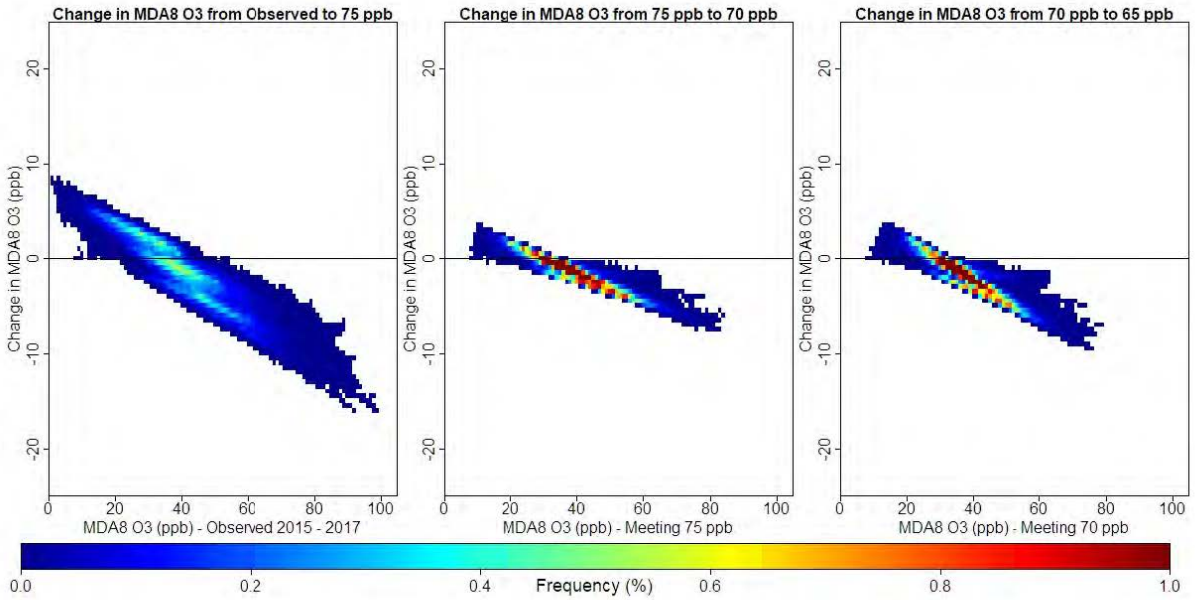



**Figure 3C-89. Changes in MDA8 $O_3$ based on HDDM adjustments in the Sacramento study area.**

**Figure 3-17. Figure 3C-89 from EPA (2022).**

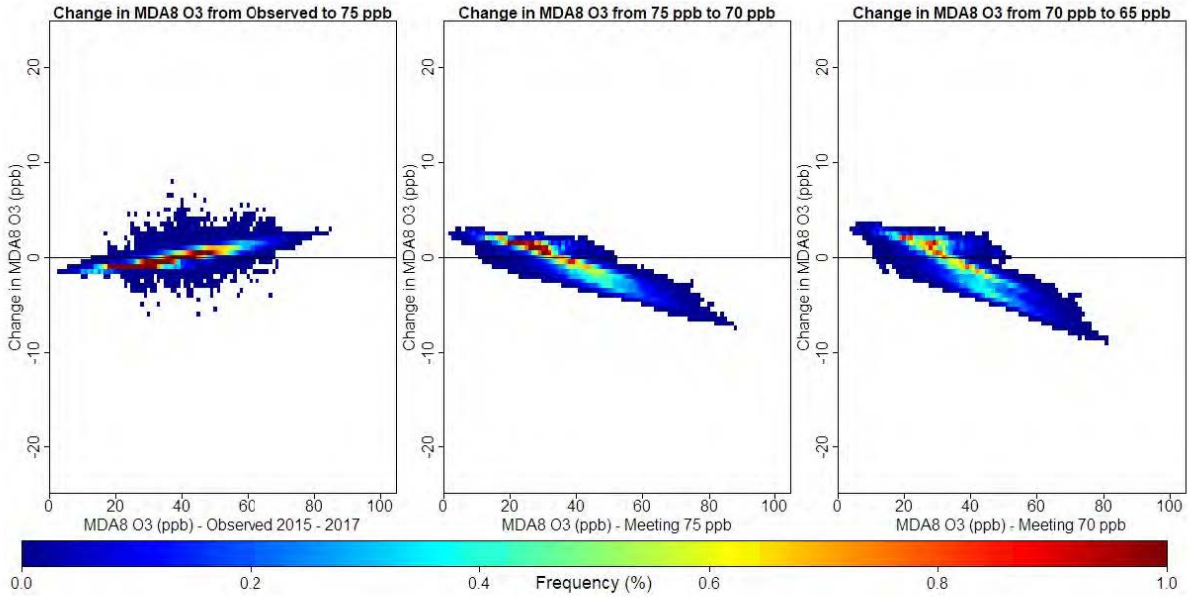



**Figure 3C-90. Changes in MDA8 $O_3$ based on HDDM adjustments in the St. Louis study area.**

**Figure 3-18. Figure 3C-90 from EPA (2022).**

Annual and summer mean and median hourly $O_3$ 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 $O_3$ 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 4$^{th}$ 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 $O_3$ 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.

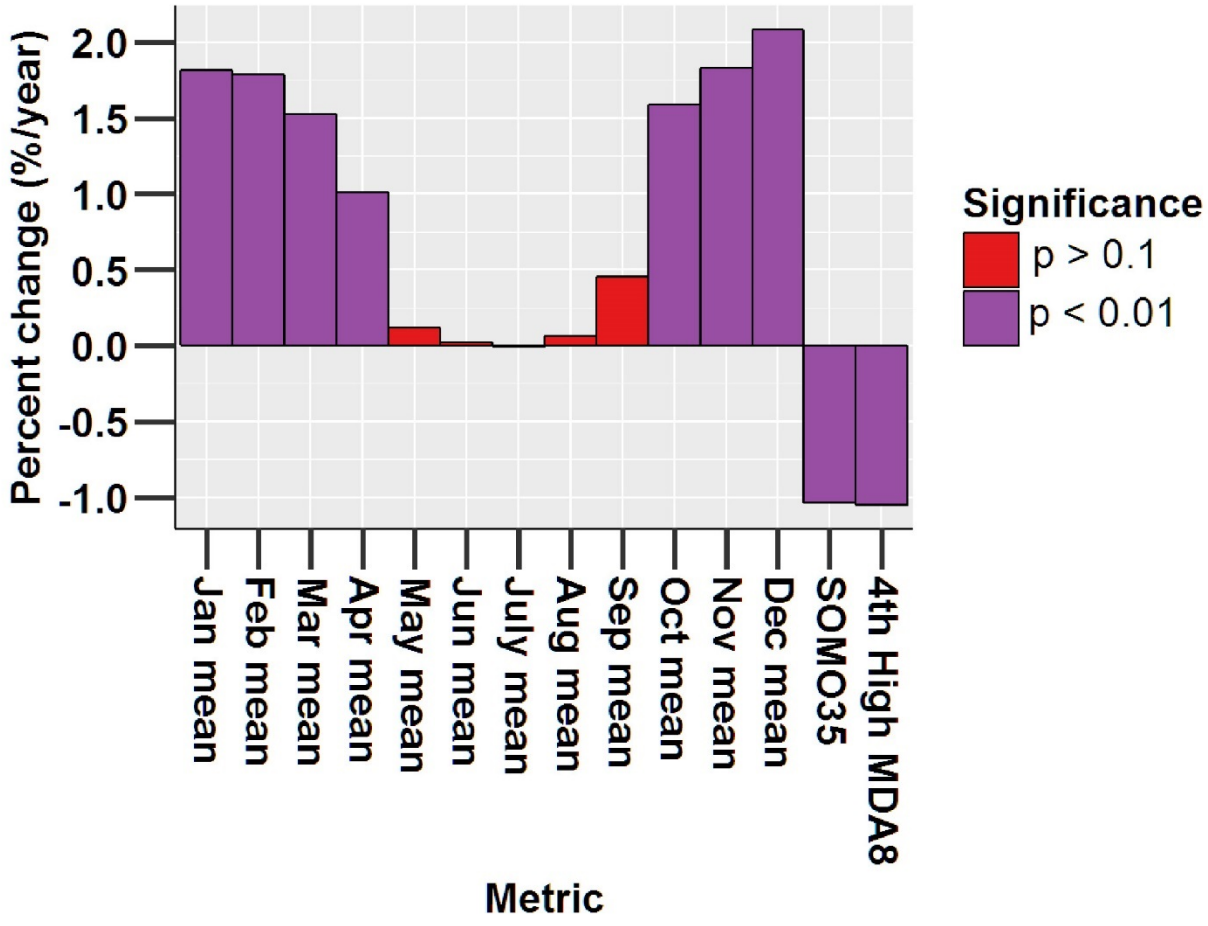


**Figure 3-19. The Theil-Sen (%/year) trend in monthly average $O_3$ levels and the annual SOMO35 and 4$^{th}$ highest MDA8 human health metrics (A4MDA8) for a suburban site for 1980-2013 in Philadelphia, Pennsylvania (EPA AQS ID: 421010024-1). The $p < 0.05$ value was used to determine significance using the Mann-Kendall test. (Source: Lefohn et al., 2018).**

Wells et al. (2021) noted that studies have shown that reductions in peak $O_3$ concentrations achieved by reducing precursor emissions often do not translate into reductions in seasonal mean $O_3$ concentrations, especially in urban areas (Lefohn et al., 2017; Simon et al., 2015). Accordingly, Wells et al. (2021) described improvements to the current method used by the EPA to adjust $O_3$ 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 $O_3$ concentrations by developing quantile regression methods for adjusting trends in the 90$^{th}$ and 98$^{th}$ percentiles of the distribution of May to September MDA8 $O_3$ concentrations.

For the period 2000 – 2018, Figs. 3-20 and 3-21 illustrate a comparison of the annual 4$^{th}$ 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 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 2016-2018, the design value was 0.111 ppm for two $O_3$ sites in San Bernardino County. In Fig. 3-20, as anticipated, the two urban sites (i.e., Simi Valley and Queens, NY) experience the highest annual 4$^{th}$ highest daily maximum 8-h average $O_3$ concentrations in comparison to the three rural National Park sites.

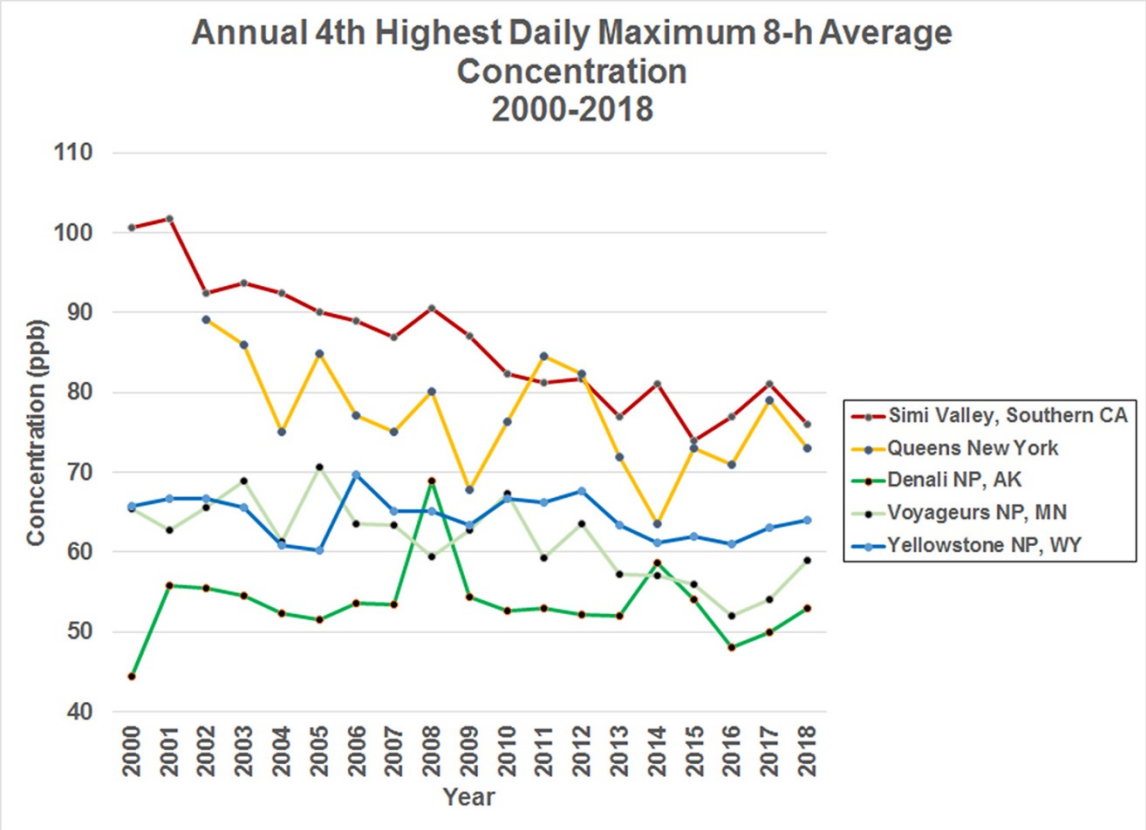



**Figure 3-20. The annual 4th highest daily maximum 8-h average $O_3$ concentration for the period 2000-2018 for Simi Valley, CA (061112002), Queens New York, NY (360810124, Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011).**

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.

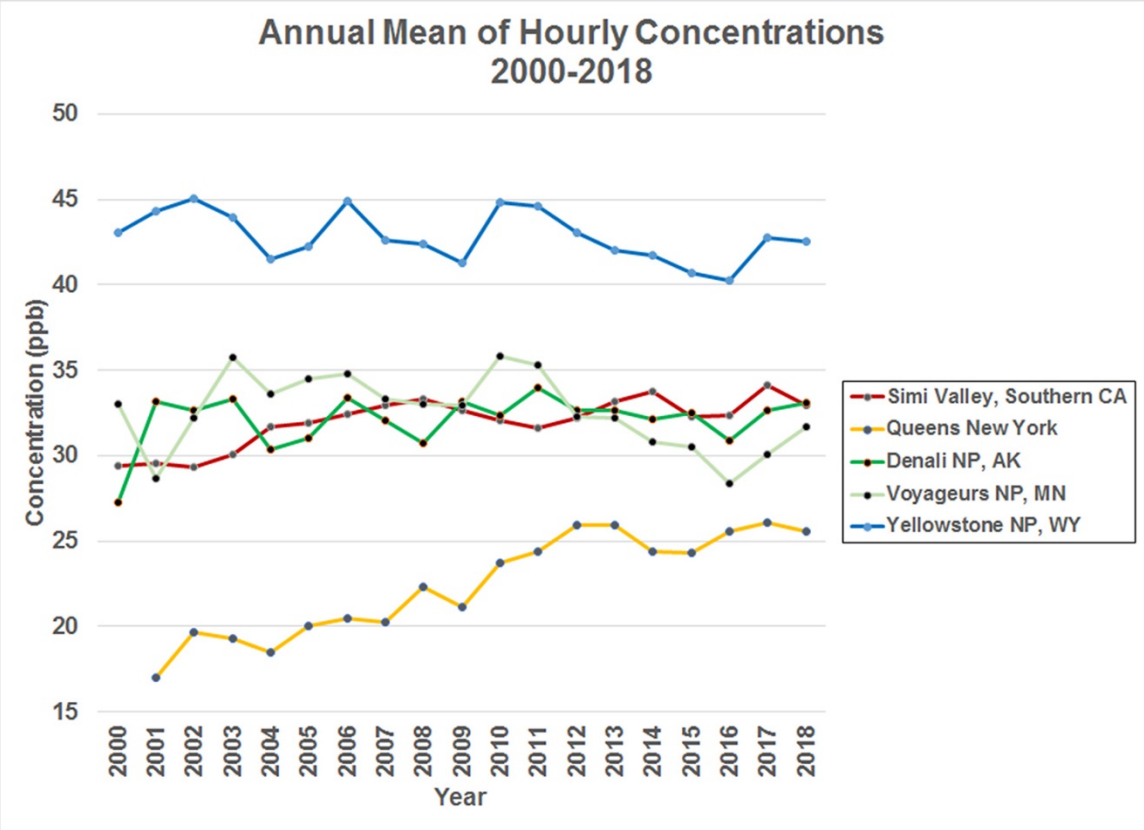



**Figure 3-21. The annual mean of the hourly average $O_3$ concentrations for the period 2000-2018 for Simi Valley, CA (061112002), Queens New York, NY (360810124, Denali National Park, AK (020680003), Voyageurs National Park, MN (271370034), and Yellowstone National Park, WY (560391011).**

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 $O_3$ monitoring sites have similar values, even though some sites experience elevated hourly $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ levels (Lefohn et al., 2001, 2011, 2012, 2014; EPA, 2014a). Fig. 3-22 illustrates for 2007 (modeled background

$O_3$ data provided by EPA) the relationship between background $O_3$ 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 $O_3$ 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 $O_3$ at specific monitoring sites. The concept of EIB will be discussed as a measure for background $O_3$ in Section 3.2. An enhanced event occurred on 2 May 2006, when a maximum hourly average $O_3$ 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 $O_3$ 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 4$^{th}$ 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 4$^{th}$ highest 8-h daily maximum concentration for the 2000-2014 period. 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 4$^{th}$ highest 8-h daily maximum concentration, the entire 24-h period is required to capture the influence of stratospheric events that enhance $O_3$ 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 $O_3$ (i.e., $USB_{AB}$) appears to play a predominant role in influencing the observed ambient levels of $O_3$.

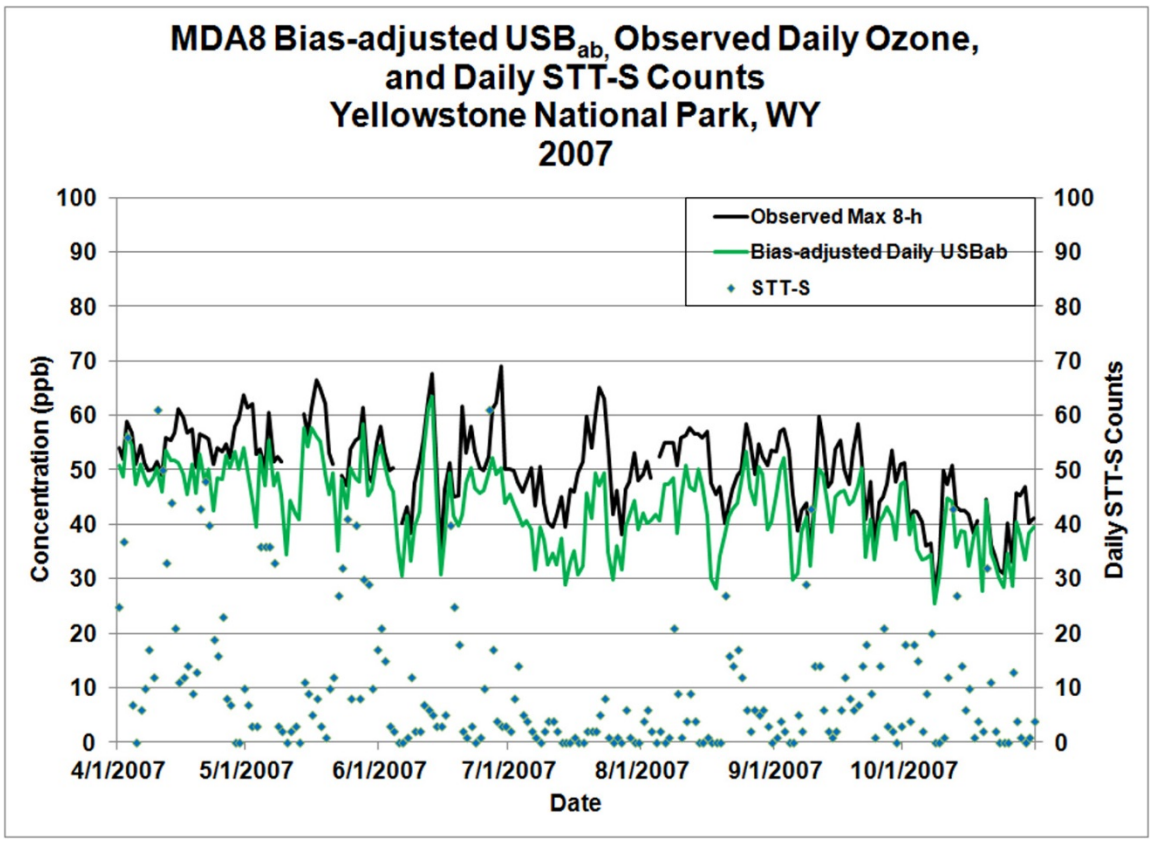



**Figure 3-22. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for Yellowstone National Park (AQS ID 560391011) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

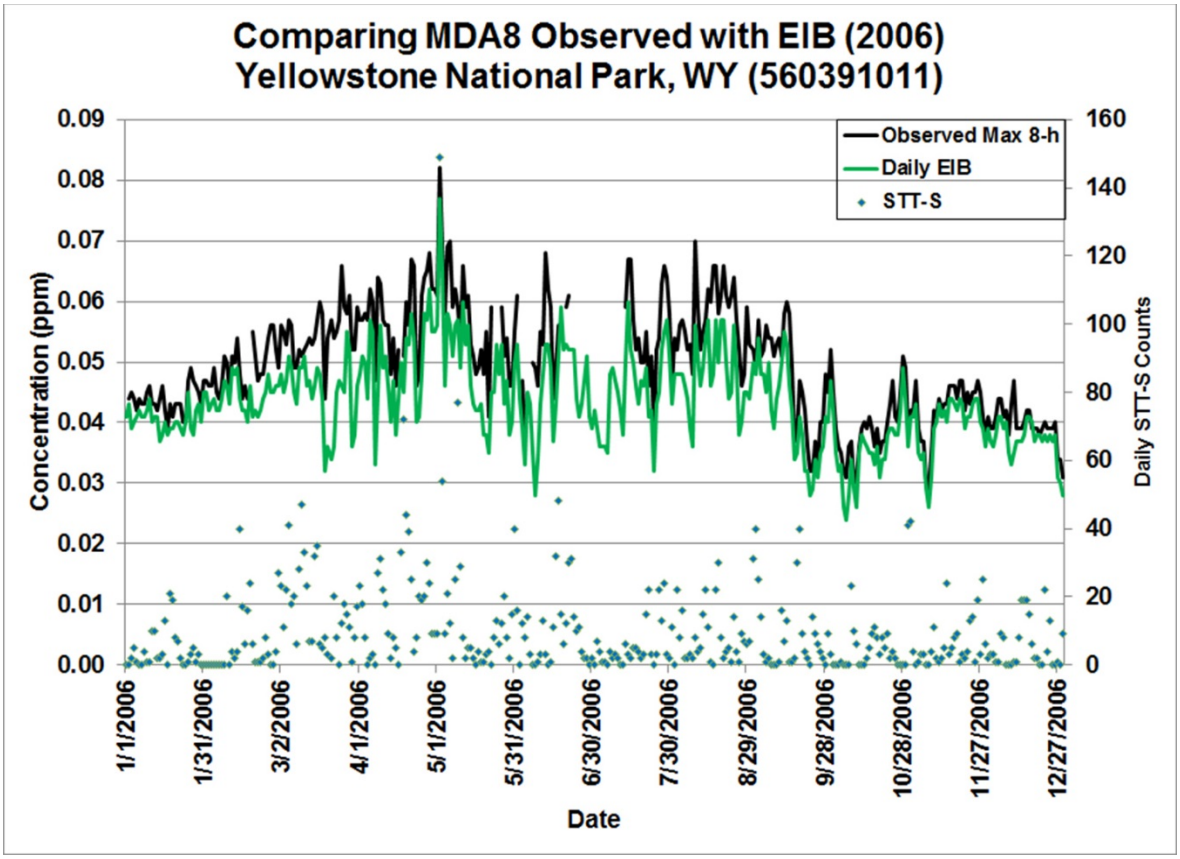



**Figure 3-23. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted Emissions Influenced Background (EIB) 8-h daily maximum concentrations for Yellowstone National Park (AQS ID 560391011) for January-December 2006. The daily stratospheric-tropospheric transport 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 $O_3$ 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 AQS data, information from the TOAR database (see Schultz et al., 2017) indicates that of 406 U.S. sites that reported both annual and 4$^{th}$ 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 for trends. Thus, a substantial number of the AQS sites show increasing trends using the annual average concentration metric.

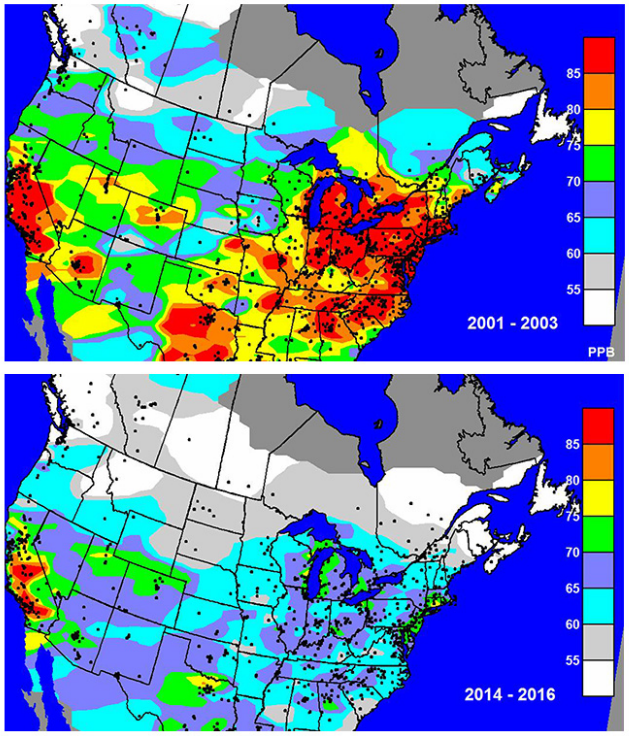
The calculation of the annual mean concentration includes the low, middle, and high hourly average concentrations. Lefohn et al. (1998), Simon et al. (2015) and Lefohn et al. (2017, 2018) have discussed the effects of $NO_x$ emissions on hourly average $O_3$ concentrations. As pointed out in the draft PA (EPA, 2022), 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 $O_3$. Simon et al. (2015) discussed the effects of reducing $O_3$ precursors in the United States on $O_3$ concentrations. Using daily 8-h average concentrations, the authors reported that decreasing $O_3$ trends generally occurred in the summer, in less urbanized areas, and at the upper end of the $O_3$ distribution (i.e., the higher 8-h concentrations). Conversely, ***increasing*** $O_3$ trends generally occurred in the winter, in more urbanized areas, and ***at the lower end of the $O_3$ 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 $O_3$ 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 $O_3$ titration by NO. Based on this, one would anticipate that as emissions were reduced, that annual $O_3$ 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 (Norther Hemisphere) and fall (Southern Hemisphere) resulted in $O_3$ increases when averaging metrics were used and $O_3$ decreases when MDA8 metrics were applied.

During the 2015 and 2020 $O_3$ 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 $O_3$ concentrations, the potential for chronic and acute health and vegetation effects are reduced (Federal Register, 2015, 2020).

### 3.1.2 Patterns of the Changes in Which Months the Highest $O_3$ Concentrations Occur as Emissions are Reduced

In the U.S., we have experienced significant reductions in $O_3$ levels. Figure 3-24 below compares the 3-year average of the annual 4$^{th}$ highest 8-h value between 2001-2003 with 2014-2016. As a result of emission reductions to attain reductions in $O_3$ exposures, important changes have occurred in the months when the highest $O_3$ concentrations are observed. Using models, Figs. 3-25 to 3-32 (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 draft PA) but for monthly rather than diurnal distributions. The figures illustrate the modeling results presented in the draft PA document (EPA, 2022) 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 4$^{th}$ highest 8-h value for the period 2001-2003 with 2014-2016.**

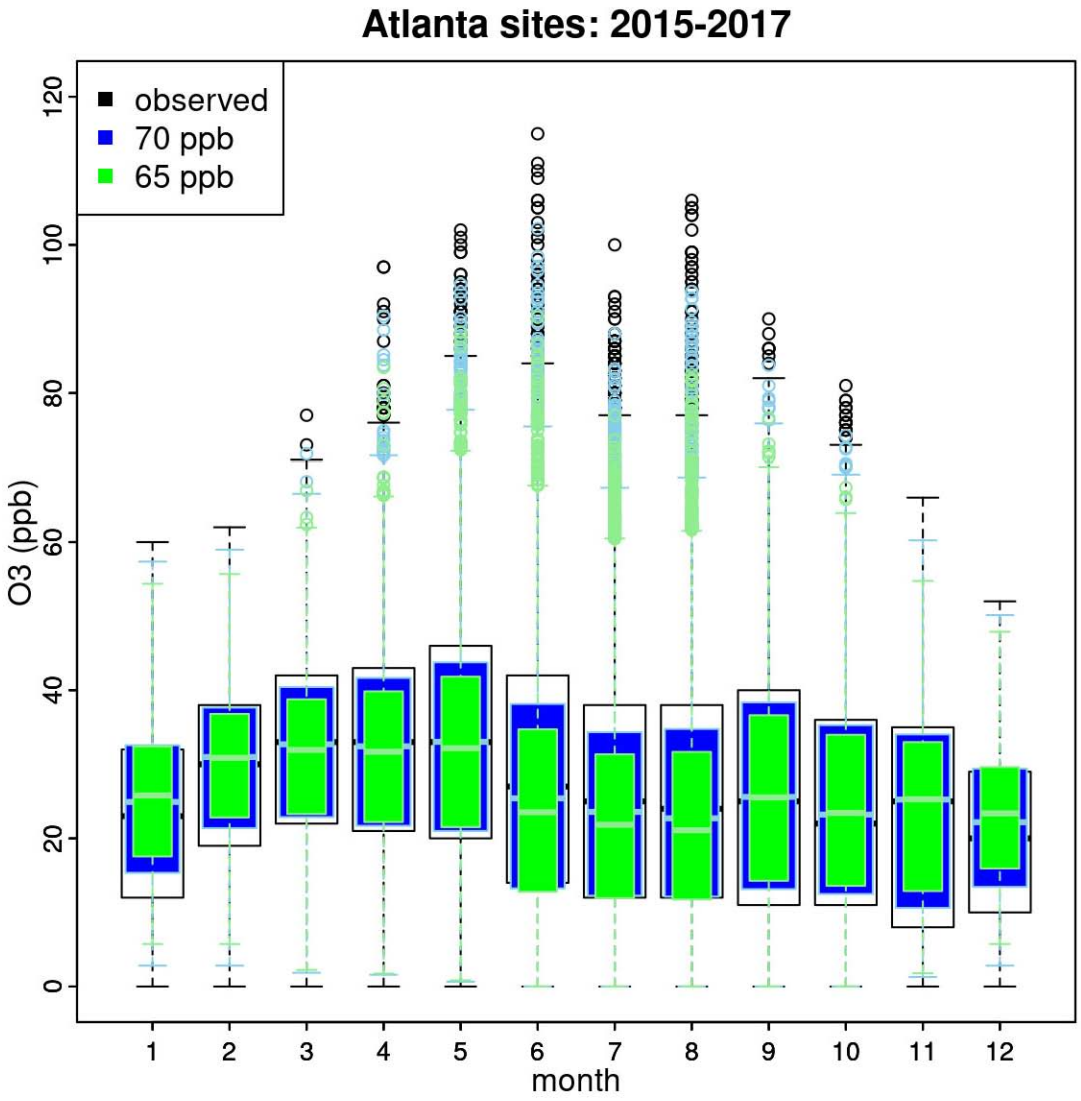



**Figure 3C-75. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Atlanta.**

**Figure 3-25. Figure 3C-75 from EPA (2022).**

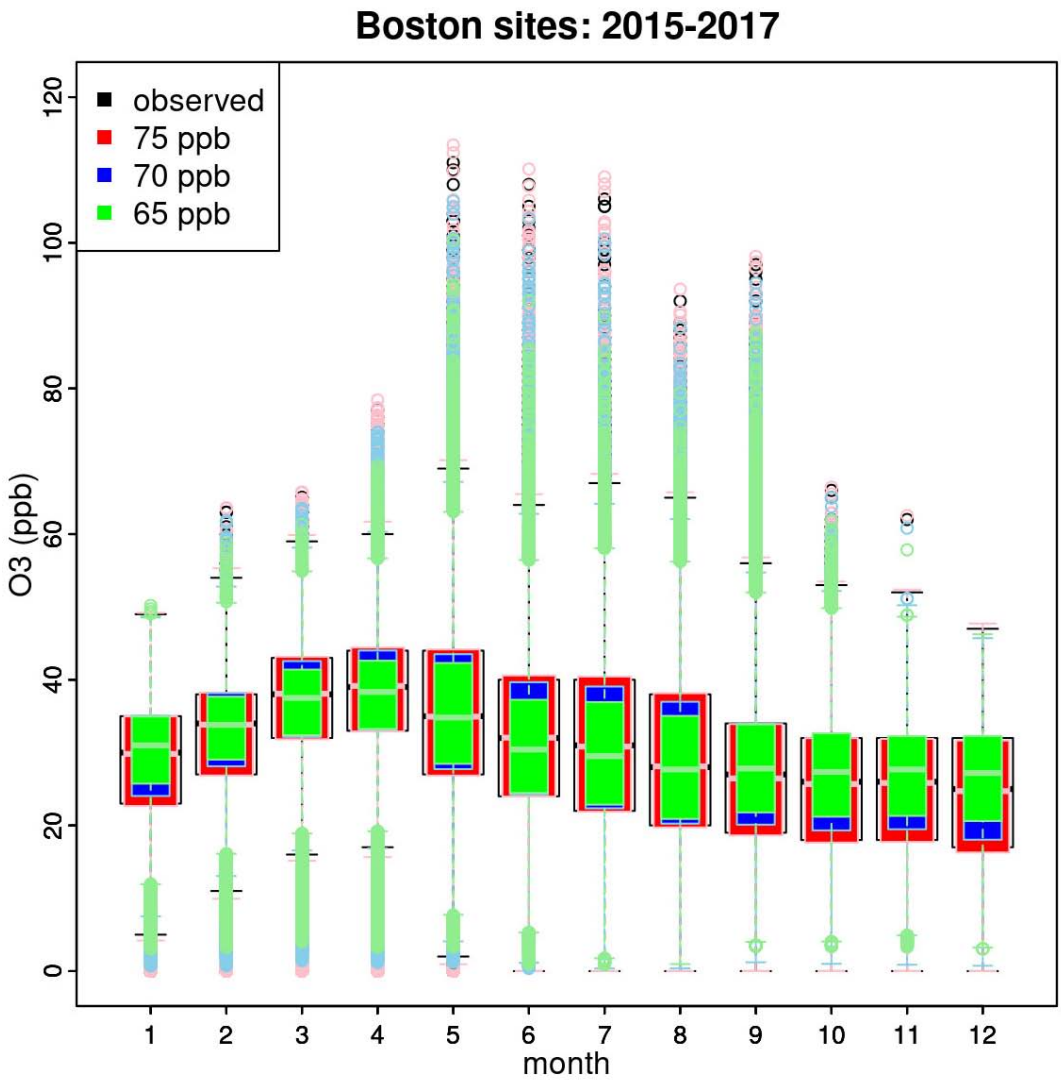



**Figure 3C-76. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Boston.**

**Figure 3-26. Figure 3C-76 from EPA (2022).**

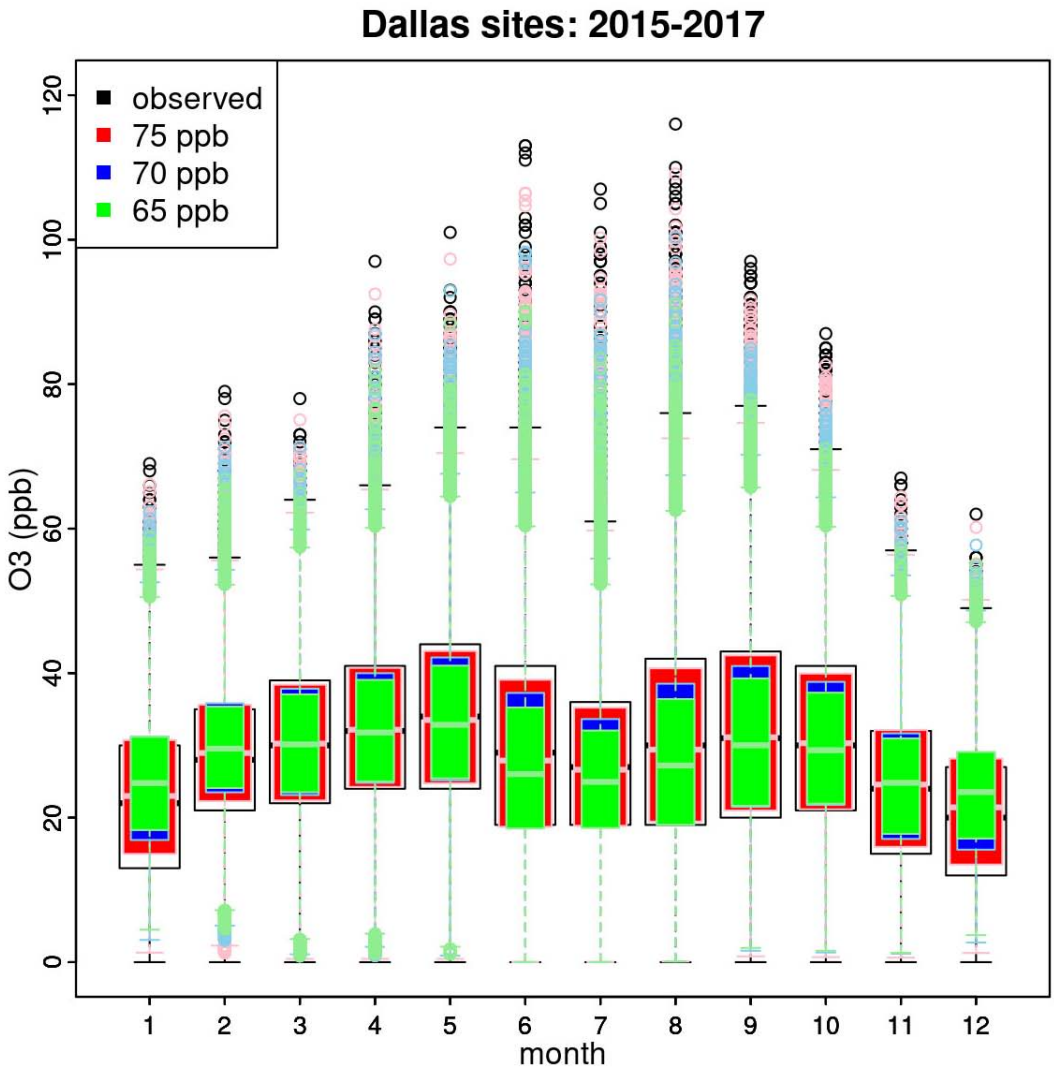



**Figure 3C-77. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Dallas.**

**Figure 3-27. Figure 3C-77 from EPA (2022).**

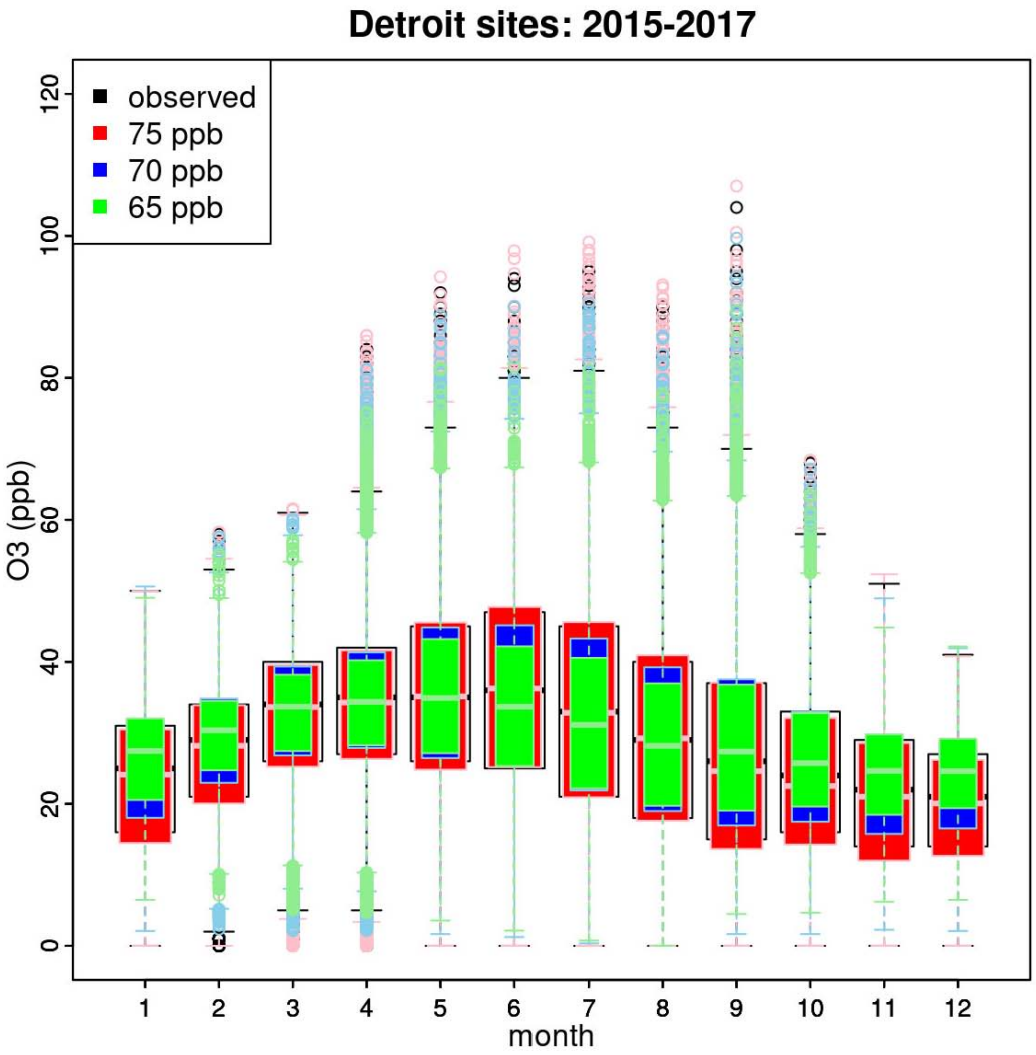



**Figure 3C-78. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Detroit.**

**Figure 3-28. Figure 3C-78 from EPA (2022).**

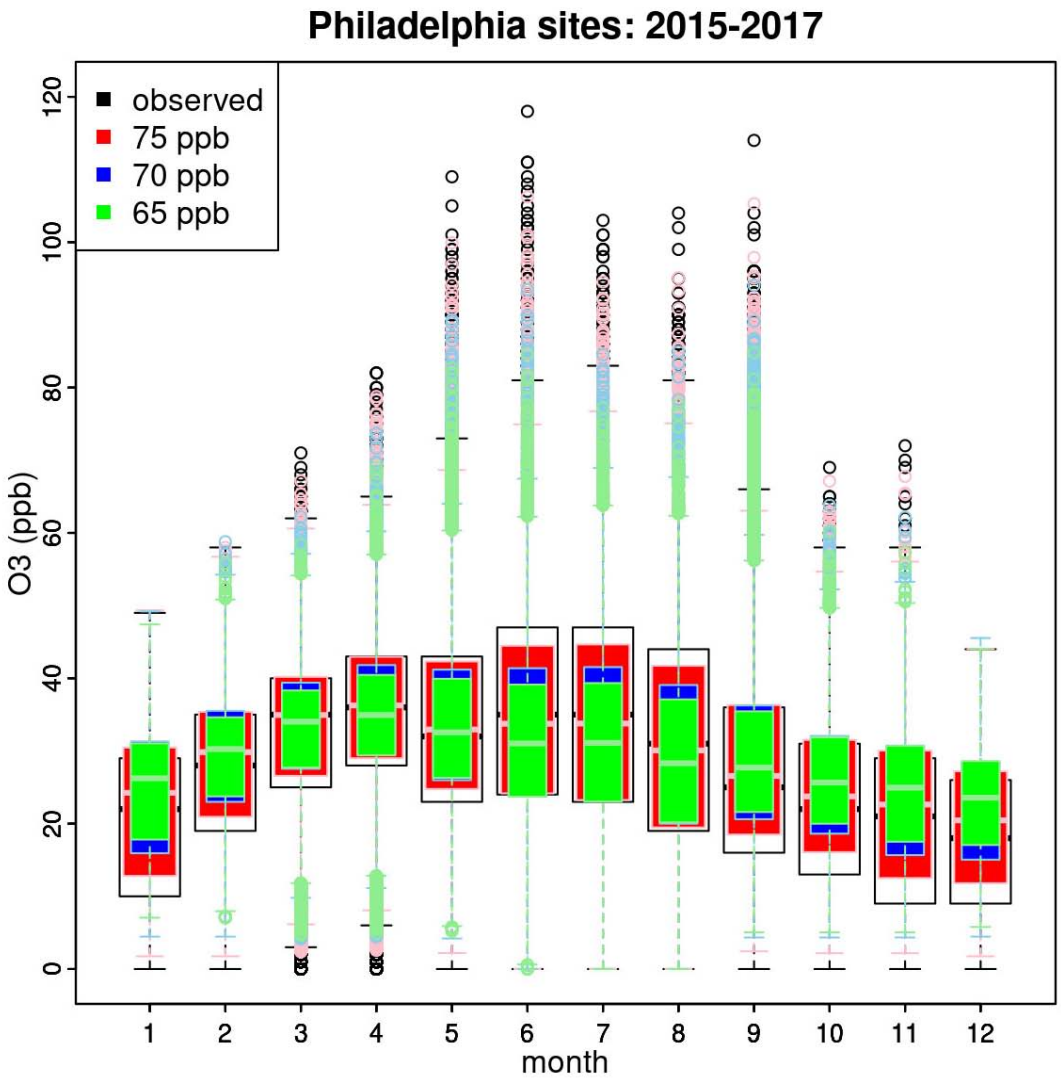



**Figure 3C-79. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Philadelphia.**

**Figure 3-29. Figure 3C-79 from EPA (2022).**

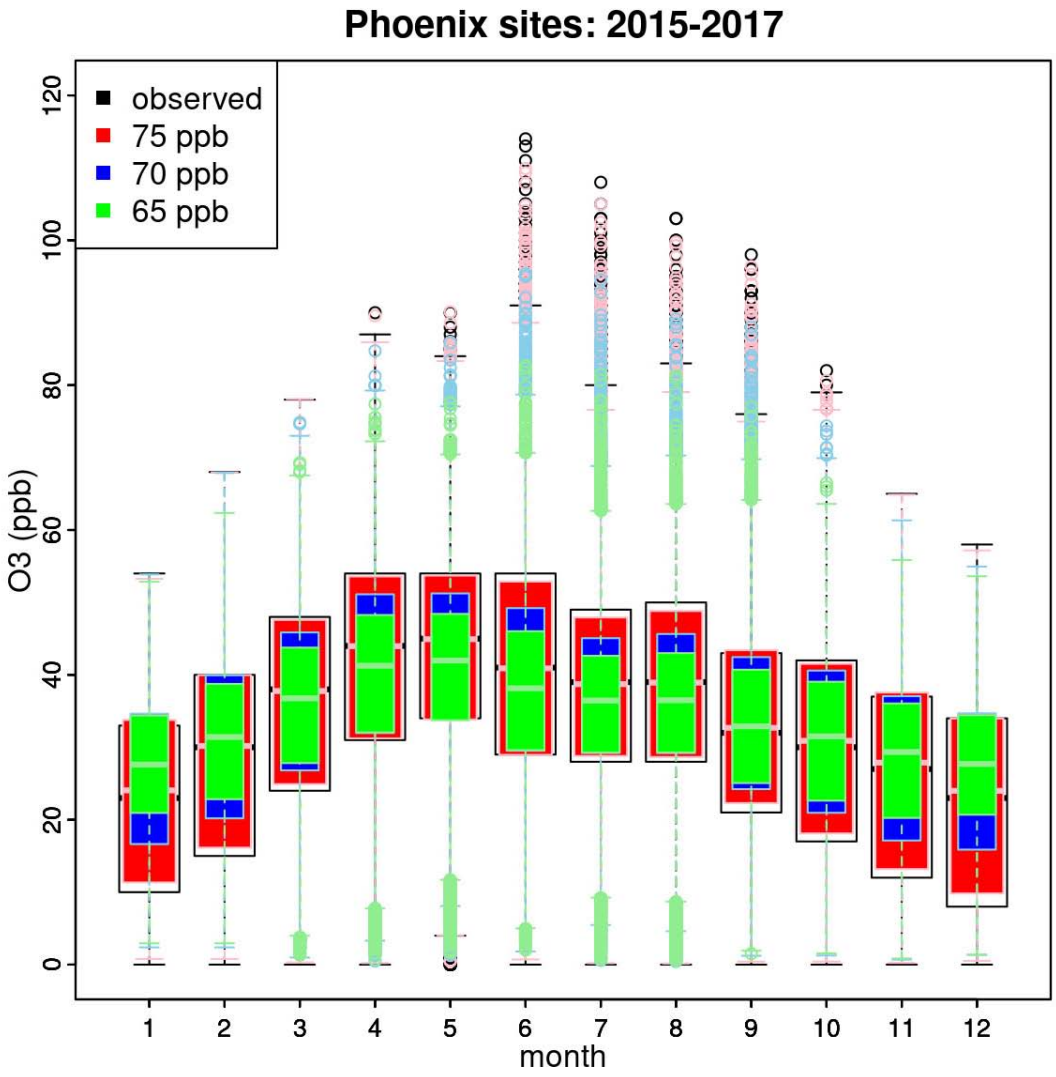



**Figure 3C-80. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Phoenix.**

**Figure 3-30. Figure 3C-80 from EPA (2022).**

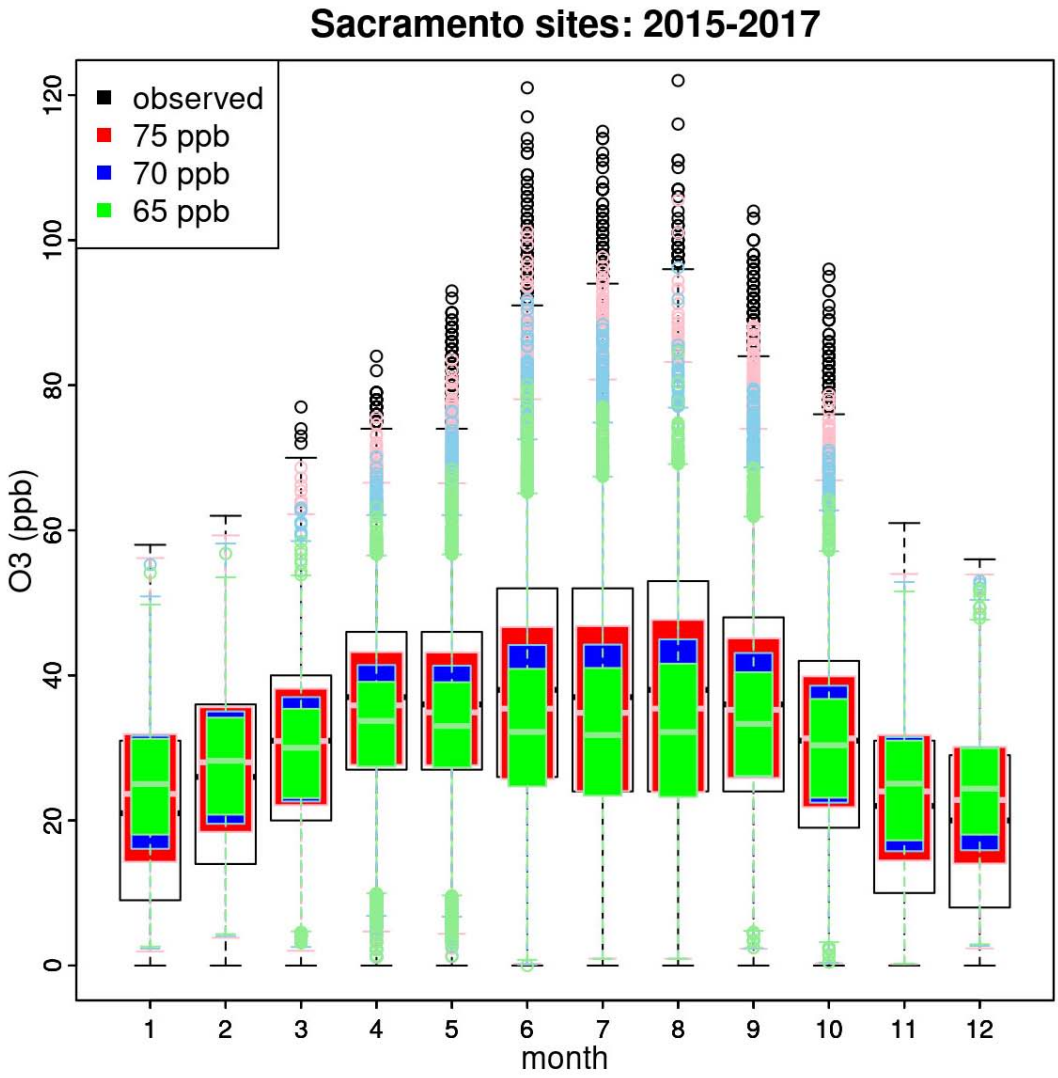



**Figure 3C-81. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in Sacramento.**

**Figure 3-31. Figure 3C-81 from EPA (2022).**

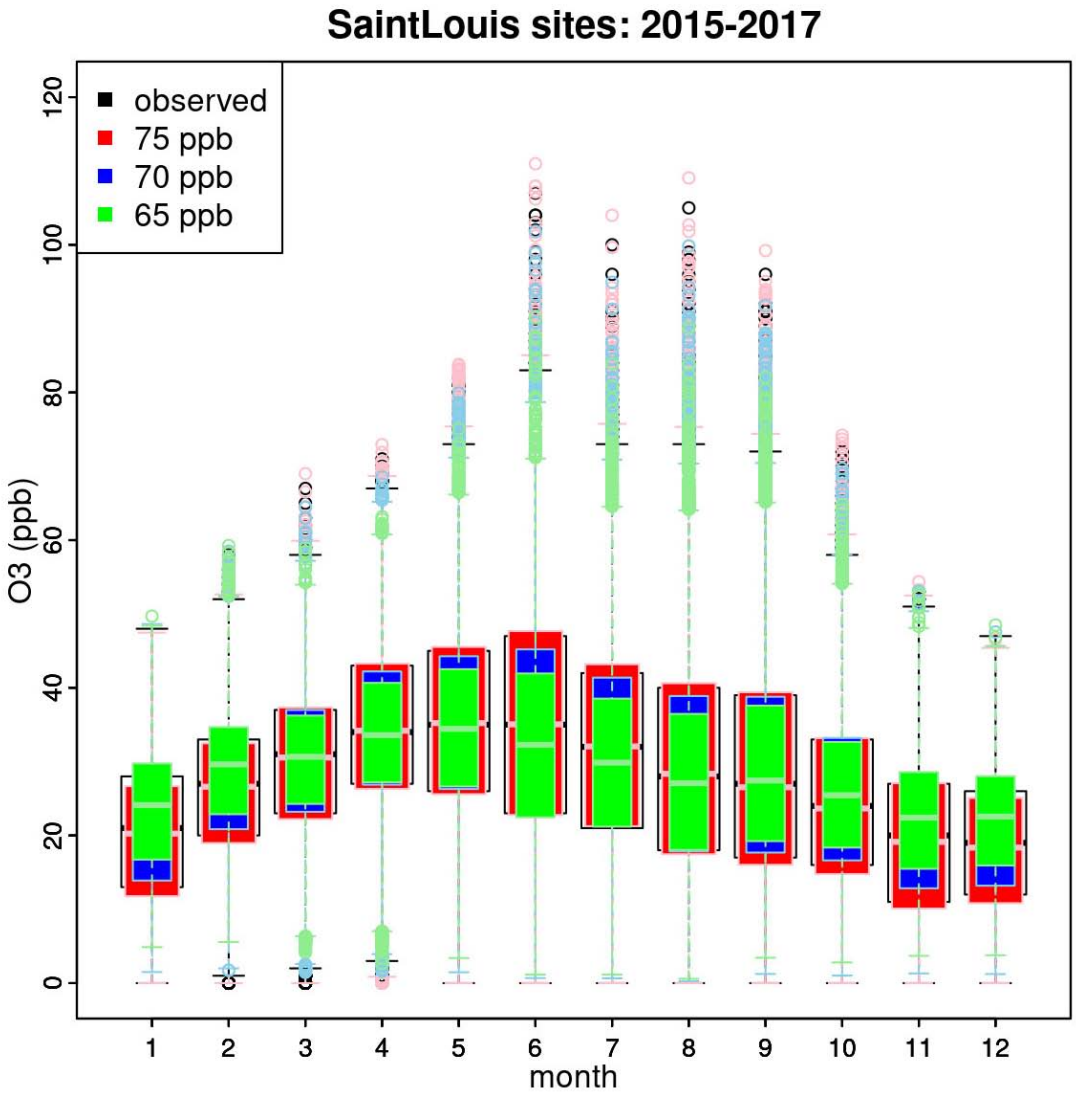



**Figure 3C-82. Monthly distribution of hourly $O_3$ concentrations at monitoring sites in St. Louis.**

**Figure 3-32. Figure 3C-82 from EPA (2022).**

Using its models, for the monthly plots for the 8 cities, the EPA notes in the draft PA (EPA, 2022, 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 $O_3$ days. This is due to more $O_3$ decreases during summer months and more $O_3$ increases in winter months. The $O_3$ 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 $O_3$ which can both create $O_3$ through photochemical pathways and destroy $O_3$ through titration. In addition, the decreases on the highest $O_3$ days and increases on the lowest $O_3$ days show a visible compression of the $O_3$ distribution in these plots, similar to what was seen in the diurnal plots.

The modeling results showed changes for midrange $O_3$ days for a pattern of shifting higher mid-range $O_3$ from the summer months to earlier in the year. While in most cities, the highest interquartile $O_3$ concentrations in the recent conditions occur in the summer months, in many areas the highest interquartile $O_3$ concentrations shift to spring months (April-May) for the adjustment scenarios. This pattern can be seen 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 $O_3$ monitoring sites. Actual data do show that the highest $O_3$ exposures occur at some sites across the U.S. during the springtime (March to mid-June). Using hourly average $O_3$ 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 $O_3$ monitors for the period 2006-2010. Table 7A-2 is provided in the pages below. Note that several of the $O_3$ 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 $O_3$ concentrations ($\geq$ 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 $O_3$ at specific monitoring sites. It is important to note that the analysis of stratospheric intrusions 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 $O_3$ concentration within the intrusion. The $O_3$ 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. Thus, Lefohn et al. (2011) chose the coincidence table approach that summarized the frequency of daily intrusions and the daily maximum hourly average $O_3$ concentrations and 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 $O_3$ 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 $SO_2$, $O_3$, and $NO_x$. Neufeld et al. (2019) analyzed $O_3$ 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 draft PA (EPA, 2022), 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. 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 $O_3$ 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 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 $O_3$ 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 $O_3$ exposures at the high-elevation Yellowstone National Park site may occur during the spring months, STT-S enhanced $O_3$ levels exist throughout the entire year.

**Table 7A-2 Ozone Exposure in 57 $O_3$ 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

7A-13

Source: EPA (2014c).

## 3.2 Background Ozone

### 3.2.1 Why is the Quantification and Spatial Distribution of Background $O_3$ Important?

The EPA (2015) recognized that periodically, in some locations in the U.S., sources other than domestic manmade emissions of $O_3$ precursors can contribute appreciably to monitored $O_3$ concentrations. EPA (2015) clarified the specific definition of background $O_3$ that the Agency wished to use and the sources and processes that led to background $O_3$ across the U.S.

The EPA revised the primary $O_3$ 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 $O_3$ concentrations in some areas. However, the EPA considered the extent and importance of background $O_3$ throughout the NAAQS review process. EPA (2015) noted that Agency considered background $O_3$ to be any $O_3$ formed from sources or processes other than U.S. manmade emissions of nitrogen oxides ($NO_x$), volatile organic compounds (VOC), methane ($CH_4$), and carbon monoxide (CO). This definition of background was specifically referred to as U.S. background (USB). It is important to recognize that USB does not include intrastate or interstate transport of manmade $O_3$, which can also influence $O_3$ 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, $O_3$ can have an atmospheric lifetime on the order of weeks. As a result, background $O_3$, and to a lesser extent background $O_3$ 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 $O_3$ originates from production of $O_3$ in the stratosphere through interactions between ultraviolet light and molecular oxygen. $O_3$ 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 $O_3$ concentrations. These events are called stratospheric intrusions and can result in relatively high USB levels of $O_3$ at the surface, especially at higher-elevation sites. Other natural sources of $O_3$ precursor emissions include wildfires, lightning, and vegetation. Biogenic emissions of methane, which can be chemically converted to $O_3$ over relatively long time (sic) scales, can also contribute to USB $O_3$ levels. Finally, manmade precursor emissions from other countries can contribute to the global burden of $O_3$ in the troposphere and to increased USB $O_3$ levels.
>
> USB $O_3$ levels can vary considerably in space and time. When assessing USB $O_3$ 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 $O_3$ days when concentrations may approach or exceed the NAAQS.

In the draft PA (EPA, 2022, page 2-28), the authors note:

> In this reconsideration, as in past reviews, the EPA generally characterizes $O_3$ concentrations that would exist in the absence of U.S. anthropogenic emissions as U.S. background (USB). An alternative phrasing for USB is the $O_3$ concentrations created collectively from global natural sources and from anthropogenic sources existing outside of the U.S. Such a definition helps distinguish the $O_3$ that can be controlled by precursor emissions reductions within the U.S. from $O_3$ 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 $O_3$ levels by assessing the remaining $O_3$ 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 $O_3$ levels. Another modeling technique, referred to as "source apportionment" modeling, can also be used to estimate the sources that contribute to modeled $O_3$ concentrations. This approach estimates the contribution of certain source categories (e.g., natural sources, non-U.S. manmade sources) to modeled $O_3$ at each model grid cell on an hourly basis.

A great deal of the focus by EPA in its background $O_3$ 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 $O_3$ have found that USB concentrations are relatively constant with increasing total $O_3$ concentration, indicating that days with higher $O_3$ 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 $O_3$ concentration as the most relevant for discussing USB $O_3$, 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 $O_3$ in contributing to the highest $O_3$ 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 $O_3$ 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 $O_3$ concentrations have been generally predicted to be similar to or smaller than seasonal mean USB $O_3$ 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 $O_3$ 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 $O_3$ concentration.

Further, in the draft PA (2022, page 2-67), the Agency's focus on USB on days when high $O_3$ concentrations occur continues:

- 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.

- The USA contributions that drive predicted MDA8 total $O_3$ 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 $O_3$ PA prepared for the 2015 review (2014 PA, Section 2.4).

From an air quality management perspective, the EPA's focus on the contribution of background $O_3$ during periods when elevated $O_3$ levels occur appears to be reasonable. Hogrefe et al. (2020) note that

> Viewed from an air quality management perspective, the magnitude of $O_3$ formed from locally controllable anthropogenic precursor emissions relative to background ozone (BG $O_3$) levels—that is, $O_3$ formed from non-local anthropogenic precursor emission sources and local and non-local natural sources, regardless of where the $O_3$ formation occurs—has generally decreased over time. This increases emphasis on representing the processes controlling BG $O_3$ and characterizing its temporal and spatial fluctuations, especially for $O_3$ exceedance events. The latter is particularly important because although the relative magnitude of typical average BG $O_3$ concentrations with respect to the NAAQS has increased, BG $O_3$ 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 $O_3$ sources, together with year-to-year variability in stratospheric influence will leave little margin for $O_3$

produced from local and regional emissions, posing challenges to achieving a potentially tightened $O_3$ NAAQS in the southwestern U.S.

While the EPA appears to be mainly focused on how much of current $O_3$ 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 $O_3$. 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. For the vegetation related W126 exposure index, which is not an average exposure metric as is incorrectly defined in the draft PA in footnote 29 (EPA, page 2-32), hourly average concentrations are weighted and accumulated over a specific period for assessing vegetation effects. The draft PA (EPA, page 4-2), correctly defines the W126 exposure metric in footnote one as follows:

> The W126 index is a cumulative seasonal metric described as the sigmoidally weighted sum of all hourly $O_3$ concentrations during a specified daily and seasonal time window, with each hourly $O_3$ 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).

For assessing vegetation risk, hourly average background $O_3$ 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 $O_3$ concentrations, which contribute to the estimated human health risk. As pointed out by Langford et al. (2022) and Li et al. (2020), background $O_3$ sources, including the year-to-year variability in stratospheric influence, play an important role in influencing $O_3$ levels in the southwestern U.S.

Background $O_3$ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total $O_3$ levels and the lower and mid-level concentrations influence mortality and morbidity risk estimates. It is important to quantify the importance of background $O_3$ 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 ($dFEV_1$ > 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 ($dFEV_1$ > 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 $O_3$ along the full range of $O_3$ concentrations (page 9-30

of the 2014 Health REA (EPA, 2014b)). As explained by the Agency (EPA, 2014b), because $O_3$ in the lower concentration range may shift upward as the result of NOx 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 $O_3$ concentrations). Fig. 3-34 illustrates the percent of short-term mortality attributable to $O_3$ 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 Heath 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 $O_3$ will predominate. Fig. 3-35 illustrates the contribution of background to ambient levels of $O_3$ 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 $O_3$ 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 $O_3$ concentrations will be expected to increase their domination of the cumulative mortality health risk estimates.

Background $O_3$ concentrations become more and more important in influencing ambient levels as emission reductions are implemented. It is recognized by the Courts that NAAQS $O_3$ levels are set to protect public health and welfare and that background $O_3$ is not a consideration in setting these levels. In the draft PA (EPA, 2022, page 1-11), the EPA notes the following:

> In the August 2019 decision, the court additionally addressed arguments regarding considerations of background $O_3$ concentrations, and socioeconomic and energy impacts. With regard to the former, the court rejected the argument that the EPA was required to take background $O_3$ 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 $O_3$ levels in any part of the country exceeded the level of $O_3$ 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 $O_3$ (id. at 624).

However, for practical purposes, if the standard were to be set at a level that ambient $O_3$ levels consisted of almost all background $O_3$, 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 draft PA (EPA, 2022, page 2-52), when episodic natural events contribute to elevated $O_3$ 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 $O_3$ 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 $O_3$ concentrations reported for specific sites in their states be considered under the Exceptional Events Rule.

While background $O_3$ currently is not a consideration in the setting of the level of the $O_3$ standard, background $O_3$ is an important consideration for assessing human health effects risks. The risk assessments play an important role in the margin of safety determinations. Background $O_3$ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total ambient $O_3$ 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 $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022). Because background $O_3$ estimates for the 8 cities were not provided in the draft PA, no information is available at this time in the current $O_3$ NAAQS reconsideration rulemaking process for assessing the contribution of background $O_3$ to the human health effects risks. This information is important in margin of safety considerations for the setting of the NAAQS for $O_3$.

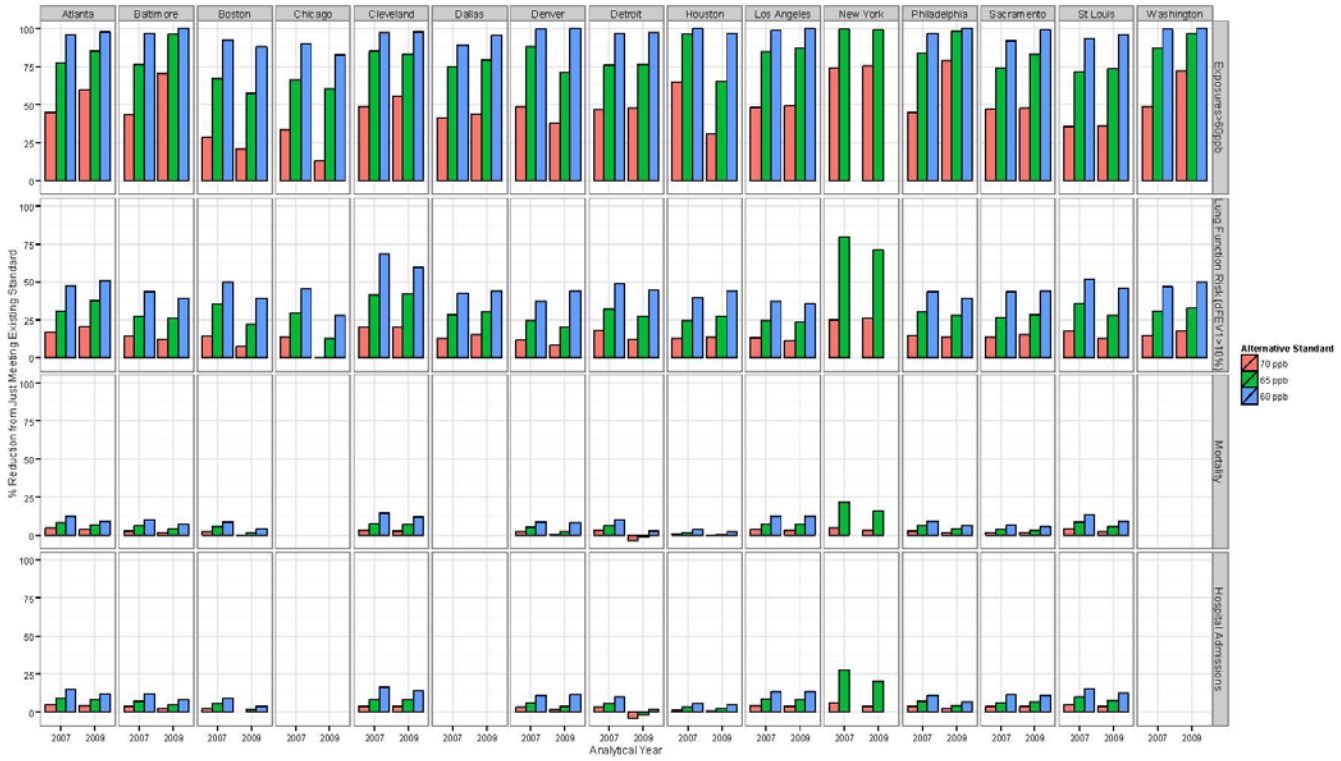



**Figure 9-8. Comparison of the Percent Reduction in Key Risk Metrics for Alternative Standard Levels Relative to Just Meeting the Existing 75 ppb Standard.**

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

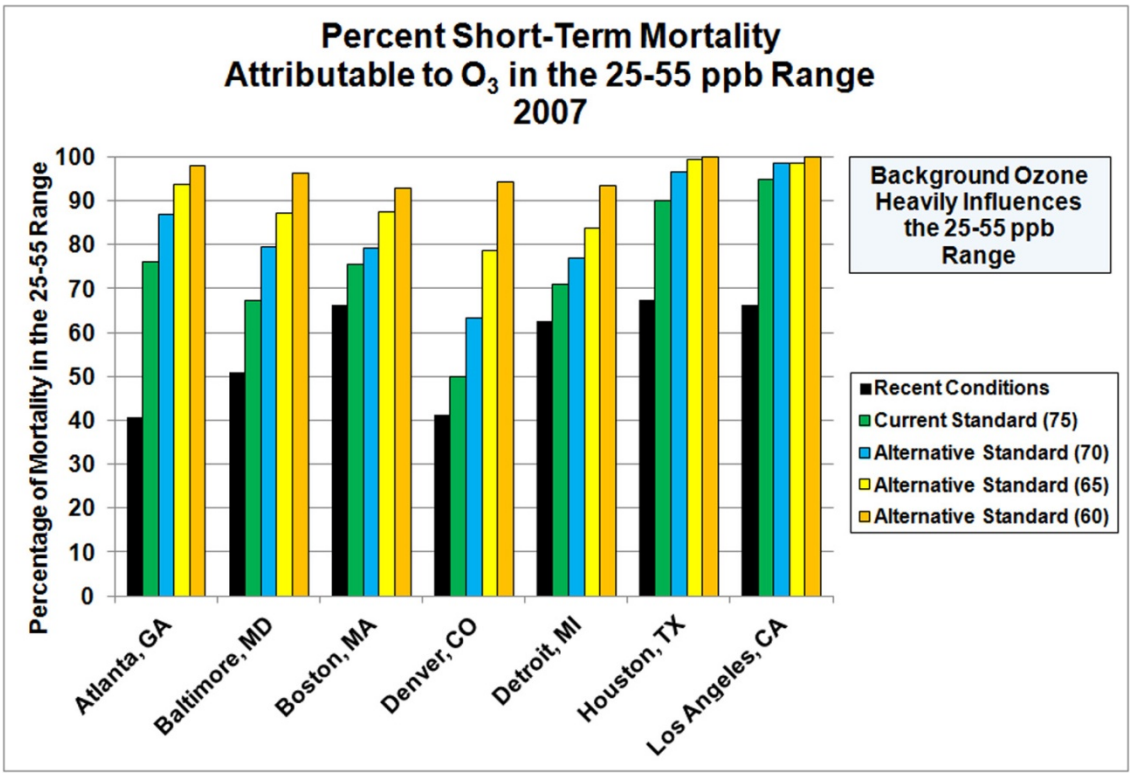



**Figure 3-34. Percent short-term ozone-attributable mortality in the 25-55 ppb range for various exposure conditions for 2007 for 7 of 12 cities. (Source: Data from Fig. 7-B1 on page 7-B3 of EPA, 2014d).**

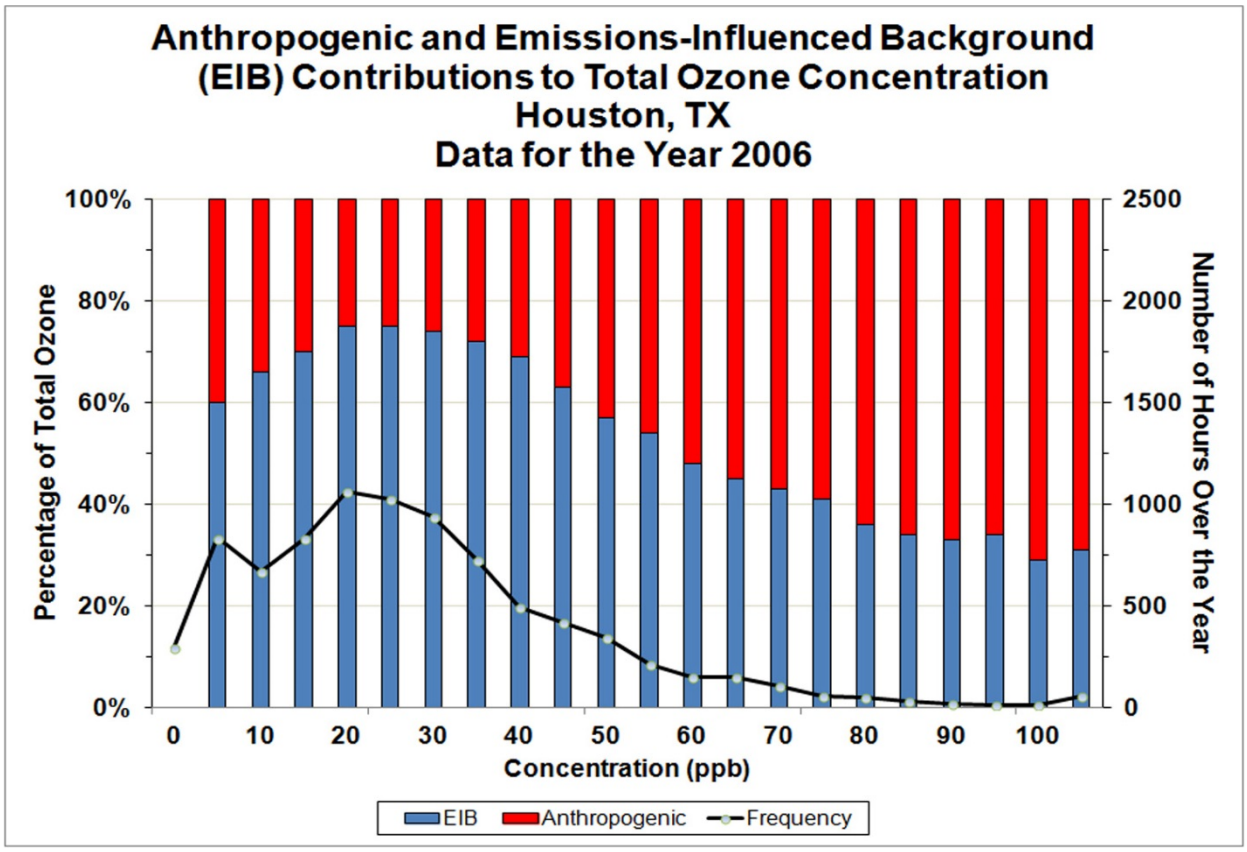



**Figure 3-35. Binned (5 ppb) frequency distribution of observed hourly total $O_3$ (black curve; right axis) and average relative binned contributions of hourly maximum Emissions Influenced Background (EIB) and anthropogenic $O_3$ (bars; left axis) for Houston, TX (AQS ID 482010055) in 2006. Lefohn et al. (2014).**

### 3.2.2 Background $O_3$ 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 $O_3$ can be attributed to sources other than U.S. anthropogenic sources?** While the Agency is focused on how much of the current $O_3$ can be attributed to sources other than U.S. anthropogenic sources on days when ambient levels exceed the standard, as discussed previously, quantifying background $O_3$ is also important for assessing human health and ecological effects risks and how background $O_3$ potentially influences the margin of safety for the setting of the $O_3$ NAAQS.

While background $O_3$ cannot be measured directly, estimating it accurately by utilizing both empirical and modeling approaches is important. Under a variety of meteorological conditions, background $O_3$ 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 $O_3$ is of interest because (1) at times background $O_3$ is associated with high concentrations experienced in the U.S. Intermountain West that affect attainability of $O_3$ air quality standards (Lefohn et al., 2001; Langford et al., 2009; McDonald-Buller et al., 2011; Lin et al., 2012; Dolwick et al., 2015) and (2) background $O_3$ contributes on a continuous basis to observed concentrations that influence human health and vegetation risk estimates, whose values influence recommended levels for Federal $O_3$ 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 $O_3$ primary standard, elevated background concentrations associated with stratospheric intrusions can affect vegetation (Skelly, 2000; FLAG, 2010).

The term "background $O_3$" 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) $O_3$ (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 $O_3$ 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) $O_3$ concentrations. The level of USB $O_3$ is defined to include contributions from global anthropogenic and natural sources in the absence of U.S. anthropogenic emissions. In other words, the USB $O_3$ concentration is defined as the $O_3$ concentration that would occur ***if all U.S. anthropogenic $O_3$ precursor emissions were removed*** (EPA, 2020a, Page ES-3). The difference associated with the hypothetical estimates using models of NAB $O_3$ and USB $O_3$ is small (EPA, 2013).

Recognizing that NAB $O_3$ and USB $O_3$ are estimated background concentrations that reflect hypothetical ***zero*** anthropogenic emissions, Lefohn et al. (2014) believed it was important to define a new metric that estimated background $O_3$ levels under ***current anthropogenic emission*** conditions. The authors, using source-apportionment based modeling, referred to these background estimates as Emissions-Influenced-Background (EIB) $O_3$. The authors defined Emission-influenced Background (EIB) $O_3$ to include contributions from natural sources throughout the globe and from anthropogenic sources outside of North America. EIB $O_3$ estimates the impact of background sources, even in situations in which local $O_3$ 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 $O_3$, except that $USB_{AB}$ $O_3$ included anthropogenic sources from Canada and Mexico. Similar to EIB, $USB_{AB}$ estimated the impact of background sources, even in situations in which local $O_3$ had been influenced by U.S. anthropogenic emissions (see Dolwick et al., 2015 for further discussion).

An important advantage in estimating either EIB $O_3$ or $USB_{AB}$ background is that policymakers have an indication of (1) the relationship between current daily background levels and daily observed $O_3$ concentrations and (2) the level of $O_3$ concentration that may occur because of implementing emissions reductions strategies. For example, if EIB $O_3$ or $USB_{AB}$ $O_3$ concentrations have a large relative contribution to observed $O_3$ 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 the previous PA (EPA, 2014a), research results based on natural background were presented. Natural background $O_3$ is defined as the $O_3$ concentrations that would occur if all anthropogenic emissions were removed worldwide. Processes that contribute to natural background $O_3$ include $O_3$ transport from the stratosphere and $O_3$ 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 $O_3$ has been defined as the measured $O_3$ concentration at rural or remote sites that have not been influenced by recent, local emissions (Jaffe et al., 2018). In contrast to USB, baseline $O_3$ 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 $O_3$ can include the $O_3$ 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 $O_3$ 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 $O_3$ may not similarly impact $O_3$ in populated locations. For instance, baseline $O_3$ 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 $O_3$ measurements cannot be used as a proxy to estimate USB $O_3$ levels in urban areas. As previously described, baseline $O_3$ can include contributions from U.S. emissions. Additionally, baseline $O_3$ monitors can be very distant from urban sites, and $O_3$ 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 $O_3$ from the monitor location to populated areas at lower elevations. The ISA (EPA, 2020a) also points out that another reason why baseline $O_3$ measurements cannot be used as a proxy for USB $O_3$ 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 $O_3$ episodes that produce the highest urban $O_3$ 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 $O_3$ measurements cannot be used directly to estimate USB (zero-out methodology) $O_3$, baseline $O_3$ data are useful for evaluating the CTMs that are used to provide model estimates of USB $O_3$.

In summary, the following terms have been used in the ISA (EPA, 2020a) to describe background $O_3$:

- 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 $O_3$ 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 $O_3$ 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 $O_3$ formed from sources other than U.S. anthropogenic sources as estimated via an apportionment technique (Dolwick et al., 2015). $USB_{AB}$ $O_3$ includes anthropogenic sources from Canada and Mexico.

- Natural background $O_3$ is defined as the $O_3$ concentrations that would occur if all anthropogenic emissions were removed worldwide. Processes that contribute to natural background $O_3$ include $O_3$ transport from the stratosphere and $O_3$ formed from precursor emissions originating from wildfires, lightning, natural methane sources, plants, and other natural VOC and $NO_X$ emissions.

- Baseline $O_3$ has been defined as the measured $O_3$ 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 $O_3$ measurements cannot be used as a proxy to estimate USB $O_3$ 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 $O_3$. Reliance on atmospheric modeling for USB, as well as $USB_{AB}$ concentrations estimates, continued in the 2013 Ozone ISA (EPA, 2013). In earlier assessments, $O_3$ 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 $O_3$. In the EPA White Paper (EPA, 2015) that is referred to in the draft PA (EPA, 2022, page 2-66), the Agency noted that

> For the purposes of this white paper and the continuing discussion of background $O_3$ issues in the NAAQS implementation context, the EPA considers background $O_3$ to be any $O_3$ 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 $O_3$." 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$*** (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 | Strength: The approach is simple to implement and provides an estimate of the lowest $O_3$ levels that can be attained by eliminating all U.S. anthropogenic emissions.<br><br>Limitation: 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 $O_3$ production because of non-linearity in the chemistry. |
| Source Apportionment | How much of the current ozone can be attributed to sources other than U.S. anthropogenic sources? | Apportionment-based USB | Strength: Provides a direct estimate of the amount of $O_3$ contributed by each source category while avoiding artifacts caused by non-linearity in the chemistry.<br><br>Limitation: While this approach identifies important sources that contribute to $O_3$, it does not predict quantitatively how $O_3$ will respond to specific emissions reduction scenarios. |

**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 $O_3$ levels. Source: 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 $O_3$. 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 $O_3$ concentrations that could be achieved if U.S. anthropogenic sources were eliminated, $USB_{AB}$ is an estimate of how much $O_3$ 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 $O_3$ 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, represents 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 $O_3$ production because of nonlinearity in the chemistry. The $USB_{AB}$ approach provides a direct estimate of the amount of $O_3$ 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 $O_3$, it does not predict quantitatively how $O_3$ will respond to specific emissions reductions scenarios. However, it is reasonable to assume that as emissions were reduced with the result that current $O_3$ 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 $O_3$ levels. *In other words, background $O_3$ would make up a higher fraction of the levels observed in future $O_3$ levels achieved with emission reductions*. This is an important concept. By estimating $USB_{AB}$ concentrations, one could obtain a lower estimate of background $O_3$ and quantify the percentage contribution of background $O_3$ 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 $O_3$ 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 $O_3$ levels and the associated increasing relative importance of background $O_3$ 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 $O_3$ can be attributed to sources other than U.S. anthropogenic sources?" The $USB_{AB}$ estimate would provide a lower estimate of background $O_3$ in most cases than that provided from USB estimates.

As noted in the draft PA (EPA, 2022, page 2-38), the zero-out approach provides simplicity for interpretation and consistency purposes. However, the use of background $O_3$ 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 draft PA (EPA, 2022, page 2-38), in urban areas, the zero-out approach will estimate higher natural and USB contributions than total $O_3$ when $NO_x$ titration is present.

### 3.2.4 EPA's Conclusion on Background $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ concentration measured. In other words, if the EPA desired to quantify the percentage of background $O_3$ associated with the top ten daily maximum 8-h concentrations during the year at a specific monitoring site, the background $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ concentrations at elevations >1 km in the western U.S., while below 1-km baseline $O_3$ concentrations were higher in spring.

EPA in its ISA (EPA, 2020a) did not resolve the conflicting conclusions about when seasonal mean background $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$:

> Model-calculated USB $O_3$ 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 $O_3$ of 20–40 ppb during the $O_3$ 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 $O_3$ data (Fig. 2 of Jaffe et al., 2018), Jaffe et al. (2018) illustrated the vertical profiles of $O_3$ 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 $O_3$ 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 $O_3$ resulting from soil $NO_x$ and isoprene. The authors noted in their paper that substantial biases in the severity and timing of high-$O_3$ 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 $O_3$ tended to be higher in the summer than in the spring in most regions.

In the draft PA (EPA, 2022), the authors summarize their conclusions from the EPA's updated background $O_3$ 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 $O_3$ 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 previous $O_3$ PA (Section 2.4). (page 2-66)

- Consistent with previous assessments, USB is higher in the West than in the East. (page 2-65).

- Days for which MDA8 total $O_3$ 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 $O_3$ contributions peak during the traditional $O_3$ season (May through September), while long-range intercontinental transport of international $O_3$ (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 $O_3$. (page 2-64).

- Ozone formed from anthropogenic emissions in Canada and Mexico can peak in late spring or early summer when total $O_3$ 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 $O_3$ during the May through September $O_3$ season. (page 2-65).

- On high $O_3$ 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. (page 2-65).

- 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 $O_3$. 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 $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022), 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 page 2-48 of the PA the following:

The temporal pattern in the regional average clearly shows that the seasonality of MDA8 predictions for each total $O_3$ 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 $O_3$ 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 $O_3$ 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.

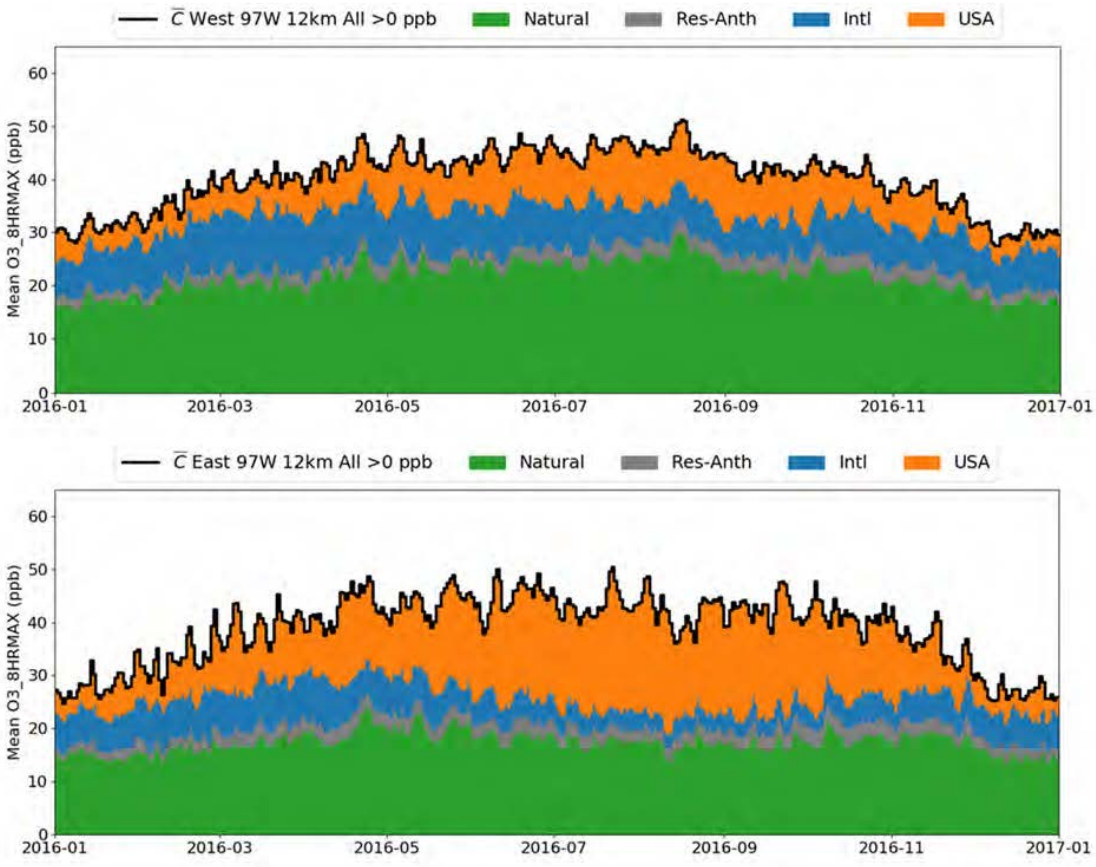


**Figure 3-36. Annual time series of regional average MDA8 $O_3$ concentration and 8-h contributions for the West (top) and the East (bottom). See legend above from the original figure. Source EPA (2022 Fig. 2-23, page 2-49).**

On page 2-50 of the draft PA, the authors note that Fig. 2-24 (page 2-51 in the 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 draft PA (EPA, 2022). 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 $O_3$ 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).

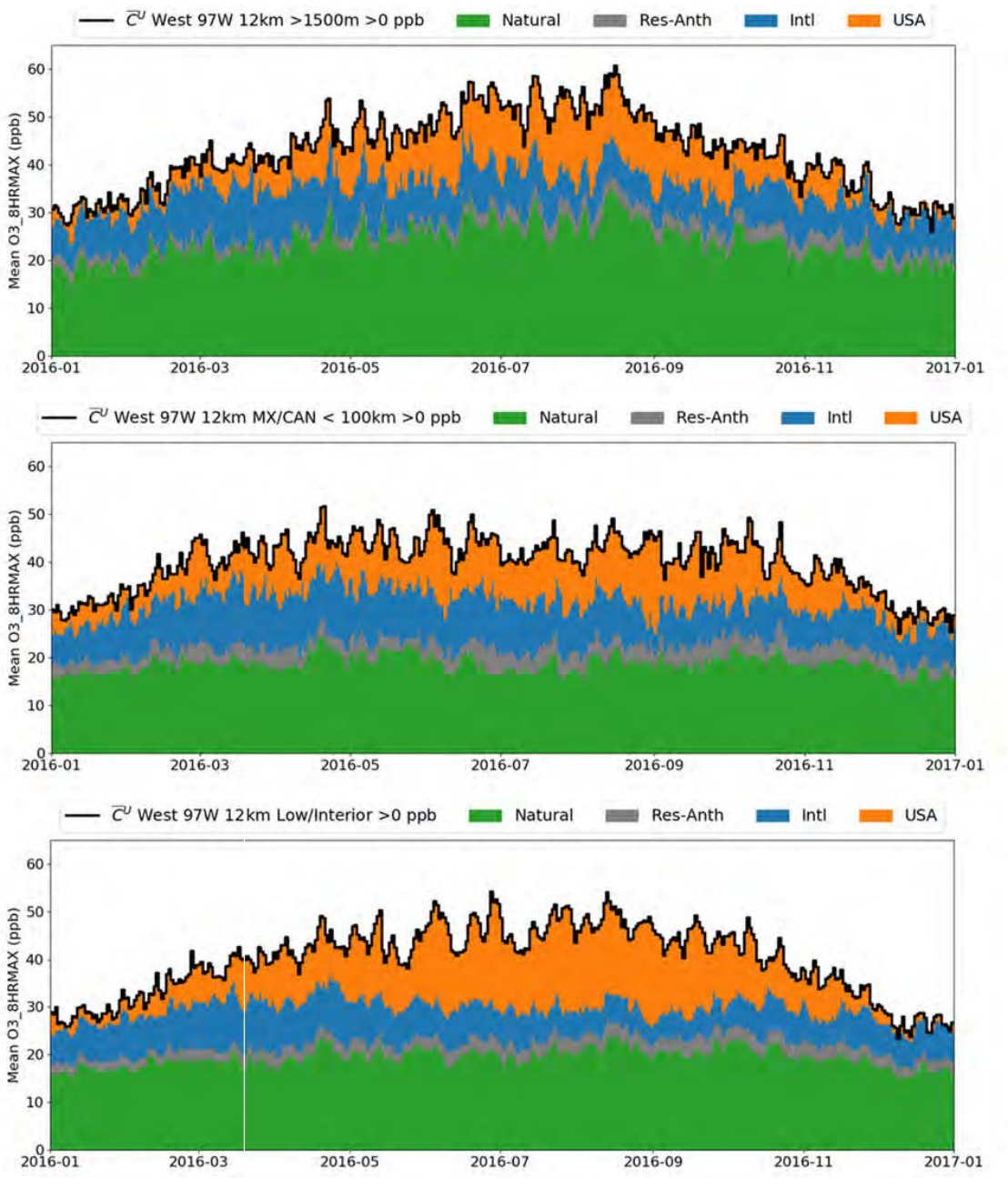



**Figure 3-37. Annual time series of regional urban area-weighted average MDA8 $O_3$ concentration and 8-h contributions for the High-elevation West (top), near-boarder West (middle), and Low/Interior West (bottom). See legend above from the original figure. Source EPA (2022 page 2-51).**

Earlier we discussed specific applications of background $O_3$ in the standard-setting process. While seasonal mean background $O_3$ estimates are of interest for identifying atmospheric processes, their use in assessing the role of background $O_3$ 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 $O_3$ is important for some human health risk analyses and hourly average background $O_3$ concentrations are important for vegetation risk analyses. It appears in the reading the ISA (EPA, 2020a) and draft PA (EPA, 2022) that the EPA is focused on understanding the percent contribution of background $O_3$ 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 $O_3$. In many cases, assessment of human health and ecological risks is based on understanding the role that background $O_3$ plays throughout the distribution of hourly average concentrations and not just the role that background $O_3$ plays during periods of highest $O_3$ 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 $O_3$ 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 $O_3$ concentrations, which contribute to the estimated human health risk. Background $O_3$ affects the overall human health risk assessments, and therefore, can influences the margin of safety determinations required for establishing the $O_3$ 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 $O_3$ exposure can be attributed to sources other than U.S. anthropogenic sources, both $USB_{AB}$ (EPA, 2014a; Dolwick et al., 2015) and Emissions Influenced Background (EIB) (Lefohn et al., 2014) daily estimates are used for illustrative purposes. Besides focusing on the highest hourly average $O_3$ concentrations (i.e., the upper end of the distribution of hourly values), it is important to understand the relative role that background $O_3$ plays over the entire distribution of total ambient $O_3$ 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 $O_3$ 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 $O_3$ 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 range that influenced EPA's human health risk estimates in 2014. As noted above, EIB $O_3$ 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.

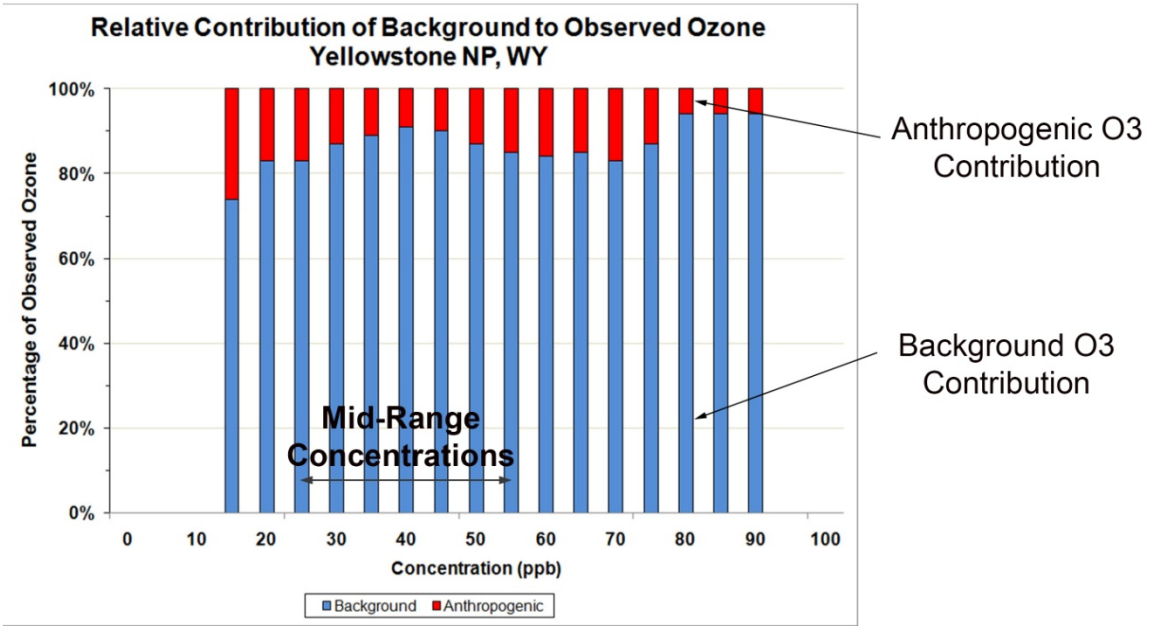



**Figure 3-38. Average relative contributions of current hourly background (blue) and anthropogenic $O_3$ (red) for Yellowstone NP (WY) (AQS ID 560391011) in 2006. (Source: Lefohn et al., 2014).**

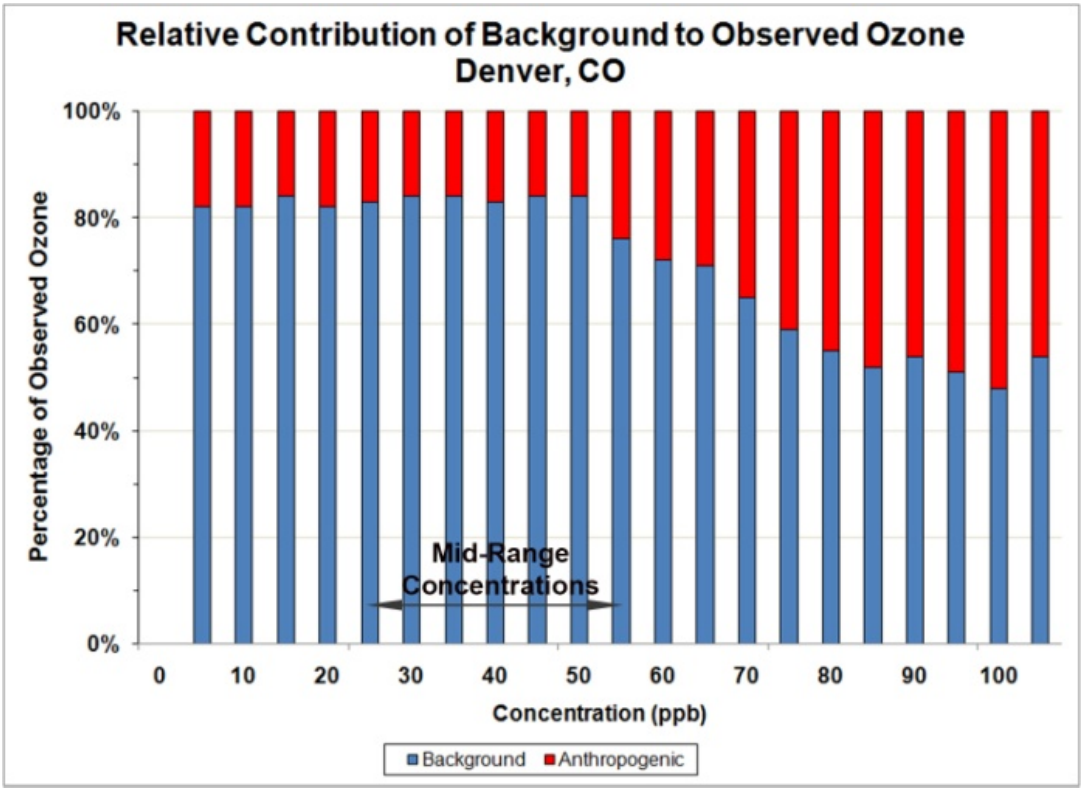



**Figure 3-39. Average relative contributions of current hourly background (blue) and anthropogenic $O_3$ (red) for Denver (CO) (AQS ID 080590006) in 2006. (Source: Lefohn et al., 2014).**

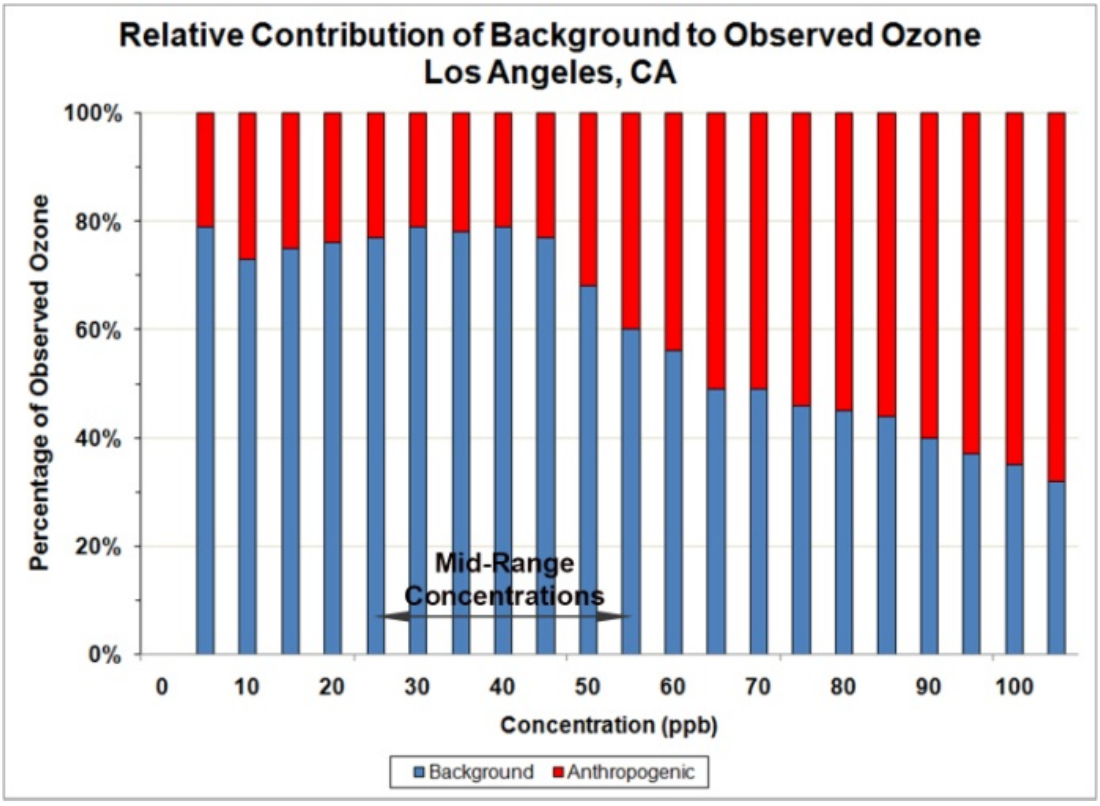



**Figure 3-40. Average relative contributions of current hourly background (blue) and anthropogenic $O_3$ (red) for Los Angeles (CA) area (AQS ID 060719004) in 2006. (Source: Lefohn et al., 2014).**

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 $O_3$ (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 $O_3$ concentrations (EPA, 2014a).

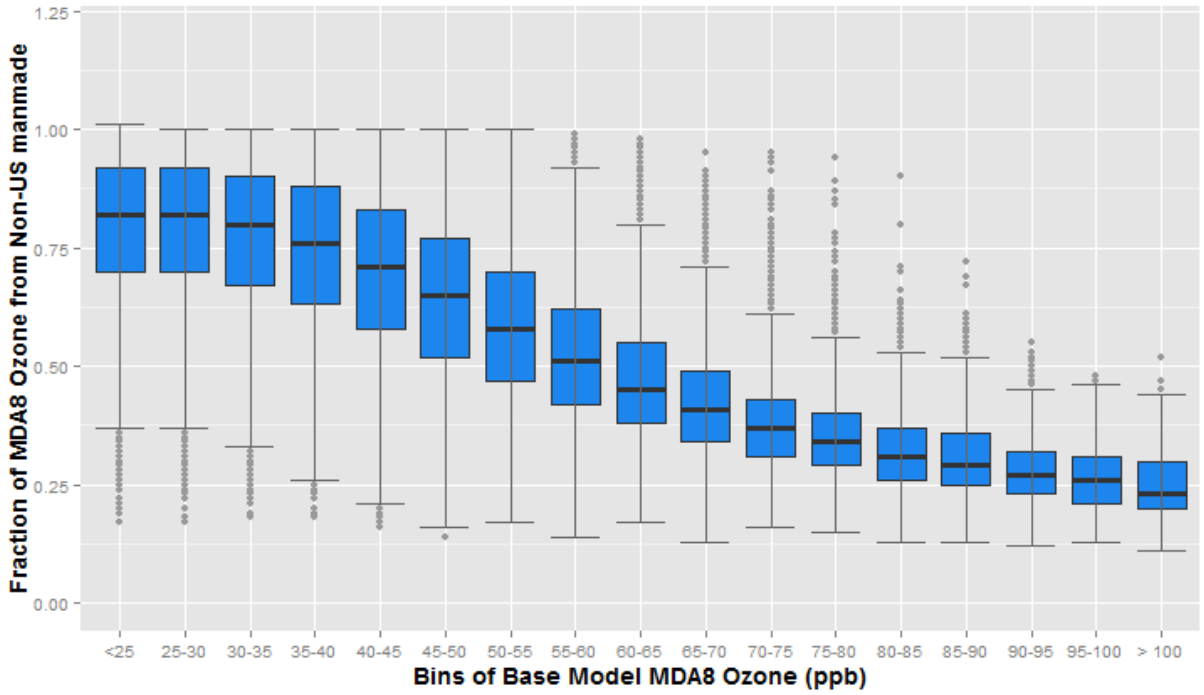
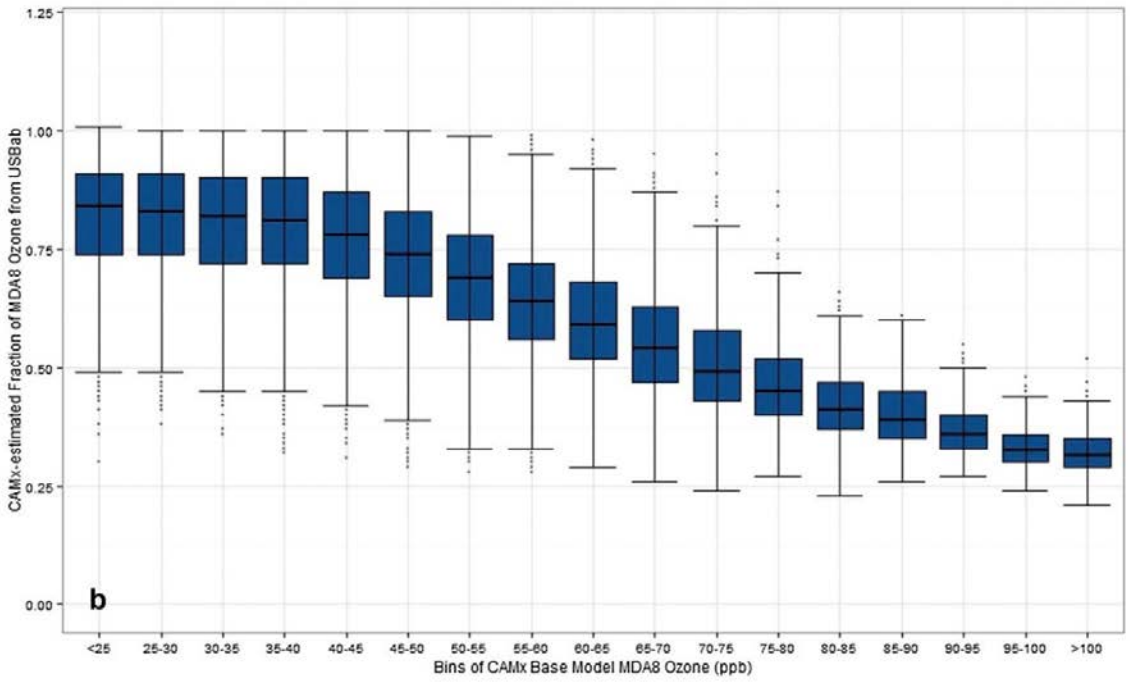



**Figure 3-41. Distributions of the relative proportion of apportionment-based U.S. Background ($USB_{AB}$) to total $O_3$ (all site-days), binned by modeled MDA8 from the 2007 source-apportionment simulation. Source: Fig. 2-15 in EPA (2014a) with slight modification.**

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 ($USB_{AB}$) $O_3$ fraction at monitoring locations across the western U.S. for the period April−October 2007, binned by base model MDA8 $O_3$ concentration ranges. 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 $O_3$ to total $O_3$ (identified by the authors as "Emissions Influenced Background") with increasing total $O_3$ concentration. At low-elevation and urban sites in the western U.S., $O_3$ concentrations estimated as USB, $USB_{AB}$, NAB, or EIB contributions were also reported to be independent of overall $O_3$ concentration, resulting in a decreasing relative background contribution with increasing total $O_3$ 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 $O_3$ to total $O_3$ with increasing $O_3$ 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 $O_3$ concentrations, U.S. background $O_3$ accounts for an important fraction of ambient $O_3$ concentrations because of stratospheric exchange, international transport, wildfires, lightning, global methane emissions, and natural biogenic and geogenic precursor emissions. At noted in the ISA (EPA, 2020a), as the literature on background $O_3$ has evolved, much of the discussion has focused on the relative

importance of stratospheric $O_3$ and intercontinental transport as the major sources of background $O_3$ (Page IS-14 of the ISA).

On Page IS-14 of the 2020 ISA (EPA, 2020a), the authors note that tropospheric $O_3$ derived from stratosphere-troposphere dynamics was described in detail in the 2013 Ozone ISA (EPA, 2013). Stratospheric air naturally rich in $O_3$ 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 $O_3$ 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 $O_3$ into the upper troposphere. Stratospheric intrusion events related to frontal passage and tropopause folding that reach the surface have less influence on surface $O_3$ during the summer months.

The relevance of stratospheric-to-tropospheric transport (STT) for influencing low-tropospheric $O_3$ 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). Lefohn et al. (2001) (authors A.S. Lefohn, S.J. Oltmans, T. Dann, and H.B. Singh) described the contribution of stratospheric $O_3$ to observed $O_3$ levels across the U.S. The authors attributed STT processes to the observation that hourly average $O_3$ concentrations $\geq$ 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 $O_3$ 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 $O_3$ 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-$O_3$ 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 $O_3$ 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 $O_3$ using lidar and surface measurements from the Front Range of the Colorado Rocky Mountains during the 1999 $O_3$ 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 $O_3$ to within 1

km of the highest peaks in the Rocky Mountains on 6 May 1999. One-minute average $O_3$ concentrations exceeding 100 ppb were subsequently measured at a surface site in Boulder, and daily maximum 8-h $O_3$ concentrations greater or equal to the 2008 NAAQS $O_3$ standard of 0.075 ppm were recorded at 3 of 9 Front Range monitoring stations. Other springtime peaks in surface $O_3$ 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 $O_3$ was significant and could lead to exceedance of the 2008 NAAQS $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ concentrations at both high- and low-elevation monitoring. Lefohn et al. (2012), quantified the frequency of STT events that result in $O_3$ 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 $O_3$ concentrations. Their results indicated that STT down to the surface (STT-S) frequently contributed to enhanced surface $O_3$ hourly averaged concentrations at sites across the U.S., with substantial year-to-year variability. The $O_3$ concentrations associated with the STT-S events appeared to be large enough to enhance the measured $O_3$ concentrations during specific months of the year. Months with a statistically significant coincidence between enhanced $O_3$ 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 $O_3$ concentrations coincident with STT-S events. Škerlak et al. (2014) noted that STT processes, which contribute to background $O_3$, 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 $O_3$ data. In the EPA AQS database, hourly average $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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).

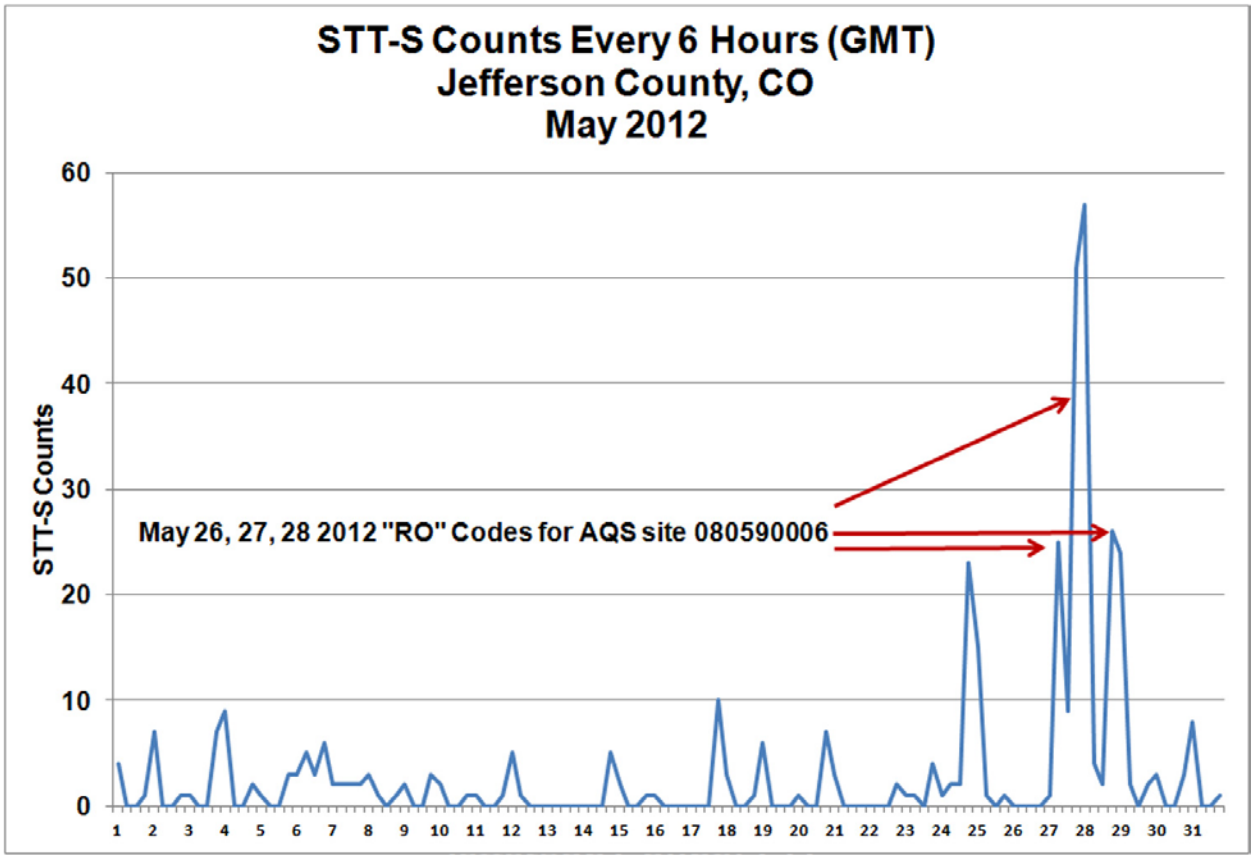



**Figure 3-43. Relating STT-S counts for every six hours (GMT) with "RO" codes for a site in Jefferson County, Colorado (AQS ID 080590006) for May 2012. See Lefohn et al. (2011, 2012, 2014) for details how the STT-S values are estimated.**

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 $O_3$ 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 $O_3$ 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 $O_3$ precursor source in Asia, have declined as confirmed by rapidly decreasing satellite derived tropospheric $NO_2$ 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 $O_3$ 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, $O_3$ concentrations continue to rise in some locations (Xu et al., 2020). Lefohn et al. (2017) analyzed trends using hourly average $O_3$ concentrations from monitoring sites in Hong Kong and mainland China and noted the following:

> The changes in $O_3$ 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, $O_3$ 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 $O_3$ concentrations have not decreased due to the contribution of long-range transport from increasing $O_3$ 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 $O_3$ 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 $O_3$ metrics were characterized through the year 2016. The authors discussed the long-term trends in the various $O_3$ metrics. Large and significant increases of $O_3$ 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 $O_3$ metrics at the Shangdianzi background site in the NCP indicate increases at rates of more than 2%/yr during 2004–2016. In contrast, $O_3$ 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 $NO_2$ has been declining over the major air pollution regions in China, strong increases of $O_3$ 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 $O_3$ levels at urban sites, though it has stopped the increases of $O_3$ 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 $O_3$ 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 $O_3$ 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 4$^{th}$ 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 $O_3$ 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 $O_3$ exposures. Oltmans et al. (2010) observed an $O_3$ episodic enhancement during April 2008 from biomass burning effluent from Eurasia that resulted in unusually high $O_3$ 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 $O_3$ 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 $O_3$ 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 4$^{th}$ highest daily maximum 8-h concentration exposure for 13 $O_3$ 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 $O_3$ 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, $O_3$ can have an atmospheric lifetime on the order of weeks. As a result, background $O_3$, and to a lesser extent background $O_3$ 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 $O_3$ originates from production of $O_3$ in the stratosphere through interactions between ultraviolet light and molecular oxygen. $O_3$ 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 $O_3$ concentrations. These events are called stratospheric intrusions and can result in relatively high USB levels of $O_3$ at the surface, especially at higher-elevation sites. Other natural sources of $O_3$ precursor emissions include wildfires, lightning, and vegetation. Biogenic emissions of methane, which can be chemically converted to $O_3$ over relatively long time scales, can also contribute to USB $O_3$ levels. Finally, manmade precursor emissions from other countries can contribute to the global burden of $O_3$ in the troposphere and to increased USB $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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).

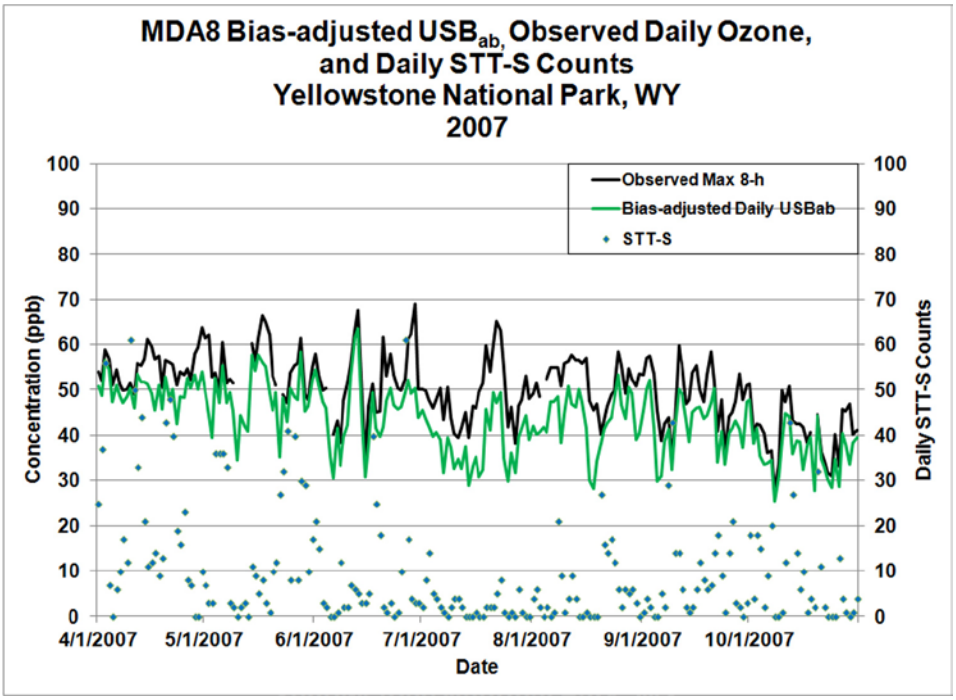



**Figure 3-44. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for**

**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 $O_3$ 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.**

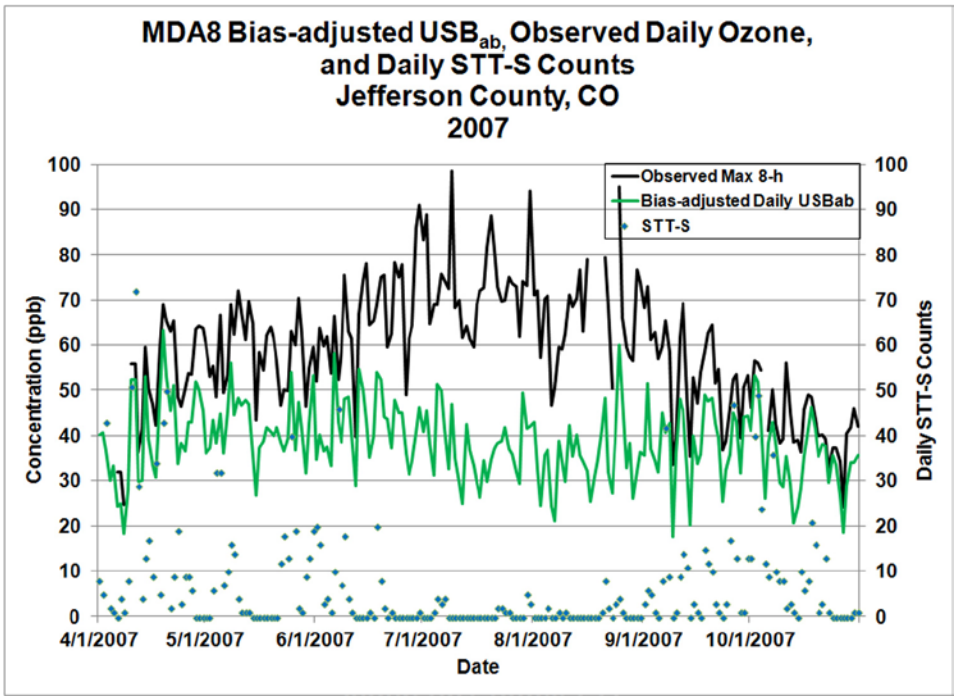



**Figure 3-45. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for Jefferson County (CO) (AQS ID 080590006) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

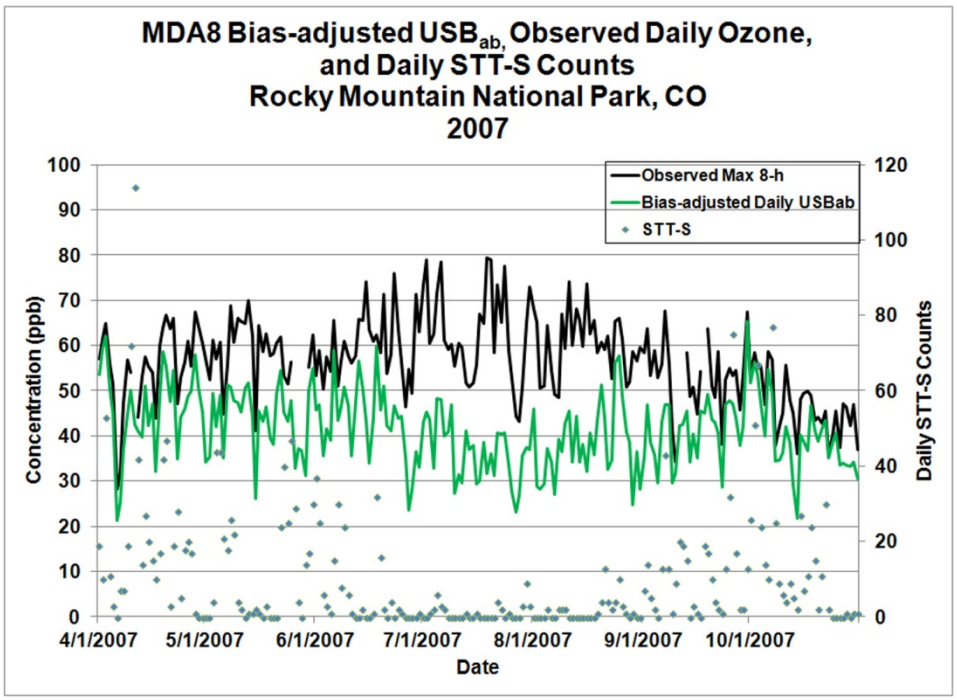



**Figure 3-46. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for Rocky Mountain National Park (CO) (AQS ID 080690007) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

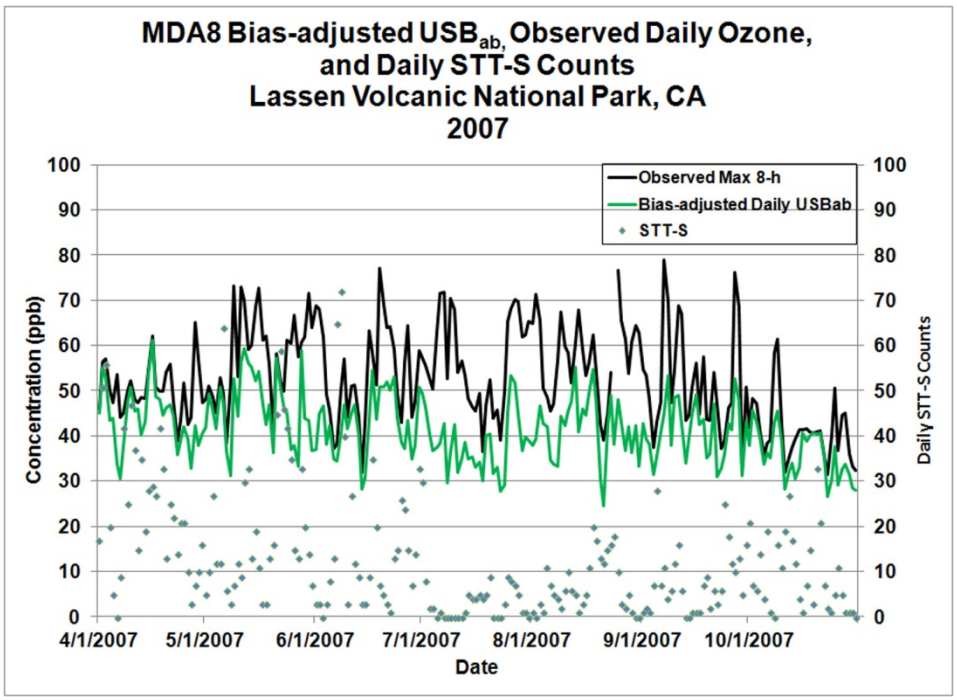



**Figure 3-47. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for Lassen Volcanic National Park (CA) (AQS ID 060893003) for April-October 2007. The daily stratospheric-tropospheric transport to 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 how the STT-S values are estimated.**

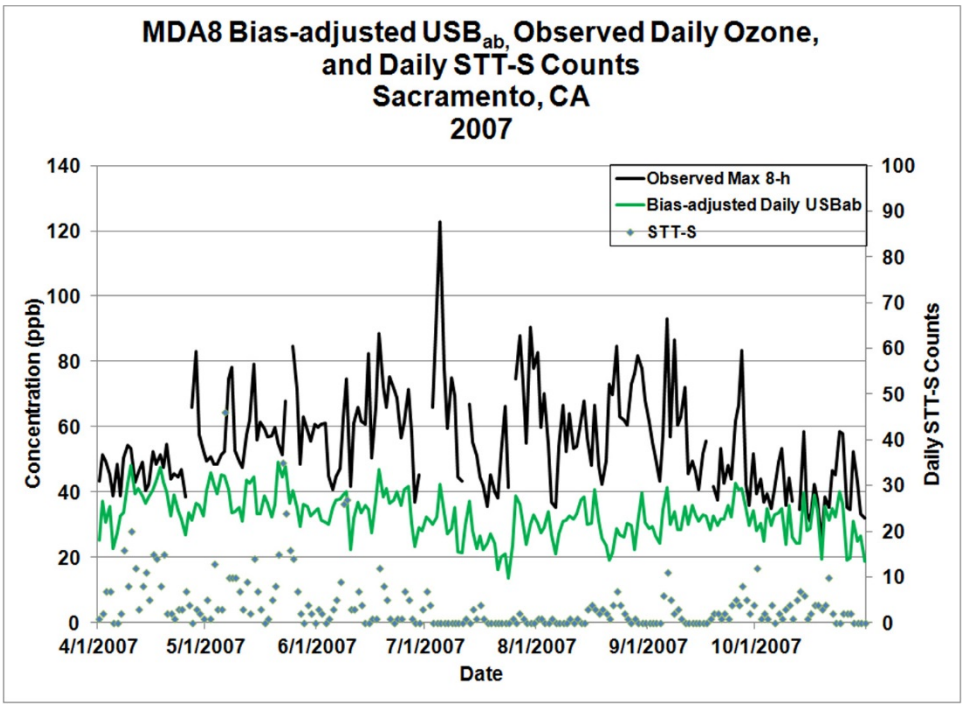



**Figure 3-48. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for Sacramento (CA) (AQS ID 060670012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

### 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 $O_3$ modeling results performed by the EPA and presented in the draft PA (EPA, 2022) 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 $O_3$. 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 $O_3$ levels. Further, background $O_3$ 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 $O_3$ 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 $O_3$ on the NAAQS review is appropriate, as based on previous modeling exercises and the more recent EPA analyses summarized herein. First, background $O_3$ exists and can comprise a considerable fraction of total seasonal mean MDA8 $O_3$ and W126 across the U.S. Air quality models can estimate the fractional contribution of background sources to total $O_3$ 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 $O_3$ exceedances of the NAAQS. Higher $O_3$ days generally have smaller fractional contributions from background. This finding indicates that the relative importance of background $O_3$ would increase were $O_3$ concentrations to decrease with a lower level of the $O_3$ NAAQS. Third and finally, while the majority of modeled $O_3$ exceedances have local and domestic regional emissions as their primary cause, there can be events where $O_3$ 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 $O_3$ exists and can comprise a considerable fraction of total seasonal mean MDA8 $O_3$ and W126 across the U.S. Air quality models can estimate the fractional contribution of background sources to total $O_3$ 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 $O_3$ exceedances of the NAAQS. Higher $O_3$ days generally have smaller fractional contributions from background. This finding indicates that the relative importance of background $O_3$ would increase were $O_3$ concentrations to decrease with a lower level of the $O_3$ NAAQS.

- Third and finally, while the majority of modeled $O_3$ exceedances have local and domestic regional emissions as their primary cause, there can be events where $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022) indicate that

- The current analysis indicates that natural and USA $O_3$ contributions peak during the traditional $O_3$ season (May through September), while long-range intercontinental transport of international $O_3$ (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 draft PA (EPA, 2022) appear to show a different seasonal pattern for when background $O_3$ is highest when these results are compared with the previous conclusions in the 2014 PA (EPA, 2014a). As noted in the draft PA (EPA, 2022), 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 draft PA (EPA, 2022) also peaks during the traditional $O_3$ 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 $O_3$ modeling, when seasonal mean background $O_3$ is greatest. Unfortunately, the draft PA (EPA, 2022) 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 $O_3$ concentrations are the greatest across the U.S. In a review of the literature, Jaffe et al. (2018) concluded that model-calculated USB $O_3$ 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 $O_3$ concentrations, as well as the seasons that exhibit the highest background $O_3$ concentrations. In the database, information from some of the $O_3$ monitoring sites illustrate the compression described in the literature about the distribution of hourly average $O_3$ 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 $O_3$ 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 $O_3$, wind speed and direction, temperature, relative humidity, and solar radiation. The network was established to provide a reasonable long-term and continuous record of $O_3$ 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 $O_3$ concentrations which were more than 0.08 ppm. Most days with elevated $O_3$ 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. 3-41 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ because of the occurrence of more frequent low hourly average concentrations.

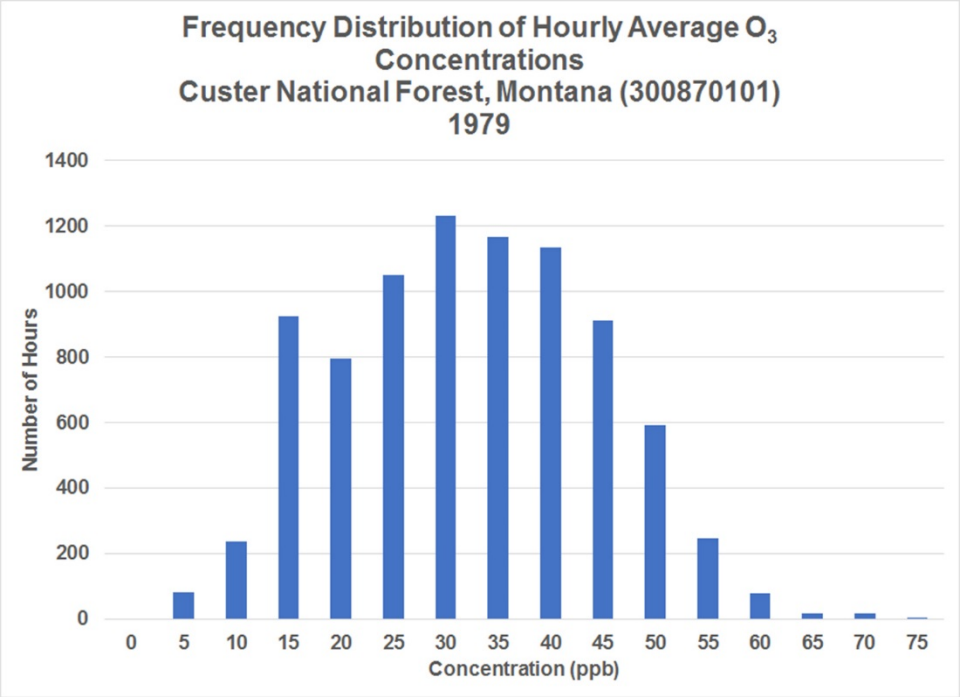



**Figure 3-49. Frequency distribution of the hourly average $O_3$ concentrations in 1979 for Custer National Forest (MT) (300870101) monitoring station. Source of data is from the EPA's AQS database.**

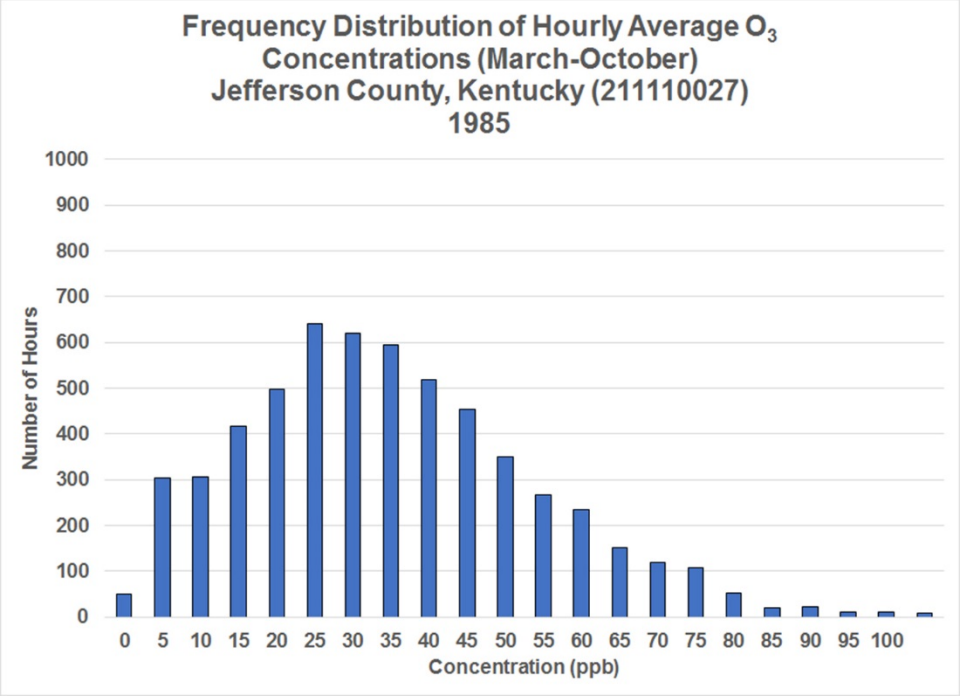



**Figure 3-50. Frequency distribution of the hourly average $O_3$ concentrations in 1985 for an urban-influenced site in Jefferson County (KY) (211110027) monitoring station. Source of data is from the EPA's AQS database.**

Based on the comparison of the two sites, Lefohn et al. (1998) hypothesized that as adequate control strategies were implemented to meet the $O_3$ 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 $O_3$ 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.

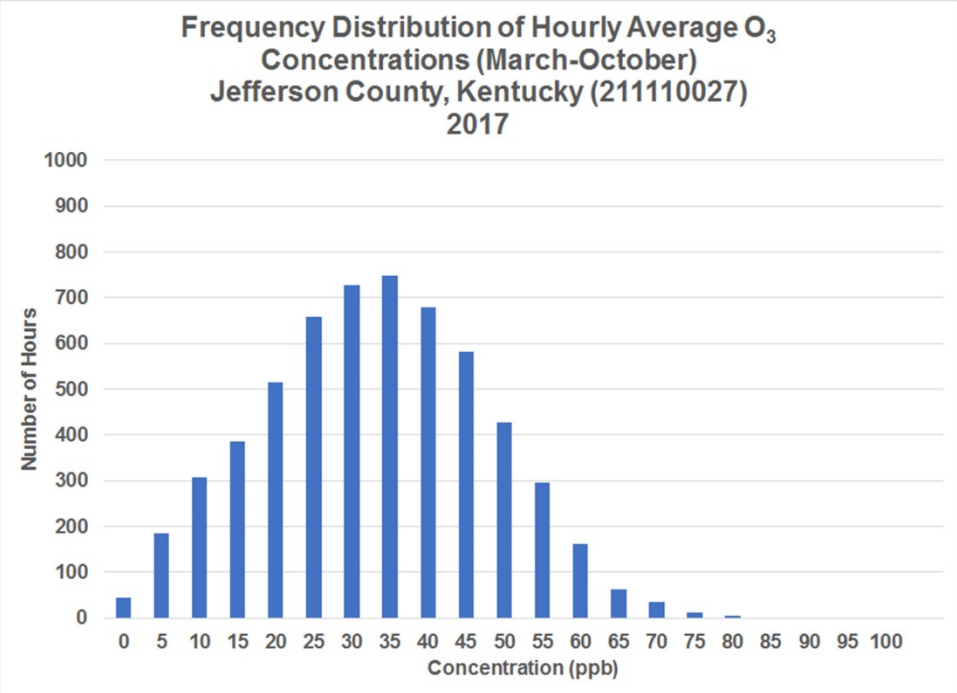



**Figure 3-51. Frequency distribution of the hourly average $O_3$ concentrations in 2017 for an urban-influenced site in Jefferson County (KY) (211110027) monitoring station. Source of data is from the EPA's AQS database.**

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 $O_3$ exposure metrics to changes in hourly $O_3$ 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 $O_3$ concentrations. The patterns of changes in hourly average $O_3$ 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 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 $O_3$ 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 $O_3$ 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 $O_3$ 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), 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 $O_3$ 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 $O_3$ increases its role in the percent contribution to the total $O_3$ 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 $O_3$ concentrations that represent background $O_3$ 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 $O_3$ 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 $O_3$ contributed a major portion of the total observed hourly average $O_3$ concentrations for all concentrations. Fig. 3-53 illustrates the distribution of hourly average $O_3$ concentrations for a site at Mesa Verde (CO) National Park. This site also exhibits the bell-shaped-like distribution pattern. The change in distribution patterns for the hourly average $O_3$ concentrations are not just occurring in the western U.S. 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 $O_3$ 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 $O_3$ concentration of 91 ppb. In 2018, the site exhibited a maximum hourly average $O_3$ concentration of 76 ppb.

The bell-shaped-like distribution curve provides us with the ability to gain insights about the distribution of background $O_3$ concentrations site by site. *While we cannot quantify the exact distribution of the hourly average background $O_3$ concentrations, we know that as the compression occurs as emission are reduced that background $O_3$ is encapsulated in the distribution.* For example, for the Yellowstone NP (WY) site in 2018 (Fig. 3-52), the highest frequency of hourly average $O_3$ concentrations is in the range of 40-45 ppb. The highest hourly average $O_3$ 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 $O_3$ 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 $O_3$ concentrations is 35 ppb. For the Monroe County (MO) site in 2018 (Fig. 3-56), the maximum frequency of $O_3$ 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 $O_3$ becomes even more dominant.

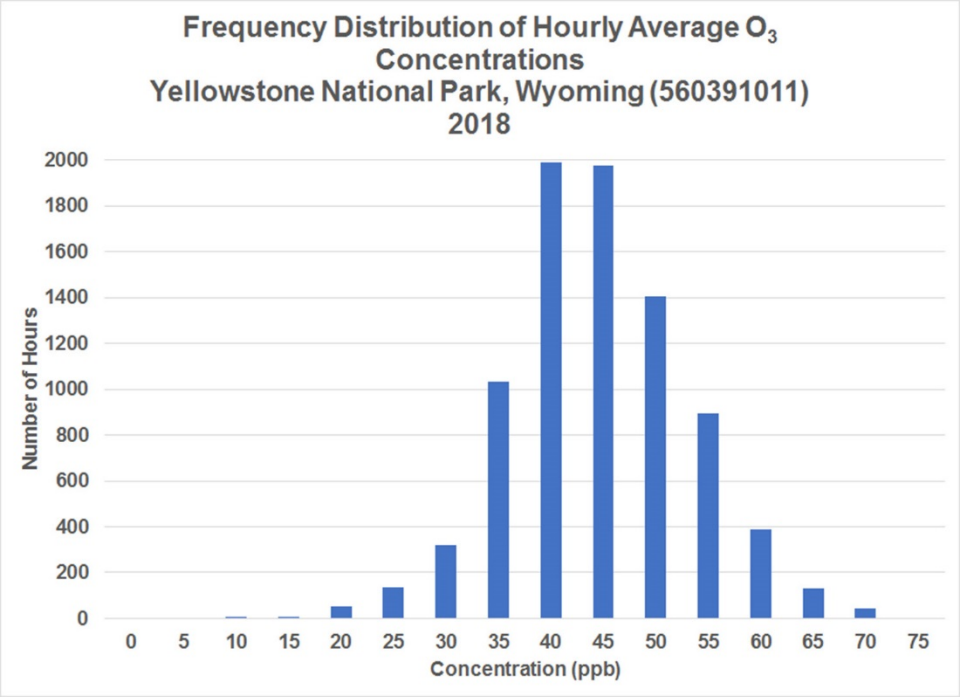



**Figure 3-52. Frequency distribution of the hourly average $O_3$ concentrations for January-December 2018 for Yellowstone NP (WY) (560391011). Source of data is from the EPA's AQS database.**

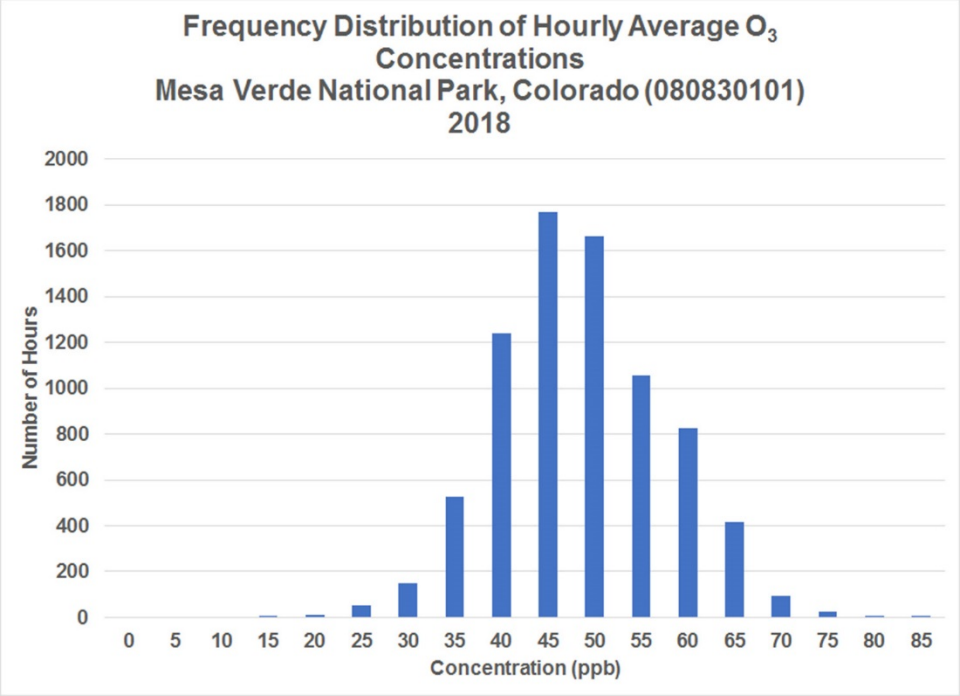



**Figure 3-53. Frequency distribution of the hourly average $O_3$ concentrations for January-December 2018 for Mesa Verde National Park (CO) (080830101). Source of data is from the EPA's AQS database.**

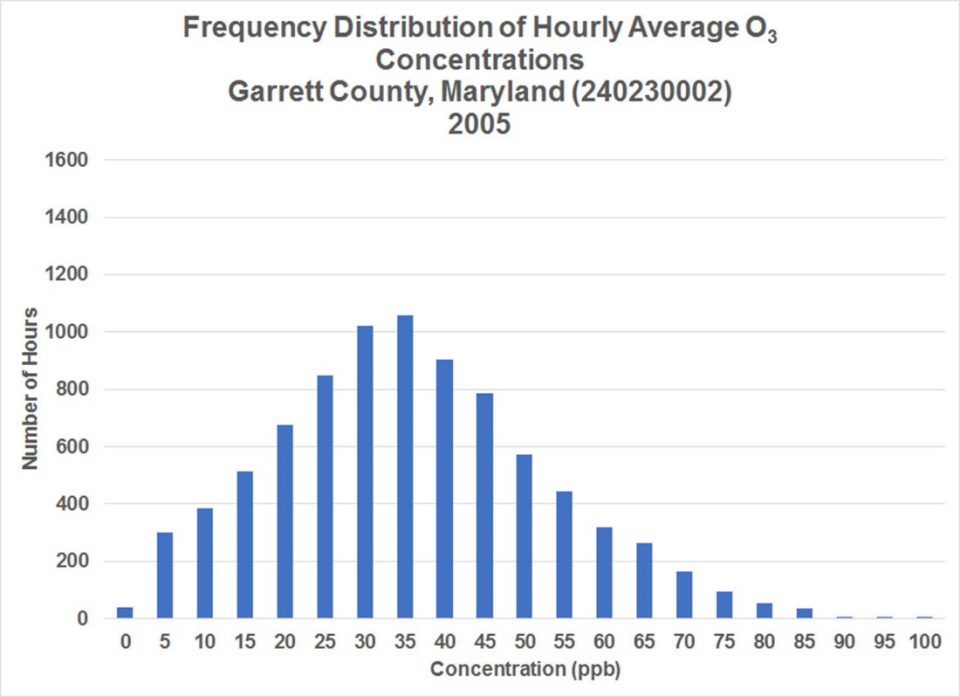



**Figure 3-54. Frequency distribution of the hourly average $O_3$ concentrations for January-December 2005 for Garrett County (MD) (240230002). Source of data is from the EPA's AQS database.**

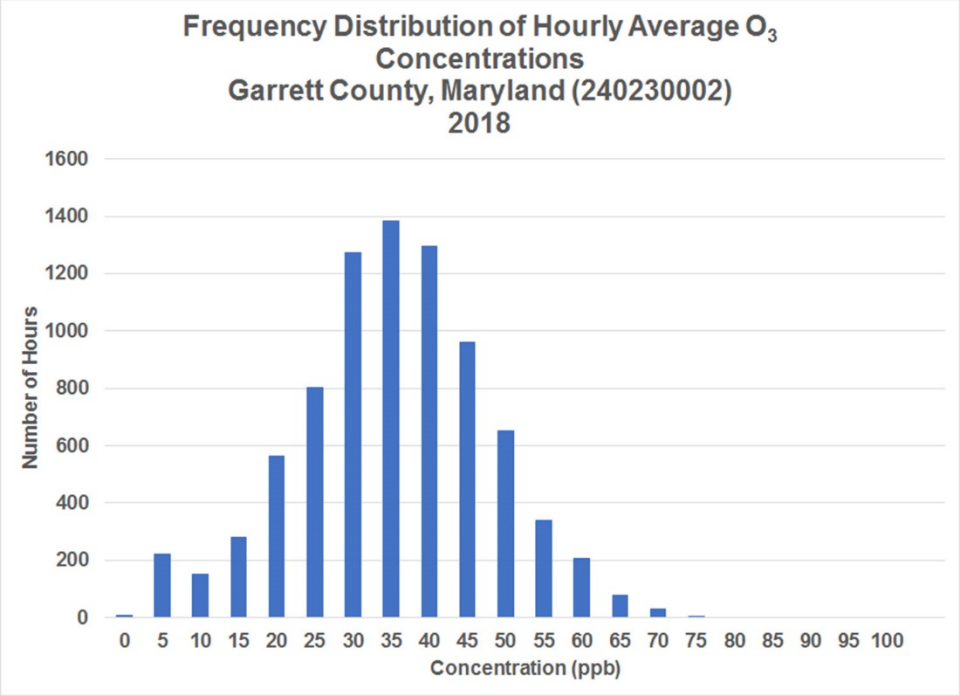



**Figure 3-55. Frequency distribution of the hourly average $O_3$ concentrations for January-December 2018 for Garrett County (MD) (240230002). Source of data is from the EPA's AQS database.**

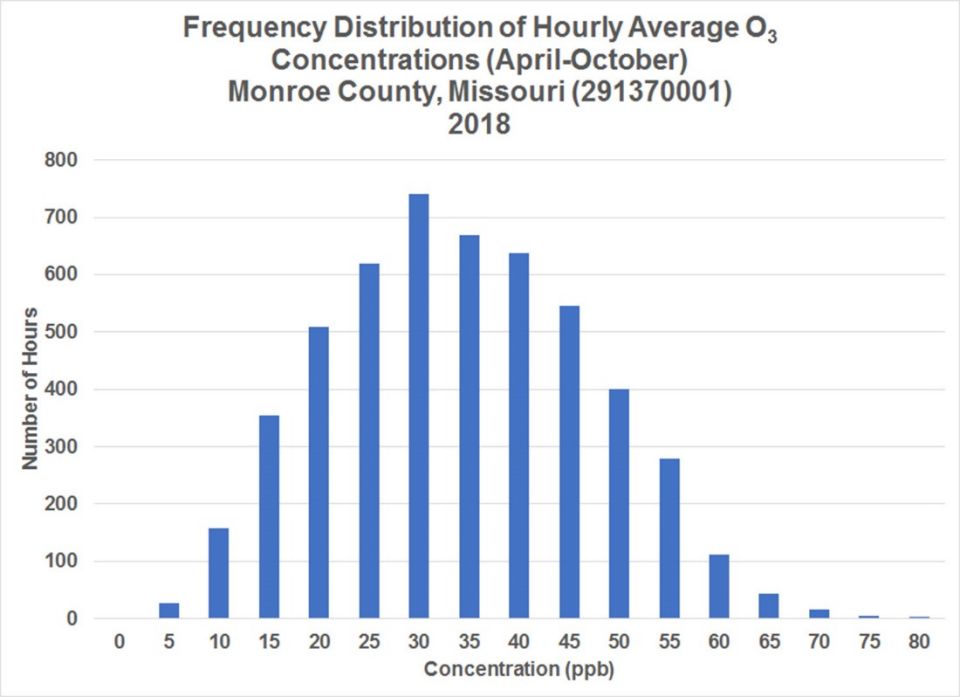



**Figure 3-56. Frequency distribution of the hourly average $O_3$ concentrations for January-December 2018 for Monroe County (MO) (291370001). Source of data is from the EPA's AQS database.**

In addition to the observation of a compression of the distribution of hourly average $O_3$ 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 $O_3$ 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 $O_3$ monitoring season defined for each state and the District of Columbia was not adequately capturing the occurrence of daily maximum 8-h $O_3$ average concentrations equal to or above 0.060 ppm. In response to this concern, in the 2015 $O_3$ NAAQS rulemaking (Federal Register, 2015 – page 65416), the EPA determined that the lengthening of the $O_3$ monitoring seasons in 32 states and the District of Columbia was required. The Agency indicated that ambient $O_3$ 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 $O_3$ seasons prior to 2015. The EPA concluded that it was important to monitor for $O_3$ during the periods when ambient concentrations could approach the level of the NAAQS to ensure that the public was informed when exposure to $O_3$ 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 $O_3$ data year-round during 2010-2013. More than half of the $O_3$ 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 $O_3$ average equal to or above 0.060 ppm in the months outside each state's current $O_3$ 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 $O_3$ 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 $O_3$ patterns varied year-to-year due primarily to highly variable meteorological conditions conducive to the formation of elevated $O_3$ concentrations early or late in the season in some years and not others. The EPA believed it was important that $O_3$ 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 $O_3$ monitoring season involved adding earlier, as well as later months to the monitoring seasons that were used prior to 2015. Changes to the required $O_3$ 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 $O_3$ 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 $O_3$ monitoring data show that the highest $O_3$ 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 $O_3$ monitors for the period 2006-2010. Using hourly average $O_3$ data from 57 National Parks, Table 7A-2 shows that several of the $O_3$ 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 $O_3$ 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 $O_3$ levels (Neufeld et al., 2019). Neufeld et al. (2019) analyzed $O_3$ 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 $O_3$ 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.

In addition to the Tennessee example, Blanchard et al. (2019) noted that the highest peak 8-h $O_3$ 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 $O_3$ 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.

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 above) 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 draft Ozone National Ambient Air Quality Standards document 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 $O_3$ 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-Kendell nonparametric test, that the most frequent hourly average $O_3$ concentrations were in the 40-45 ppb range. Fig. 3-57 illustrates the average relative contributions of current hourly background (blue) and anthropogenic $O_3$ (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 $O_3$ concentrations in the 40-45 ppb compared to total observed $O_3$ values consist mostly of estimated background $O_3$ 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 $O_3$ 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 $O_3$ increases its role as indicated by the percent contribution to the total $O_3$ measured. As U.S. anthropogenic contributions approach zero (which is not possible socially), what will remain is a distribution of hourly average $O_3$ concentrations that represent background $O_3$ 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) $O_3$ 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 $O_3$ 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) $O_3$ monitoring site. All available data over the entire period of record were included in this analysis independent of the EPA-defined $O_3$ 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.

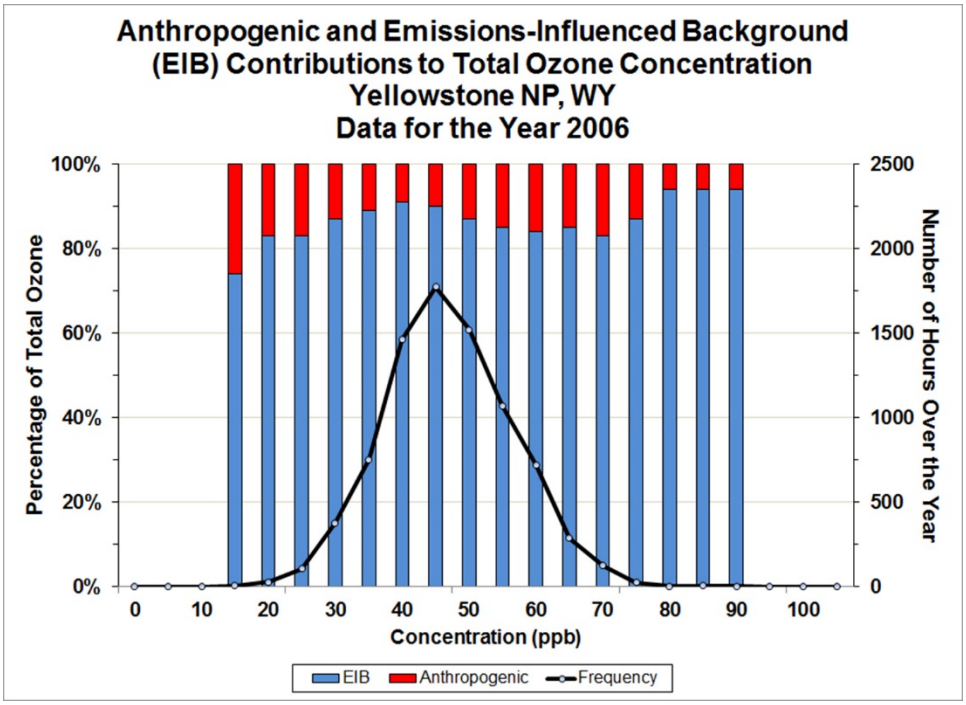



**Figure 3-57. Average relative contributions of hourly background (blue) and anthropogenic $O_3$ (red) for Yellowstone NP (WY) (AQS ID 560391011) in 2006. The black line indicates the distribution of the hourly average $O_3$ concentrations. (Source: Lefohn et al., 2014).**

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 $O_3$.

Besides discussing the compression of the hourly average $O_3$ 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 $O_3$ modeling.

### 3.2.9 Observed Ozone Exposure Patterns and Model Performance

The USB modeling results described in the draft PA (EPA, 2022) indicate the following seasonal patterns:

- The current analysis indicates that natural and USA $O_3$ contributions peak during the traditional $O_3$ season (May through September), while long-range intercontinental transport of international $O_3$ (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 draft PA (EPA, 2022) show a different seasonal pattern for when background $O_3$ 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 $O_3$) 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 $O_3$ was greatest over the U.S. during the spring and early summer (i.e., June).

Bias adjustment to estimated background $O_3$ 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 $O_3$ ($GBO_3$) or anthropogenic $O_3$ modeling components. Global background $O_3$ ($GBO_3$) 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 $GBO_3$ 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 $GBO_3$ 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 $GBO_3$ 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 $GBO_3$. For some urban sites (i.e., Boston, Dallas, Detroit, New York, and Sacramento), the model also under predicted during the spring when higher $GBO_3$ 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 $GBO_3$ coincided with STT-S events. For urban sites with higher spring anthropogenic influences, it was more likely that over predictions were associated with anthropogenic $O_3$.

The important role that the stratosphere played in episodic (i.e., short-term, high concentration events), as well as enhancements (subtle increases in $O_3$ concentration), to surface $O_3$ values was noted in Lefohn et al. (2012). The authors quantified the frequency of stratosphere-troposphere exchange (STE) events that result in $O_3$ 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 $O_3$ concentrations. The authors results indicated that STT down to the surface (STT-S) frequently contributed to enhanced surface $O_3$ hourly averaged concentrations at sites across the U.S., with substantial year-to-year variability. The $O_3$ concentrations associated with the STT-S events appeared to be large enough to enhance the measured $O_3$ concentrations during specific months of the year. Months with a statistically significant coincidence between enhanced $O_3$ 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 $O_3$ 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 $O_3$ 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 $GBO_3$ 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 draft PA (EPA, 2022), 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 draft PA (EPA, 2022), 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 draft PA (EPA, 2022), 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 observed biases in the model were observed may explain why the seasonal patterns exhibited in the model described in the draft PA (EPA, 2022) did not match the patterns observed when one characterizes ambient data (see previous section). As noted above, the EPA model described in the draft PA (EPA, 2022) showed 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 $O_3$ surface sites across the U.S. Enhanced, as well as episodic, contributions to ambient $O_3$ 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 draft PA (EPA, 2022).

**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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ sources. As pointed out by the authors, the simulations started in a USB $O_3$ 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 $O_3$ 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 $O_3$ or local). The authors noted that simulations performed even better when USB $O_3$ 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 $O_3$ contributions was weak.

Fig. 3-58 illustrates for the Chatfield site in 2011 the comparison of observed daily $O_3$ MDA8 concentrations and the STT-S counts (estimated for a site in the Denver area for 2011). The USB $O_3$ 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 draft PA (EPA, 2022, 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 draft PA (EPA, 2022). The draft PA (EPA, 2022) 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, 2022, page 2-48). It is interesting to note that the pattern described in the draft PA (EPA, 2022) 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 $GBO_3$, 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) $O_3$ were followed by lower values during the summer, due to heightened chemical interaction with anthropogenic sources, which were then followed by rising EIB $O_3$ 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 $O_3$ estimated throughout the year. EIB $O_3$ at all high-elevation sites contributed a significant proportion to total $O_3$ throughout the year and throughout the observed total $O_3$ frequency distribution, while EIB $O_3$ at most urban sites contributed a major portion to total $O_3$ 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 draft PA (EPA, 2022) and Lefohn et al. (2012) may be attributable to the lack of bias adjustment in the EPA model described in the draft PA (EPA, 2022). One might hypothesize that if a bias adjustment were performed on the daily MDA8 predictions in the EPA model described in the draft PA (EPA, 2022) 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.

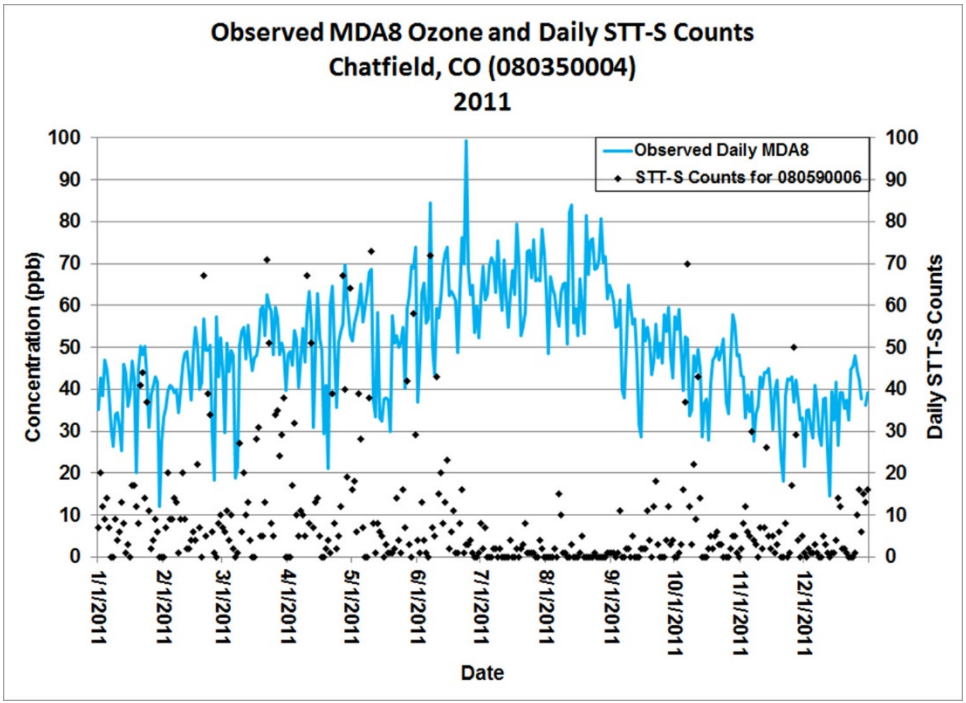



**Figure 3-58. Observed MDA8 $O_3$ and daily STT-S counts for 2011 for the Chatfield (CO) (080350004) monitoring site. The STT-S counts, which were quantified for a site in Jefferson County (CO) (080590006), were superimposed over the observed data at the Chatfield site.**

For the eight sites used in the risk assessment (Atlanta, Boston, Dallas, Detroit, Philadelphia, Phoenix, Sacramento, and St. Louis), the draft PA (EPA, 2022) does not provide examples for specific sites that describe the observed and USB concentrations. In the draft PA (EPA, 2022), 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 $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022).

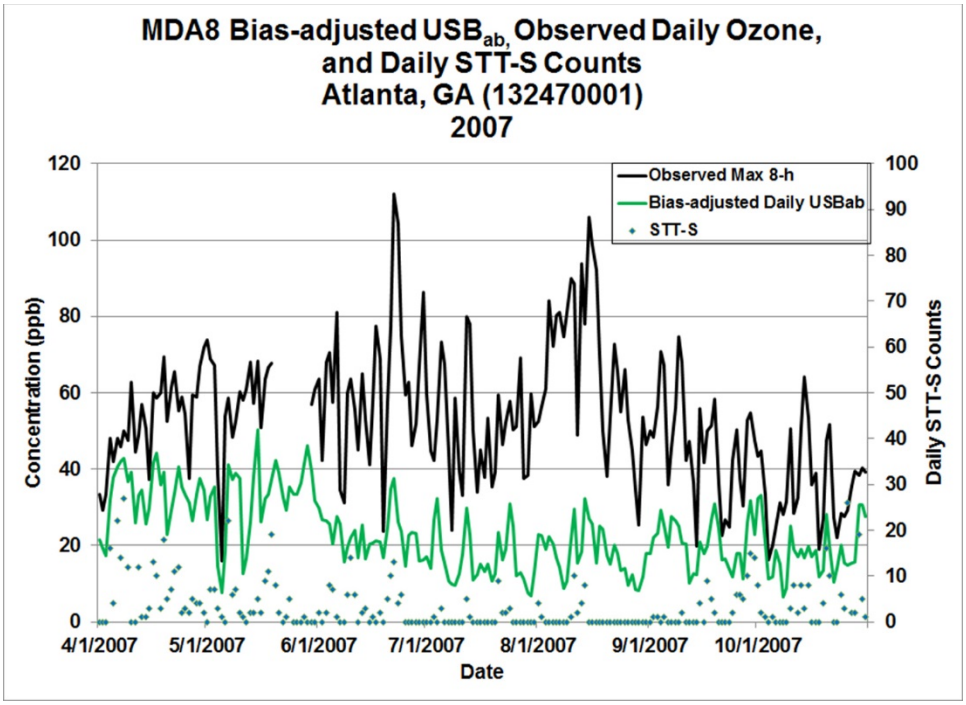



**Figure 3-59. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Atlanta, Georgia (AQS ID 132470001) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

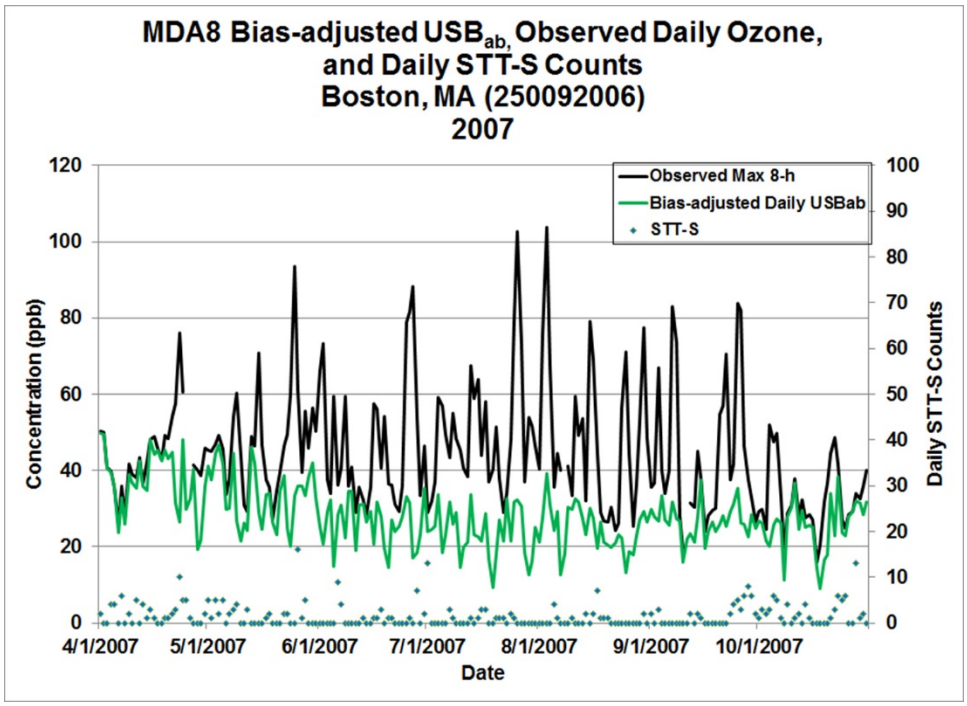



**Figure 3-60. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Boston, Massachusetts (AQS ID 250092006) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

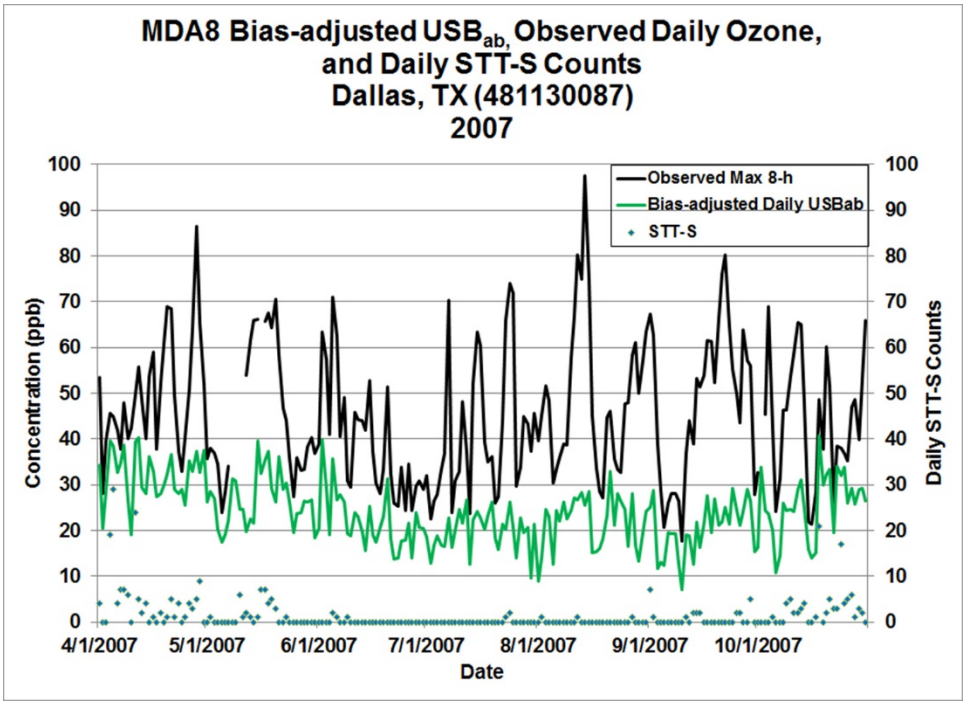



**Figure 3-61. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Dallas, Texas (AQS ID 481130087) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

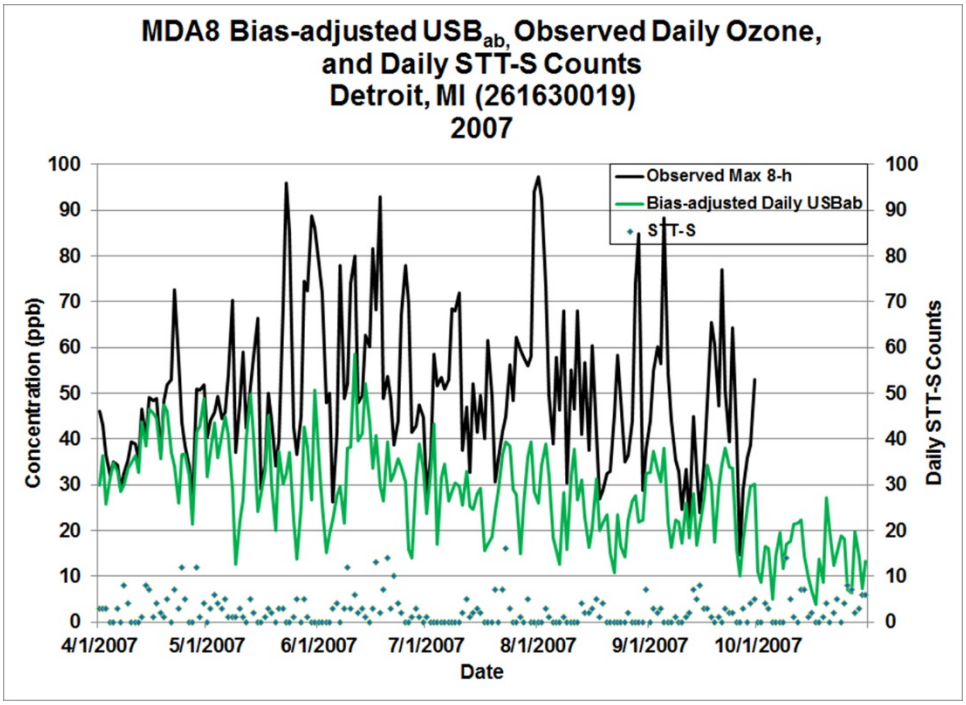


**Figure 3-62. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Detroit, Michigan (AQS ID 261630019) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

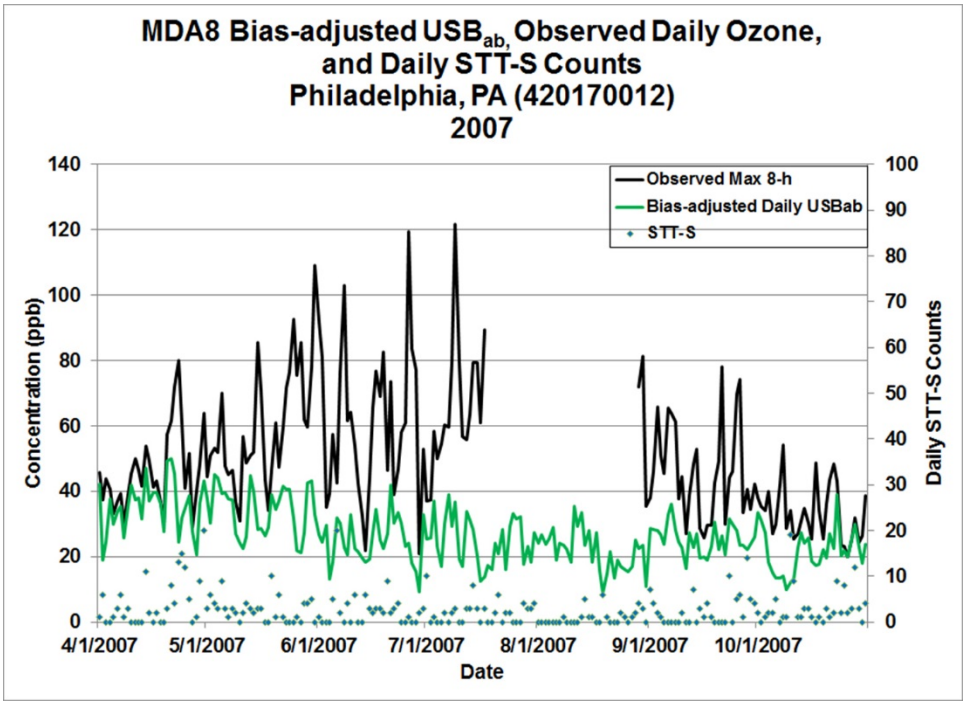



**Figure 3-63. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Philadelphia, Pennsylvania (AQS ID 420170012) for April-October 2007. The daily stratospheric-tropospheric transport 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 $O_3$ 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.**

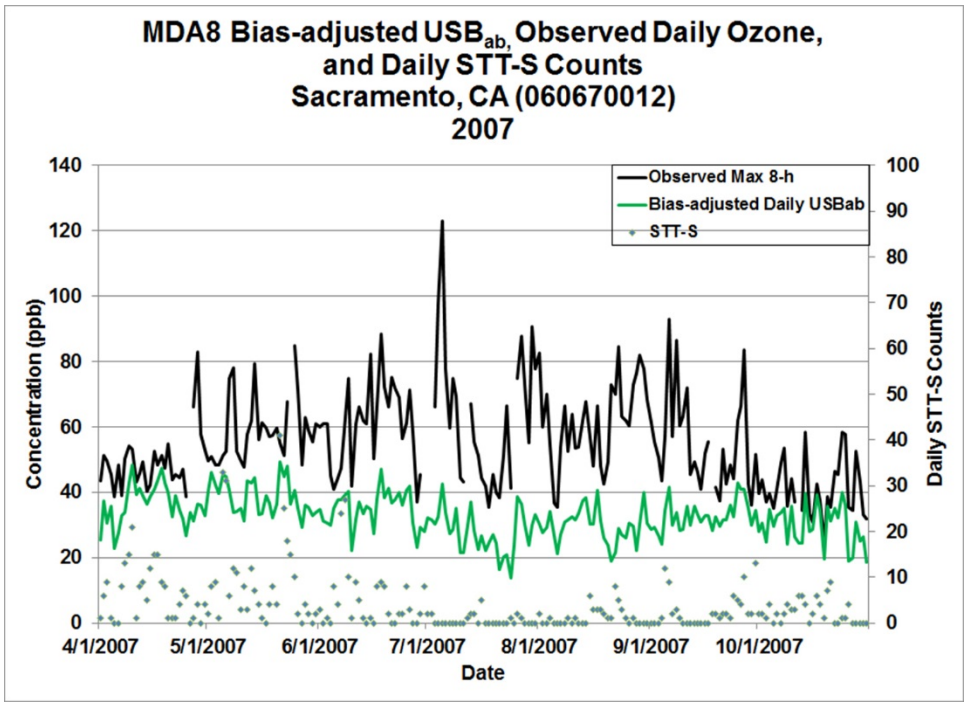



**Figure 3-64. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in Sacramento, California (AQS ID 060670012) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

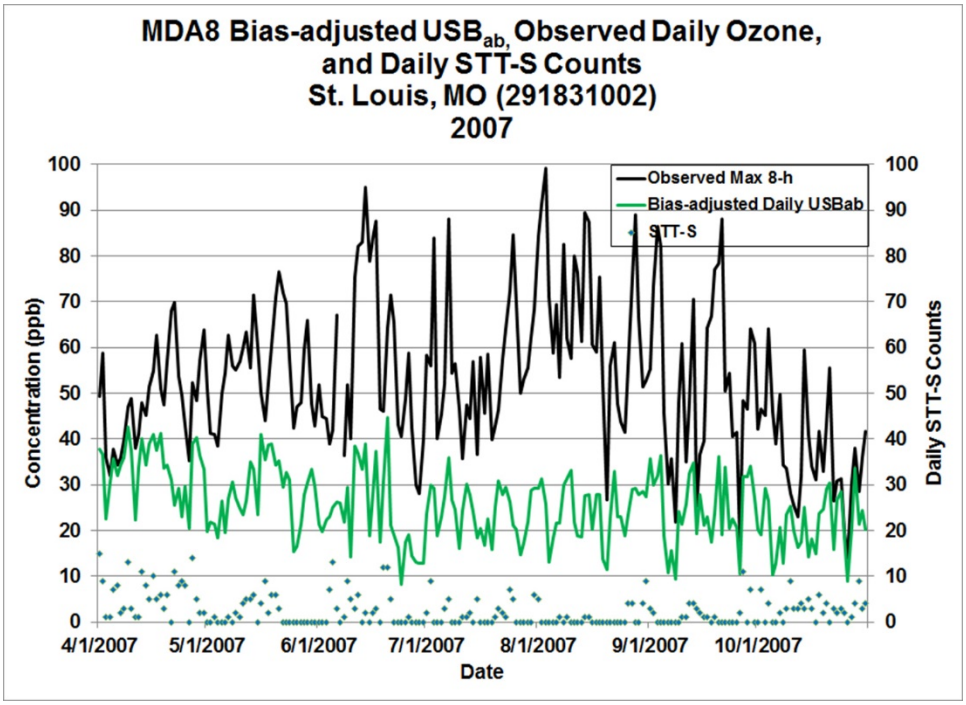



**Figure 3-65. A comparison of the observed 8-h daily maximum concentration with the estimated bias-adjusted U.S. Background ($USB_{AB}$) 8-h daily maximum concentrations for a site in St. Louis, Missouri (AQS ID 291831002) for April-October 2007. The daily stratospheric-tropospheric transport to surface (STT-S) trajectories, as estimated by Professor Heini Wernli, Institute for Atmospheric and Climate Science, ETH Zurich, Switzerland, are overlaid with the daily $O_3$ 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.**

For many of the sites (Figs. 3-59 - 3-65), higher spring $USB_{AB}$ $O_3$ was followed by lower values during the summer, which was then followed by rising $USB_{AB}$ $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ should be higher than $USB_{AB}$. This is because the USB estimates are defined as the $O_3$ concentration that would occur if all U.S. anthropogenic $O_3$ precursor emissions were removed (ISA, 2020a, Page ES-3). While USB is an estimate of $O_3$ concentrations that could be achieved if all U.S. anthropogenic sources were eliminated, $USB_{AB}$ is an estimate of how much $O_3$ can be attributed to background sources when those anthropogenic sources are still present. Background

$O_3$ 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 $O_3$. 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 $O_3$ 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 $O_3$ 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 $O_3$ in the CAMx tracer species used to track the background $O_3$ contribution.

As noted in earlier sections, the USB modeling results described in the draft PA (EPA, 2022) indicate the following seasonal patterns: (1) natural and USA $O_3$ contributions peak during the traditional $O_3$ season (May through September), while long-range intercontinental transport of international $O_3$ (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 $O_3$ 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 $O_3$ 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 $O_3$ modeling results in the draft PA (EPA, 2022) from previous reported patterns for background $O_3$.

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 draft PA (EPA, 2022) 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 draft PA (EPA, 2022). The draft PA (EPA, 2022) 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 $O_3$ 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 $O_3$. There continues to be strong evidence, as supported in the literature, as well as EPA's own analyses (e.g., EPA, 2014a), that background $O_3$ across the U.S. is highest at many sites during the springtime (including into the month of June) and background $O_3$ 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 $O_3$ 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 $O_3$ concentrations increase their percentage in the observed total $O_3$ concentration with the result that the compressed distribution of hourly average concentrations begins to resemble the distribution of background $O_3$. Hopefully, it one were to apply a bias adjustment to the estimated modeled USB estimates described in the draft PA (EPA, 2022), patterns would result that resemble the background $O_3$ 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 draft PA (EPA, 2022), then further model sensitivity analyses should be undertaken. The patterns described earlier that are 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 $O_3$ 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 $O_3$ standard.

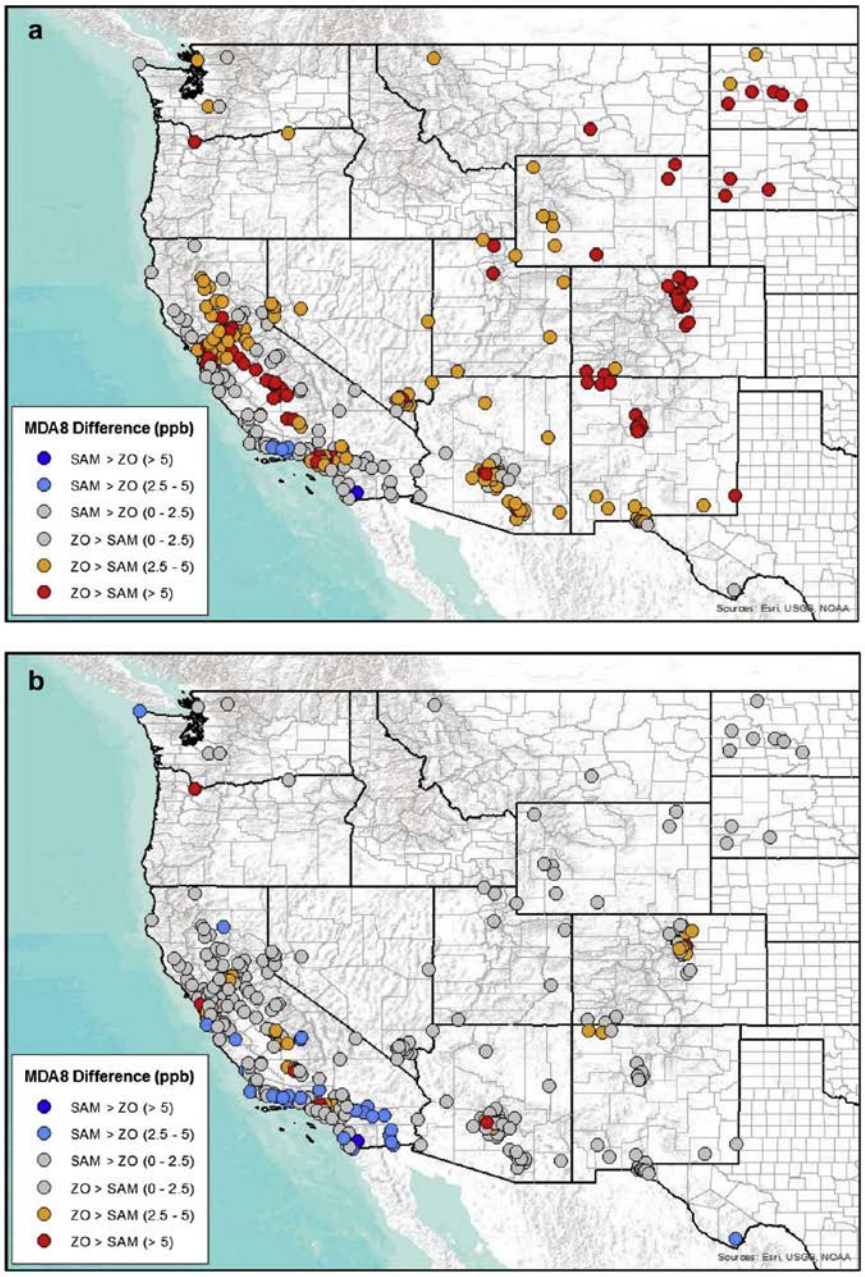



**Fig. 5.** Difference (ppb) in unadjusted April–October mean MDA8 USB ozone vs mean MDA8 $USB_{AB}$ ozone at monitoring locations across the western U.S. Brighter colors indicate sites where zero out (ZO) estimates of USB exceed source apportionment (SAM) estimates of $USB_{AB}$. b. Difference (ppb) 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).**

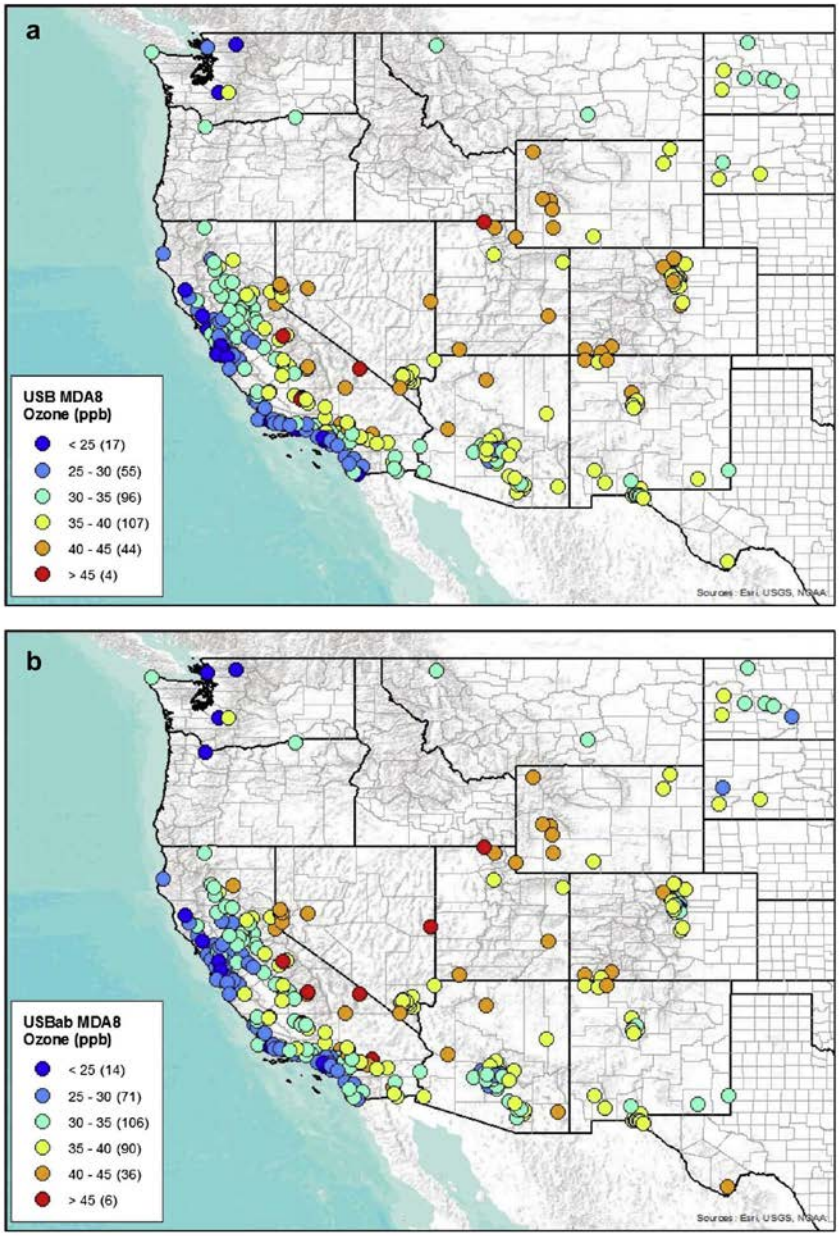



**Fig. 6.** April–October mean bias-adjusted USB MDA8 ozone (ppb) at monitoring locations across the western U.S., as estimated by a 2007 CMAQ zero out simulation. b. April–October mean bias-adjusted $USB_{AB}$ MDA8 ozone (ppb) at monitoring locations across the western U.S., as estimated by a 2007 CAMx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

**Figure 3-67. April-October mean bias-adjusted USB MDA8 $O_3$ (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 $O_3$ (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 $O_3$ and the W126 Exposure Metric

The Lapina et al. (2014) analysis is summarized in the draft PA (EPA, 2022, pages 2-32, 2-34, 2-35, 2-66) for discussing the contribution for background $O_3$ 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 $O_3$ 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 $O_3$ in influencing the W126 cumulative exposures, the estimates performed in the 2014 PA (EPA, 2014a) may provide more insight on the importance of background $O_3$ 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 $O_3$ on annual W126 levels in four locations. Fig. 3-70 (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 $O_3$ 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 $O_3$ 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 the seasonal mean MDA8 metric.

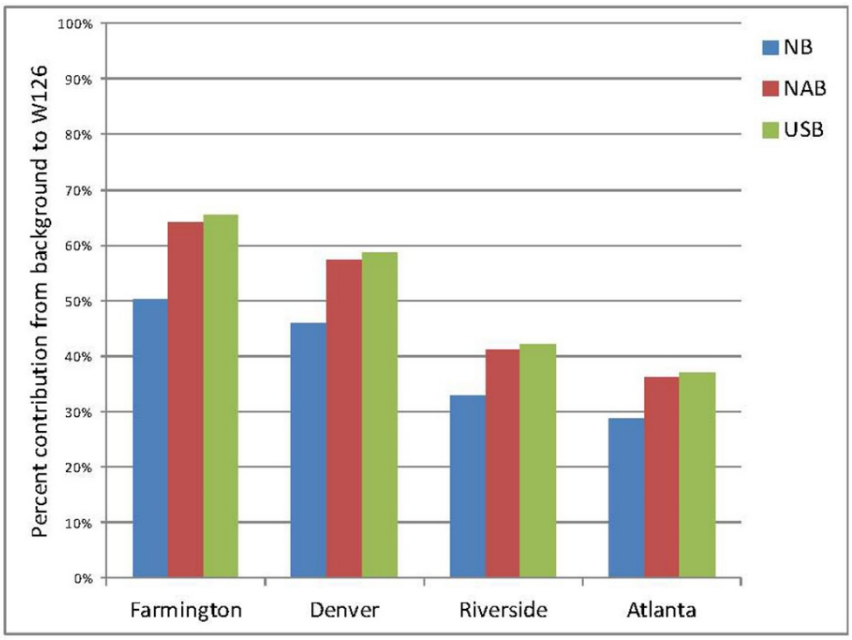



**Figure 2-16. Fractional influence of background sources to W126 levels in four sample locations. Model estimates based on 2007 CMAQ zero-out modeling.**

**Figure 3-70. 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 $O_3$ concentrations are *reduced*, and at many sites the lowest concentrations are *increased* (due to less NO titration of $O_3$ as $NO_x$ emissions are reduced). Simon et al. (2015) and Sicard et al. (2016) reported more ambient increases in low $O_3$ concentrations during winter, the time of year where less NO titration of $O_3$ is most likely to be favored, than during the summer months when $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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* $O_3$ during lockdown, while other researchers reported *decreasing* $O_3$. As will be discussed below, there are possible reasons responsible for explaining the increases and decreases in $O_3$ 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 $O_3$. Compared to the same period in 2017-2019, the daily $O_3$ 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 $O_3$ concentrations was mainly explained by an unprecedented reduction in $NO_x$ emissions leading to a lower $O_3$ 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 $NO_2$, $PM_{2.5,}$ $O_3$, 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 $O_3$ concentrations and nighttime $NO_3$ 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 $O_3$ in winter, reduced nitrogen oxides resulted in $O_3$ 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 ($PM_{10}$, $PM_{2.5}$, $SO_2$, $NO_2$, CO, $O_3$, and $NH_3$) 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 $PM_{10}$ 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 $PM_{10}$ and $PM_{2.5}$ was as high as about 60% and 39%, respectively. Among other pollutants, $NO_2$ (−52.68%) and CO (−30.35%) level 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 $O_3$ increased in the industrial and transport dominated locations (>10% increase). Mahato et al. (2020) noted that the cause for the increase in $O_3$ concentration, especially in the industrial and transport dominated areas, was the decrease of NO, which led to the lowering of the $O_3$ consumption (titration, $NO + O_3 = NO_2 + O_2$) and caused an increase in $O_3$ 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 $O_3$ 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. 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 $NO_2$ showed reductions, while $PM_{10}$ 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 $O_3$ concentrations increased probably due to the decrease in the nitrogen oxides level. When compared to the same period of 2019, $NO_2$ 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 $NO_2$, $O_3$, BC, $PM_{2.5}$, and $PM_{10}$ 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 $NO_2$ of only 34–57% and a reduction in BC of 55–75%. While these two pollutants were reduced, $O_3$ concentrations increased. The authors noted that a lesser increase in $O_3$ would be anticipated due to the already low background levels of $O_3$ observed in the Southern Hemisphere. The observed reductions in $PM_{2.5}$ (still likely to be dominated by traffic emissions), and $PM_{10}$ (dominated by sea salt, traffic emissions to a lesser extent, and affected by seasonality) were found to be significantly less (8–17% for PM2.5 and 7–20% for $PM_{10}$).

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 $NO_2$ 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 $NO_2$ 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 $NO_2$ 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 $NO_2$ 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 $NO_2$ 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 $NO_2$ 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 $NO_2$ 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. 3-71 is presented using Method 1. Note that the largest decreases in $NO_2$ 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. 3-71 illustrates the column $NO_2$; in rural areas, column $NO_2$ changes may be de-coupled to near-surface $NO_2$ changes.

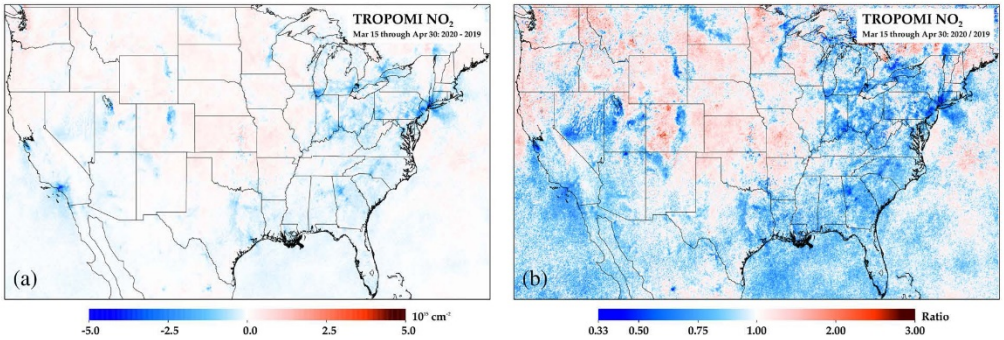



**Figure 2.** TROPOMI $NO_2$ differences between 2019 and 2020, using 15 March to 30 April 2020 as the post-COVID-19 period. Plots are showing (a) the absolute difference and (b) the ratio between years.

**Figure 3-71. TROPOMI $NO_2$ 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 $NO_2$ 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 $NO_2$ 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 $PM_{10}$) only occurred in the Northeast and California/Nevada metropolises, where $NO_2$ declined the most, while the changes in $O_3$ concentrations were mixed and relatively minor.

Using data from the EPA databases, Sommer et al. (2020) reported $O_3$ decreases at many U.S. locations. Sommer et al. (2020), in a National Public Radio (NPR) article, noted that $O_3$ 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 $O_3$ concentrations decreased by 15% or less. Fig. 3-72 below, summarizes the results of their analysis. For the areas investigated, $O_3$ 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 $O_3$ levels, while the Portland area experienced small reductions in $O_3$ concentrations. The Northeastern U.S. experienced little change. The Mid-Atlantic states experienced modest declines in $O_3$ levels. In comparing Fig. 3-71, illustrating that the $NO_2$ changes from Goldberg et al. (2020), with Fig. 3-72, it appears that the spatial patterns for $NO_2$ reductions were similar to the spatial patterns of change observed for $O_3$ by Sommer et al. (2020).

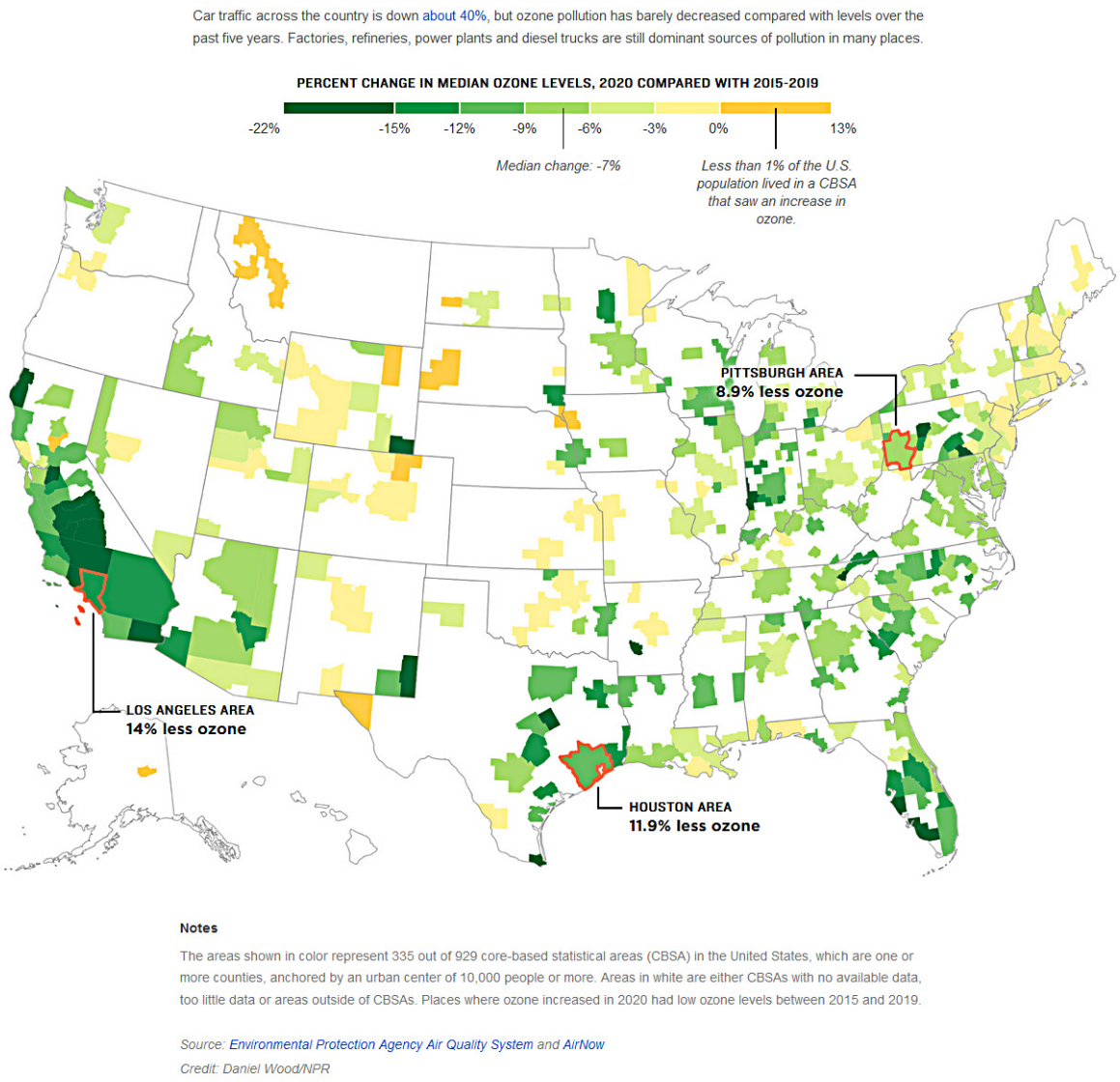



**Figure 3-72. Percent change in median ozone levels of daily maximum 8-h average concentrations, with 2020 spring levels compared with 2015-2019. Source: Sommers et al. (2020). National Public Radio article (May 19, 2020). Traffic Is Way Down Because of Lockdown, But Air Pollution? Not So Much.** https://www.npr.org/sections/health-shots/2020/05/19/854760999/traffic-is-way-down-due-to-lockdowns-but-air-pollution-not-so-much.

Using the same $O_3$ 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. $O_3$ monitoring sites for the period March 15 – April 25, 2020. For most of the 52 monitoring sites, the EPA-defined $O_3$ 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 $O_3$ 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 $O_3$ concentrations during lockdown. Similar to the results reported by Sommer et al. (2020), the Seattle area showed modest declines in $O_3$ levels, while the Portland area experienced small reductions in $O_3$ concentrations. Table 3-14 summarizes for 10 $O_3$ 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 $O_3$ 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 $O_3$ levels. This analysis showed little changes

in $O_3$ 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 $O_3$ 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 $O_3$ consumption (titration, $NO + O_3 = NO_2 + O_2$). As emissions are reduced, Simon et al. (2015) note that in the U.S., increasing $O_3$ 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 $O_3$ distribution. As indicated in this section, the response of $O_3$ 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 $O_3$ 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 $O_3$ concentrations, median concentration hourly average values, the ratio of mean concentrations, and the median $O_3$ levels of the daily maximum 8-h average concentrations. Some of these metrics focus on the lower $O_3$ concentrations, while other metrics focus on the higher $O_3$ concentrations. Using the same hourly average $O_3$ concentration data, Lefohn et al. (2017, 2018) have discussed how the selection of different exposure metrics result in different trend outcomes.

As discussed earlier, as emission reductions occur, the different parts of the distribution of hourly average $O_3$ concentrations shift differently. Both ends of the distribution of hourly average $O_3$ 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 $O_3$ exposures during lockdown compared to previous years, different results would be anticipated. A good question to ask is: "Which $O_3$ exposure metric is most appropriate when comparing differences between exposures during lockdown and other periods?" The answer is that it depends upon that 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 $O_3$ concentrations," one would focus on a metric associated with the higher part of the $O_3$ 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 $O_3$ concentrations were associated greater effects. Alternatively, if one were interested in the chemical phenomena associated with how $O_3$ concentrations change as $NO_x$ concentrations were reduced during lockdown, one might ask the question: "Do the lower hourly average $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ Concentrations Will Remain Well above 0 Parts per Billion (ppb) Even if all Anthropogenic Emissions Were Eliminated Worldwide**.

# 4. The Adequacy of the 4th Highest Daily 8-h Maximum Concentration Exposure Metric to Control the Level of the W126 Exposure Index 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 $O_3$ 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 $O_3$ concentrations results in a nonlinear response to $O_3$ for both human health and vegetation. Information provided in Section 3 indicate that both modeling and empirical data show that as emission reductions occur, the distribution of hourly $O_3$ 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 $O_3$ 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. The information in Sections 2 and 3 establishes an important scientific foundation for the discussion of the secondary $O_3$ NAAQS.

In summarizing the available evidence considering the secondary $O_3$ NAAQS, the draft PA (EPA, 2022, pages 4-120 and 4-121) concludes

> Thus, the available information leads us to conclude that the combined consideration of the body of evidence and the quantitative air quality and exposure analyses, including associated uncertainties, does 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. In reaching these conclusions, we recognize that the Administrator's decisions in secondary standard reviews, in general, are largely public welfare judgments, as described above. We further note that different public welfare policy judgments (e.g., from those in both 2020 and 2015) could lead to different conclusions regarding the extent to which the current standard provides the requisite protection of the public welfare. Such public welfare judgments include those related to the appropriate level of protection that should be afforded to protect against vegetation-related effects of public welfare significance, as well as with regard to the appropriate weight to be given to differing aspects of the evidence and air quality information, and how to consider their associated uncertainties and limitations. For example, different judgments might give greater weight to more uncertain aspects of the evidence or reflect a differing view with regard to public welfare significance. Such judgments are left to the discretion of the Administrator. We note, however, that the scientific evidence and quantitative air quality, exposure and risk information in the record on which this reconsideration

is based are largely unchanged. Staff conclusions regarding the adequacy of the current standards thus remain unchanged from those reached in the 2020 PA.

In summary, the evidence characterized in the 2020 ISA is consistent with that available in the 2015 review for the principal effects for which the evidence is strongest (e.g., plant growth, reproduction, and related larger-scale effects, as well as visible foliar injury) and for key aspects of the current standard. The evidence regarding RBL and air quality in areas meeting the current standard does not appear to call into question the adequacy of public welfare protection afforded by the standard. With regard to visible foliar injury, the currently available evidence for forested locations across the U.S., such as studies of USFS biosites, does not indicate an incidence of significant visible foliar injury that might reasonably be concluded to be adverse to the public welfare under air quality conditions meeting the current standard. For the insect related effects that the ISA newly concludes likely to be causally related to $O_3$, the new information does not support an understanding of the potential for the occurrence of such effects in areas that meet the current standard to an extent that they might reasonably be judged significant to public welfare. Thus, we do not find the current information for these newly identified categories to call into question the adequacy of the current standard. Similarly, key uncertainties recognized in the 2015 review remain in the evidence for $O_3$ contribution to radiative forcing or effects on temperature, precipitation and related climate variables, including specifically uncertainties that limit quantitative evaluations that might inform consideration of these effects (as discussed above). Based on all of the above considerations, we conclude that the currently available evidence and quantitative exposure/risk information does not call into question the protection afforded by the current secondary standard, such that it is appropriate to consider retaining the current standard without revision. In light of this conclusion, we have not identified any potential alternative standards for consideration.

It is important to review the evidence that led to the EPA's decision in the draft PA (EPA, 2022) to not call into question the retaining of the current welfare $O_3$ standard without revision. The pages that follow review the evidence and provide detailed comments on this recommendation.

The draft PA (EPA, 2022, page 4-1) notes that

As a result of the $O_3$ NAAQS review completed in 2015, the level of the secondary standard was revised to 0.070 ppm, in conjunction with retaining the indicator ($O_3$), 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 $O_3$, conducted both in and outside of the U.S., that documents the impacts of $O_3$ 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 index[1]) that may be allowed by such a standard (80 FR 65292, October 26, 2015).

The draft PA (EPA, 2022, page 4-2) describes the definition of the W126 index in footnote 1 as

> The W126 index is a cumulative seasonal metric described as the sigmoidally weighted sum of all hourly $O_3$ concentrations during a specified daily and seasonal time window, with each hourly $O_3$ 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 draft PA (EPA, 2022, 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 $O_3$ 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 $O_3$ effects (80 FR 65389, October 26, 2015; 73 FR 16496, March 27, 2008; 85 FR 87339-41, December 31, 2020).

The draft PA (EPA, 2022, 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). 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, 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 concluded it was 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 $O_3$ 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 draft PA (EPA, 2022, 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 draft PA (EPA, 2022, page 4-7) notes that

In 2020, as in 2015, the Administrator considered the available information regarding the appropriate $O_3$ 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

concentrations[8] (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 draft PA (EPA, 2022, 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 draft PA (EPA, 2022, 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022), 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 draft PA (EPA, 2022, 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 draft PA (EPA, 2022, page 4-9) 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 $O_3$ on plants.

The draft PA (EPA, 2022, 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 above points introduced by the EPA in its draft PA (EPA, 2022), it is important to review the historical development of the W126 exposure index and the intent to which the exposure metric was proposed to be applied for the protection of vegetation.

## 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 $O_3$ concentrations were more important than the mid-and low-level values, the W126 exposure index was created by the author of these comments. The W126 exposure index is a non-threshold metric that is described as the sigmoidally weighted sum of all hourly $O_3$ values observed during a specified daily and seasonal time window, where each hourly $O_3$ value is given a weight that increases from zero to one with increasing value. Lefohn and Runeckles (1987) proposed the use of a 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 exposure-response relationships. The W126 exposure index has the form: $W126 = \Sigma\, w_i \times C_i$ 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 $O_3$ concentration in units of ppb. The M and A constants were derived based on the desire to weight the hourly average levels (1) at a value of one at ≥ 100 ppb and (2) at extremely low values below 40 ppb. The weighting value of "one" for all hourly average $O_3$ 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 $O_3$ concentrations were mostly associated with these levels (EPA, 2006). As is recognized today, hourly average concentrations associated with background $O_3$ 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). The weightings applied to the hourly average $O_3$ concentrations for the calculation of the W126 exposure index are illustrated in Fig. 4-1. The form of the W126 exposure metric applies the experimental evidence that the higher $O_3$ 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 relationship between the higher part of a distribution of hourly average $O_3$ concentrations compared to the middle and lower part of the distribution. It is important to note that 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 $O_3$ 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 $O_3$ concentrations can contain or not contain "peak" hourly average concentrations. The term "peak" hourly average concentrations for the purpose of this discussion refers to hourly average values ≥ 100 ppb. Based on the discussion in the draft PA (EPA, 2022), if it is found experimentally that a specific $O_3$ regime, for example one that does not experience "high peak concentrations," is responsible for eliciting an important adverse effect on vegetation, then it is assumed that the EPA (as a

policy decision implemented by the Administrator) would employ a secondary $O_3$ NAAQS that is protective of such exposure regimes.

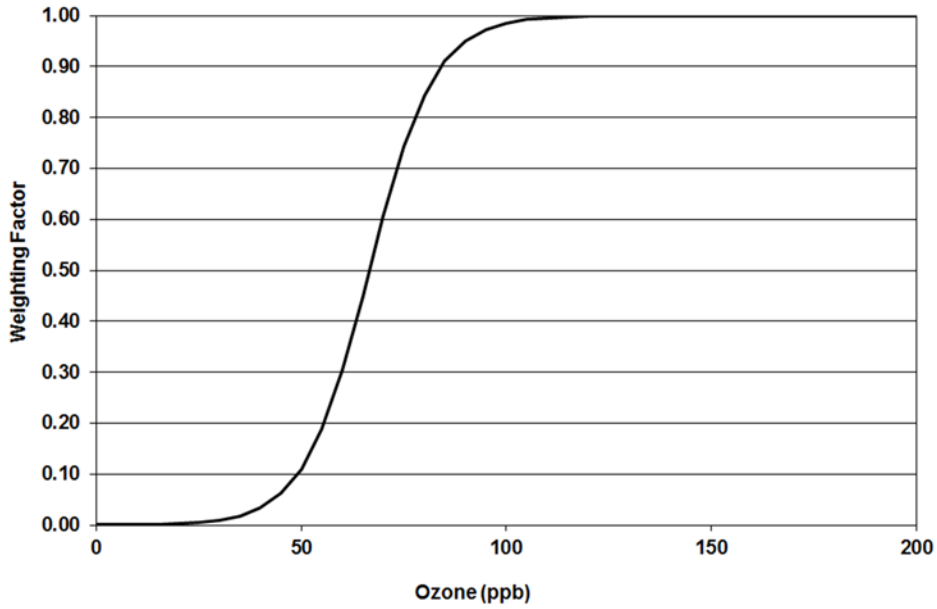



**Figure 4-1. The weighting applied to hourly average ozone values for the calculation of the W126 exposure index (see Lefohn et al., 1988).**

In the draft PA (EPA, 2022, page 4-84), 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*** (*emphasis added*) and those without such concentrations.[88] 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** (*emphasis added*) that can be of concern.

Footnote 88 in the draft PA (EPA, 2022, page 4-84 states

> [88] 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 draft PA (EPA, 2022, page 4-85) notes

> Accordingly, in our consideration of the potential for vegetation-related effects to occur under air quality conditions associated with the current standard, we continue to focus on the W126 index as the appropriate metric, while also being aware of the importance of considering the occurrence and frequency of particularly high concentrations. We also recognize that this metric may not well describe the key circumstances of $O_3$ exposure for occurrences of 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., 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 also recognize the need for attention to the pattern and magnitude of peak concentrations.

As noted above, the W126 exposure index is a cumulative seasonal metric described as the sigmoidally weighted sum of all hourly $O_3$ concentrations during a specified daily and seasonal time window, with each hourly $O_3$ 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 $O_3$ concentrations during a specified daily and seasonal time window. Although the weighting scheme for the W126 metric (Fig. 4-1) allows for all hourly average $O_3$ concentrations, as noted previously, it is not a necessary requirement that hourly average $O_3$ concentrations $\geq$ 100 ppb are present for vegetation damage to occur. The weighting scheme indicates that within any distribution of hourly average $O_3$ concentrations that occur during a specified daily and seasonal time window, the higher hourly average $O_3$ concentrations during this defined period are weighted more than the middle and lower values. Throughout EPA's discussion in the draft PA (EPA, 2022), the EPA has emphasized its concern about the occurrence of hourly average $O_3$ concentrations $\geq$ 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 $O_3$ values as "high peak concentrations." As indicated previously, there is no requirement that hourly average $O_3$ concentrations $\geq$ 100 ppb must occur to elicit adverse vegetation effects. What is most important is the identification of those $O_3$ exposure regimes responsible for causing adverse vegetation effects. In some cases, the number of occurrences of hourly average $O_3$ concentrations $\geq$ 100 ppb may be minimal (e.g., less than 5 occurrences) or perhaps even zero.

In its discussion of the secondary standard, the draft PA (EPA, 2022) 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 many of the treatments experiencing elevated $O_3$ 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. Furthermore, some of the exposure treatments using the NCLAN protocols mimicked $O_3$ exposures experienced under ambient conditions. Lefohn and Foley (1992) noted that when high hourly average concentrations were present in an exposure regime, single-parameter cumulative indices appeared to relate $O_3$ exposures with vegetation growth reductions. However, as noted by the authors, if high hourly average concentrations were not necessarily present, 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 $O_3$ concentrations $\geq 0.10$ ppm in their 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 values $\geq 0.10$ ppm. Similar to Lefohn and Foley (1992), the authors were concerned if ambient data were used to predict growth losses using exposure-response relationships derived from experimental data that contained numerous occurrences of hourly average $O_3$ concentrations $\geq 0.10$ ppm, biomass loss might possibly be overestimated. Fig. 4-2, 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 $\geq$ 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 $\geqslant 0.10$ ppm |
|---|---|---|---|
| Minimal | $\geqslant 0$ | and | $\geqslant 0$ |
| Level 1 (only high sensitive species affected) (e.g. black cherry) | $\geqslant 5.9$ | and | $\geqslant 6$ |
| Level 2 (moderately sensitive species affected) (e.g. yellow-poplar) | $\geqslant 23.8$ | and | $\geqslant 51$ |
| Level 3 (resistant species affected) (e.g. red oak) | $\geqslant 66.6$ | and | $\geqslant 135$ |

**Figure 4-2. The ozone W126 and N100 24-h exposure levels as 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 ≥ 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 $O_3$ 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 $O_3$ 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 $O_3$ uptake at night may interfere with recovery and the FCPC believed that this evidence should be considered in establishing an appropriate time period for accumulation. Vegetation growing in remote, high-elevation, and rural areas is more likely to experience some conductance, enhanced $O_3$ 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 $O_3$ enhancing surface $O_3$ 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 $O_3$ in the spring associated with stratospheric $O_3$ in the FCPC Class I area or other areas in the region. However, by minimizing increases in $O_3$ exposures from anthropogenic sources, the FCPC believed that vegetation foliar injury and damage could be kept at a minimum.

The draft PA (EPA, 2022, page 4-43) discussed results from Davis and Orendovici (2006). During 1993-1996 and 2001-2003, the investigators evaluated the percentage of plants (incidence) exhibiting $O_3$-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 $O_3$ symptoms, across all plant species, occurred in 1996, followed by 2001 > 1995 > 1994 > 1993 > 2003 > 2002. As noted in the draft PA (EPA, 2022, page 4-43), 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 $O_3$ 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 $O_3$ symptoms in the field.

It is important to note that the draft PA (EPA, 2022, page 4-112) 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 $O_3$ exposures. More specifically, as discussed in the prior sections, the available information suggests that $O_3$ 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 $O_3$ 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 $O_3$ at lower exposures. Developing a reliable relationship between visual foliar injury and $O_3$ exposure has been a difficult challenge for vegetation researchers. Some $O_3$ exposures result in visible foliar injury to the plant without growth reduction; other exposures result in growth reduction and decrease in productivity without visible 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 $O_3$ exposure. The draft PA (EPA, 2022, page 4-82) 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 $O_3$ exposure, as well as factors influential in those relationships, such as soil moisture conditions.

In summary, as noted in the draft PA (EPA, 2022) in Section 4, Lefohn and Foley (1992), Lefohn et al. (1992), and Lefohn et al. (1997) have documented the frequency of the occurrence of elevated $O_3$ concentrations in 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 $O_3$ exposures.

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

As noted in the draft PA (EPA, 2022), the threshold value of 100 ppb has been applied in some studies focused on $O_3$ effects on vegetation as an indicator of peak hourly $O_3$ 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. 4-1) includes all hourly average $O_3$ concentrations, including hourly average $O_3$ concentrations ≥ 100 ppb, it is not a necessary requirement that hourly average $O_3$ concentrations ≥ 100 ppb must be present for vegetation damage to occur. Fig. 4-3 (Fig. 4-7 in the draft PA, page 4-61) illustrates the number of hourly average $O_3$ concentrations ≥ 100 ppb (N100) values at $O_3$ 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. 4-4 (Fig. 4F-13, page 4F-22) illustrates the Trend in N100 values from 2000 to 2020 based on data from 808 U.S. $O_3$ monitoring sites.

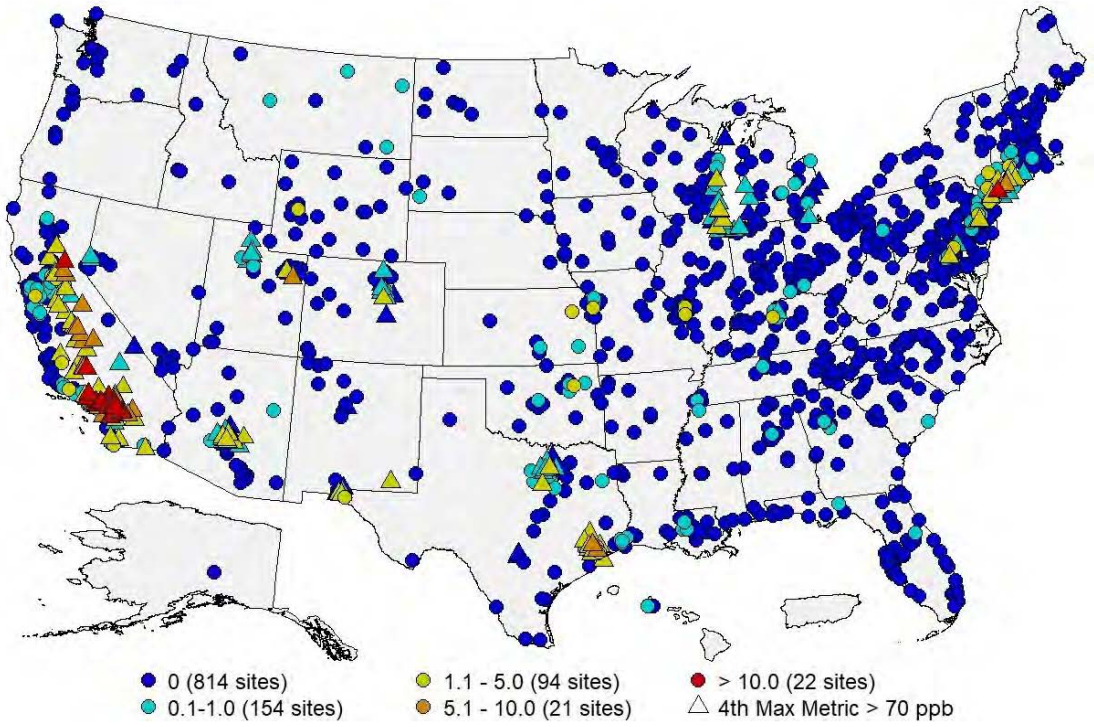



**Figure 4-7. N100 values at monitoring sites with valid design values (2018-2020 average).**

**Figure 4-3. N100 values at monitoring sites with valid design values (2018-2020 average). (EPA, 2022, page 4-61).**

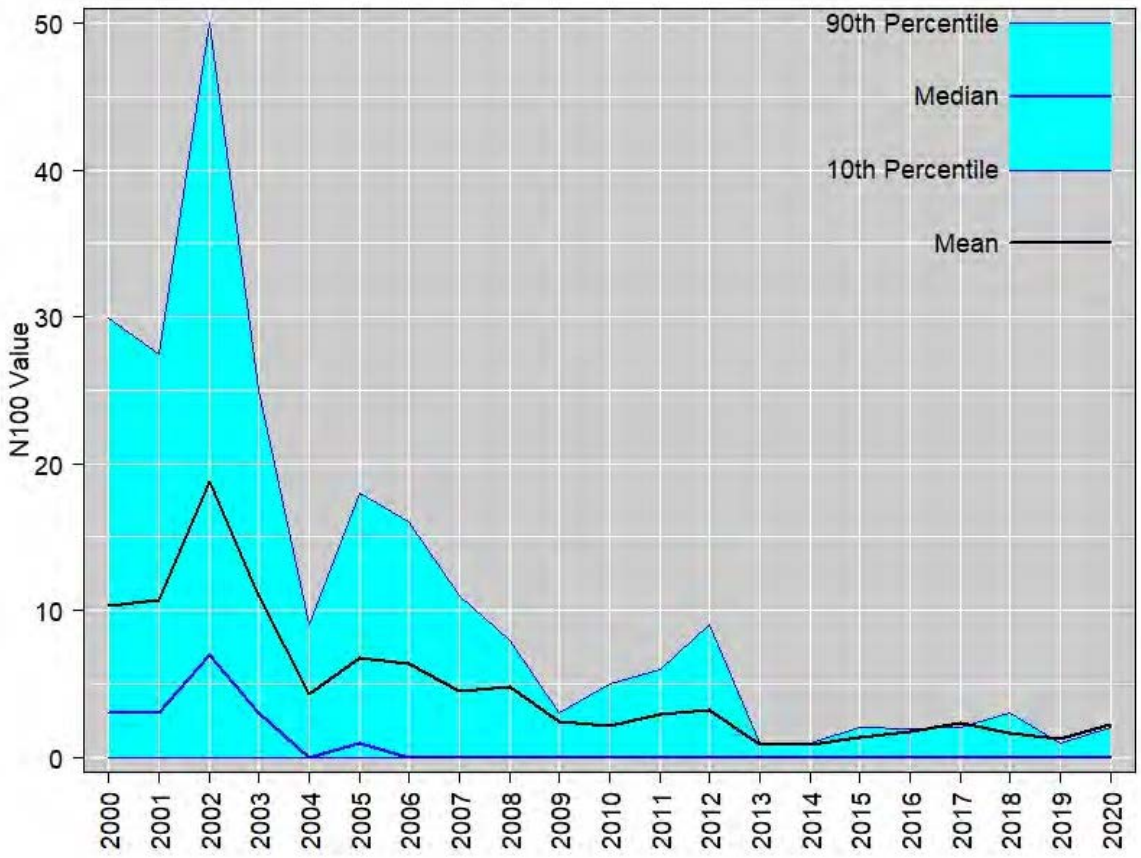



**Figure 4F-13. Trend in N100 values from 2000 to 2020 based on data from 808 U.S. $O_3$ monitoring sites**

**Figure 4-4. Trend in N100 values from 2000 to 2020 based on data from 808 U.S. $O_3$ monitoring sites. (EPA, 2022, page 4F-22).**

As the draft PA (EPA, 2022, 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 even a faster rate. Thus, in recent years, a little less than 75% of the monitors included in the draft PA (EPA, 2022) analysis experience no N100 values.

The draft PA (EPA, 2022, page 4F-3) notes that the W126 metric values were calculated using the hourly $O_3$ 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 $O_3$ monitoring site were weighted using the following equation:

$$\text{Weighted } O_3 = O_3 / (1 + 4403*\exp(-126 * O_3)).$$

The weighted values were summed over each calendar month, then adjusted for missing data as described in the draft PA (EPA, 2022, 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. It appears from the description of the W126 calculation presented in the draft PA (EPA, 2022) that 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 draft PA (EPA, 2022, page 4F-3) notes that the N100 values were calculated as the maximum number of hours with an hourly $O_3$ concentration of 100 ppb or greater in the three consecutive calendar months yielding the highest number each year.

As noted in Section 3, actual data show that the highest $O_3$ exposures occur at some sites across the U.S. during the springtime (March to mid-June). Using hourly average $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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. 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.

While the EPA in the draft PA (EPA, 2022) 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 $O_3$ exposures affecting tree seedling biomass loss. The vegetation may not be as susceptible to $O_3$ biomass loss as during the warmer months. The observations reported by the EPA (2014c) and Neufeld et al. (2019) may suggest that the EPA limit its characterization of W126 and N100 values to those periods when $O_3$ exposures have their greatest impacts on tree seedling biomass loss. As indicated in the draft PA (EPA, 2022, pages 4A-17 – 4-21), most of the tree seedling experiments began in June and were terminated in September/October (please see Table 4A-6 in the draft PA).

**Table 7A-2 Ozone Exposure in 57 $O_3$ 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 |



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

7A-13

Source: EPA (2014c).

The use of the W126 to characterize $O_3$ 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 draft PA (EPA, 2022, pages 1-9 – 1-11) as follows:

> The EPA initiated the fourth periodic review of the air quality criteria and standards for $O_3$ 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 $O_3$ standards. On September 16, 2009, the EPA announced its intention to reconsider the 2008 $O_3$ 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 $O_3$ 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 $O_3$ 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.

At the time of the court's decision, the EPA had already completed significant portions of its next statutorily required periodic review of the $O_3$ 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 PA[12] 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 $O_3$ 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 $O_3$ effects on welfare, particularly the evidence of $O_3$ 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 requirements[14] (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 $O_3$ 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 draft PA (EPA, 2022, page 1-12), the EPA announced its initiation of the periodic review of the air quality criteria for photochemical oxidants and the $O_3$ 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 proposed to retain both the primary and secondary $O_3$ standards, without revision. In December 2020, the EPA 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 $O_3$ exposures and (2) applied the current 8-h form of the $O_3$ NAAQS to control for those W126 values and peak hourly concentrations associated with vegetation effects.

Fig. 4-5, which is a reproduction of Table 4A-3 in the draft PA (EPA, 2022), shows a summary of the relative biomass loss for eleven individual tree seedlings and median at various W126 index values identified in the draft PA (EPA, 2022, page 4A-12). Fig. 4-5 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-3. 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% | 0.8% | 1.7% | 3.8% | 28.1% | 10.4% | 12.8% | 18.6% | 27.7% | 25.2% | 53.8% | 12.8% | 3 | 4 | 4 | 6 |
| 29 | 0.0% | 0.7% | 1.7% | 3.6% | 23.7% | 10.0% | 12.3% | 17.9% | 26.1% | 24.0% | 52.6% | 12.3% | 3 | 4 | 5 | 6 |
| 28 | 0.0% | 0.7% | 1.6% | 3.5% | 19.9% | 9.6% | 11.8% | 17.2% | 24.5% | 22.8% | 51.4% | 11.8% | 3 | 4 | 5 | 6 |
| 27 | 0.0% | 0.7% | 1.6% | 3.3% | 16.4% | 9.2% | 11.4% | 16.5% | 23.0% | 21.6% | 50.1% | 11.4% | 3 | 4 | 5 | 6 |
| 26 | 0.0% | 0.7% | 1.5% | 3.1% | 13.4% | 8.8% | 10.9% | 15.8% | 21.4% | 20.5% | 48.8% | 10.9% | 3 | 4 | 5 | 7 |
| 25 | 0.0% | 0.6% | 1.4% | 3.0% | 10.9% | 8.4% | 10.4% | 15.2% | 19.9% | 19.3% | 47.5% | 10.4% | 3 | 4 | 5 | 7 |
| 24 | 0.0% | 0.6% | 1.4% | 2.8% | 8.7% | 8.0% | 10.0% | 14.5% | 18.4% | 18.2% | 46.2% | 8.7% | 3 | 4 | 7 | 8 |
| 23 | 0.0% | 0.6% | 1.3% | 2.7% | 6.9% | 7.6% | 9.5% | 13.8% | 17.0% | 17.1% | 44.8% | 7.6% | 3 | 4 | 7 | 8 |
| 22 | 0.0% | 0.6% | 1.3% | 2.5% | 5.3% | 7.2% | 9.0% | 13.1% | 15.6% | 15.9% | 43.3% | 7.2% | 3 | 4 | 7 | 8 |
| 21 | 0.0% | 0.5% | 1.2% | 2.4% | 4.1% | 6.8% | 8.6% | 12.4% | 14.3% | 14.9% | 41.9% | 6.8% | 3 | 5 | 7 | 10 |
| 20 | 0.0% | 0.5% | 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% | 0.5% | 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% | 0.5% | 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% | 0.4% | 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% | 0.4% | 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% | 0.4% | 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% | 0.4% | 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% | 0.3% | 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% | 0.3% | 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% | 0.3% | 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% | 0.3% | 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% | 0.2% | 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% | 0.2% | 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% | 0.2% | 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 4-5. Relative biomass loss (RBL) estimates for eleven individual tree seedlings and median at various W126 index values (EPA, 2022, page 4A-12).**

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% (EPA, 2022, page 4-5). 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. 4-6 below from the draft PA (EPA, 2022, page 4-63) illustrates the relationship between the W126 metric and the design values for the current 8-h $O_3$ NAAQS over the period 2018-2020. The W126 index is analyzed for both the 3-year design value period, as well as the annual period.

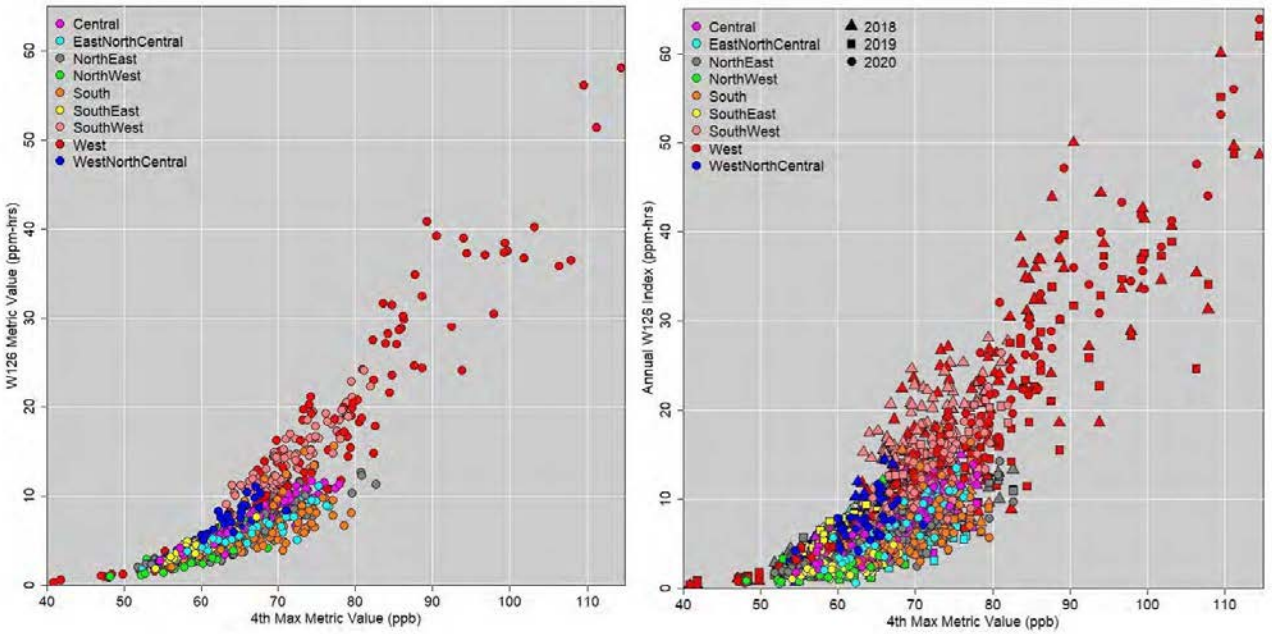



**Figure 4-9. Relationship between the W126 index and design values for the current standard (2018-2020). The W126 index is analyzed in terms of averages across the 3-year design value period (left) and annual values (right).**

**Figure 4-6. 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, 2022, page 4-63).**

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. At the 7 ppm-hrs annual level, Fig. 4-6 (Fig. 4-9 in EPA (2022, page 4-63)), the right side of the figure shows that the range of values for the current form of the $O_3$ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites

meeting the $O_3$ 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. 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 $O_3$ NAAQS. The W126 metric and the 8-h $O_3$ NAAQS each behave differently at levels of 7 ppm-hrs and below. A strategy of using the $O_3$ NAAQS to limit annual W126 values to 7 ppm-hrs and below is questionable.

The draft PA (EPA, 2022, page 4-69) notes that

> In considering the prevalence of peak concentrations occurring at monitoring sites, it can be seen that $O_3$ 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.

Fig. 4-7 is a reproduction of Table 4-1 in the draft PA (EPA, 2022, page 4-69) that illustrates the percent of monitoring sites during the 2018 to 2020 period with $4^{th}$ max or W126 metrics at or below various thresholds that have N100 or D100 values above various thresholds. (EPA, 2022, page 4-69). 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 $4^{th}$ Max $\leq$ 70 ppb.

In summary, based on the above discussions associated with limiting the W126 and N100 values, the $O_3$ 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 $4^{th}$ 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 $O_3$ NAAQS and that a strategy of using the $O_3$ NAAQS to limit annual W126 values to 7 ppm-hrs would be questionable.

**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 |
| | | *Average percent of sites exceeding N100 or D100 threshold per year** | | | | | |
| 3-year 4th 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 4th Max ≤ 70* | 802-1000 | 3.7% | 0% | 0% | 3.7% | 0% | 0% |
| | | *Percent of sites exceeding N100 or D100 threshold in maximum year of the three* | | | | | |
| 3-year 4th 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 4th 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 4th Max or W126 thresholds in each of the three years (as presented in Table 4F-2, Appendix 4F).

**Figure 4-7. 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, 2022, page 4-69).**

Using the 6% RBL criterion noted by the Administrator in 2015, two of the most sensitive species were black cherry and aspen. In the draft PA (EPA, 2022, pages 4A-17 – 4A-21), Table 4A-6 is provided. The table has been updated from the version published in the 2020 PA (EPA, 2020b). The updated version of Table 4A-6 now includes the N100 values associated with each of the tree seedling experiments. The draft PA (EPA, 2022, pages 4A-37 and 4A-38) 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 $O_3$ on which the established E-R functions

for the 11 tree seedling species are based, involved W126 index levels well above 20 ppm-hrs and had many (tens to more than a hundred) of hours of $O_3$ concentrations above 100 ppb (Appendix 4A, Table 4A-6; Lefohn et al 1997). [44] [45]

Footnotes 44 and 45 indicate the following:

[44] 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), extend up above 10, to more than 40. Additionally, in a study that has reported the distributions of hourly concentrations, the 90$^{th}$ percentile in replicates for one of the elevated $O_3$ treatments ranged from 142 to 156 ppb, and the maximum ranged from 210 to 260 ppb (Appendix 4A, Table 4A-6; Lefohn et al 1992).

[45] Similarly, the experimental exposures in studies supporting some of the established E-R functions for 10 crop species also include many hours with hourly $O_3$ concentrations at or above 100 ppb (Lefohn and Foley, 1992).

Table 4A-6 in the draft PA (EPA, 2022, pages 4A-17 – 4A-21) shows that W126 exposures associated with levels approximating 7 ppm-hrs experience less than 7 occurrences of hourly average $O_3$ 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. 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 draft PA (EPA, 2022) does not include the results of Lee et al. (2022) because the authors' paper was not published at the time the draft PA was prepared. However, the paper is an important contribution to the literature, and it is recommended that the authors' results be included in the revision to the current draft PA (EPA, 2022). Lee et al. (2022) classified 16 tree species (11 of these species are described in the draft PA) according to their sensitivity, based on biomass loss response functions to protect from a 5% biomass loss. The seedling $O_3$ exposure studies 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 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 $O_3$ 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. 4-8 – 4-11 (Tables 4B-1, 4B-10, 4B-5, and 4B-6). These four tree species when combined cover a large part of the U.S.

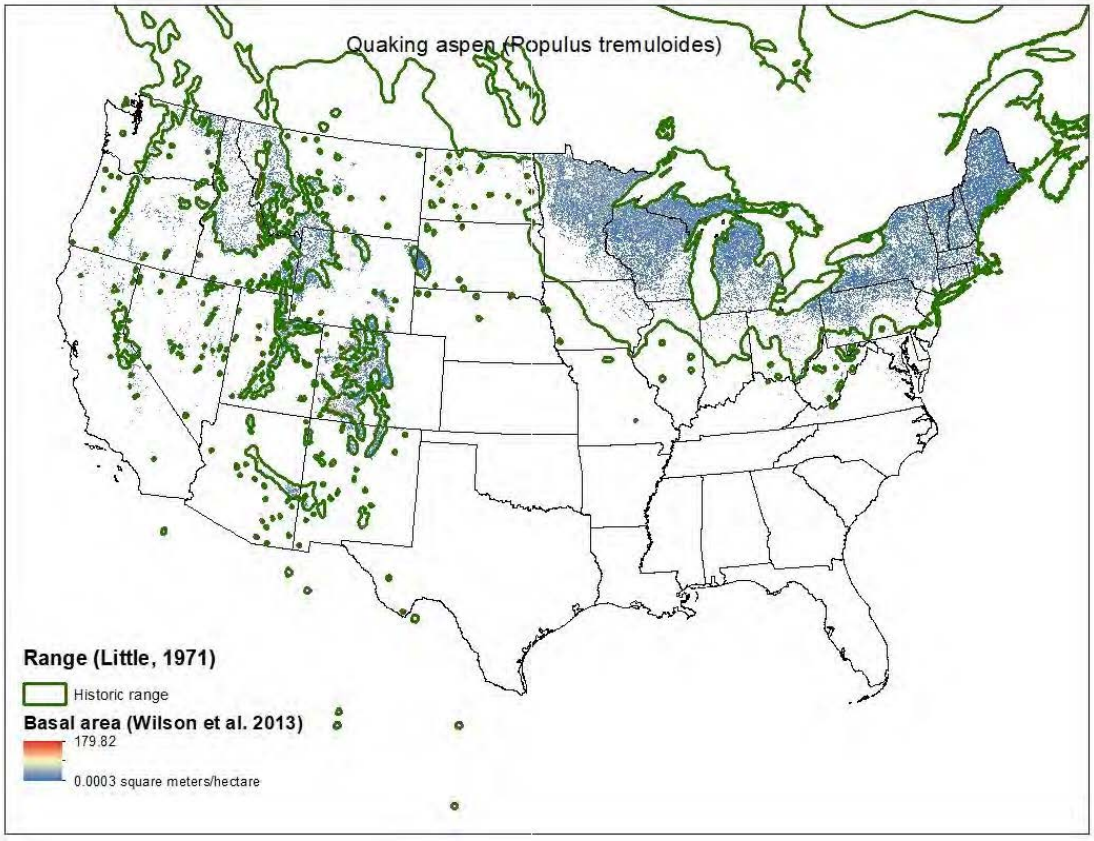



Table 4B-1. Distribution of quaking aspen (*Populus tremuloides*) in the continental U.S.

**Figure 4-8. Distribution of quaking aspen in the continental U.S. (EPA, 2022, page 4B-3).**

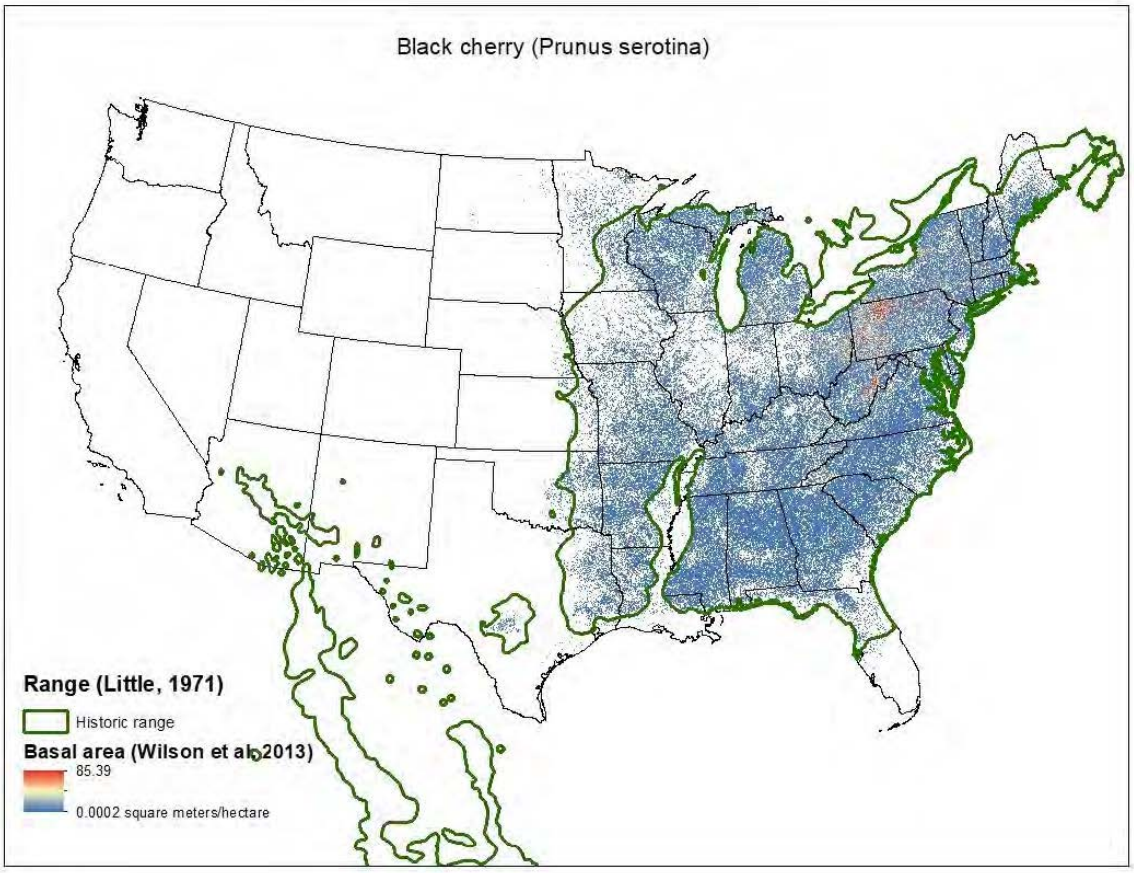



Table 4B-10. Distribution of black cherry (*Prunus serotina*) in the continental U.S.

**Figure 4-9. Distribution of black cherry in the continental U.S. (EPA, 2022, page 4B-12).**

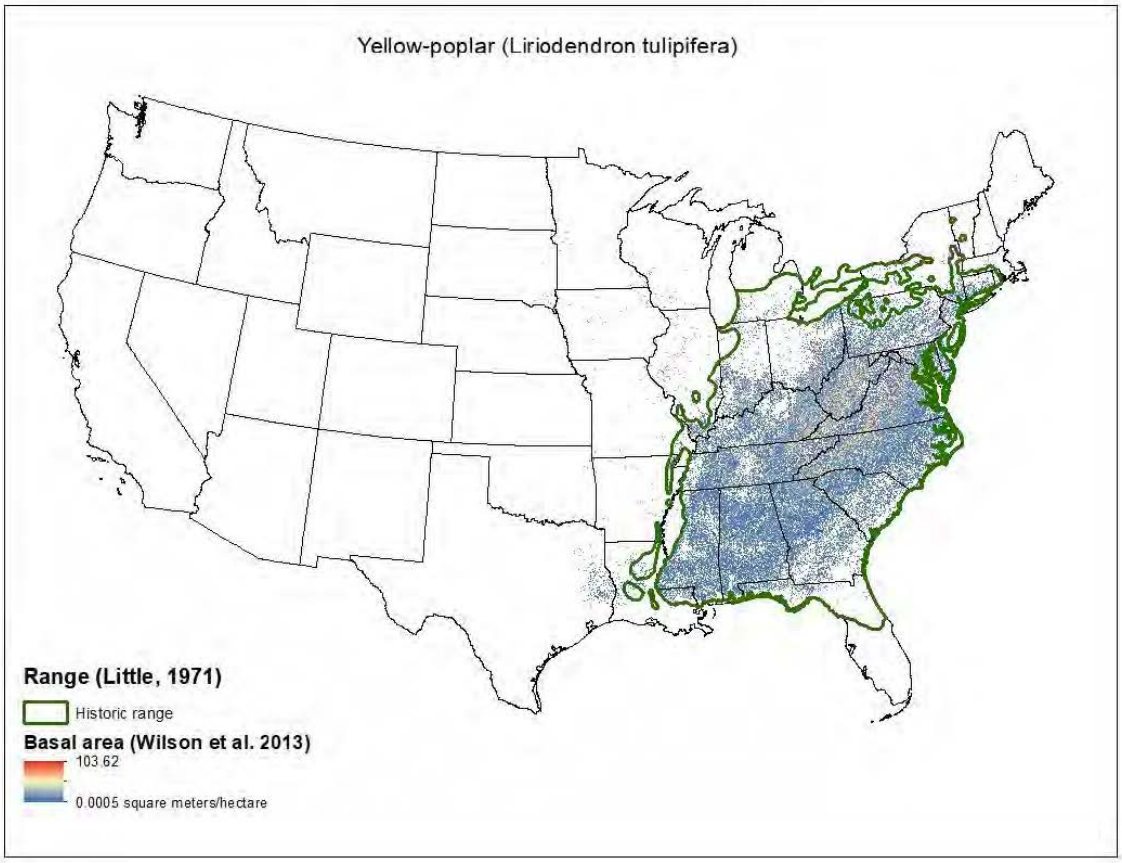



Table 4B-5. Distribution of tulip poplar (*Liriodendrun tulipifera*) in the continental U.S.

**Figure 4-10. Distribution of tulip poplar in the continental U.S. (EPA, 2022, page 4B-7).**

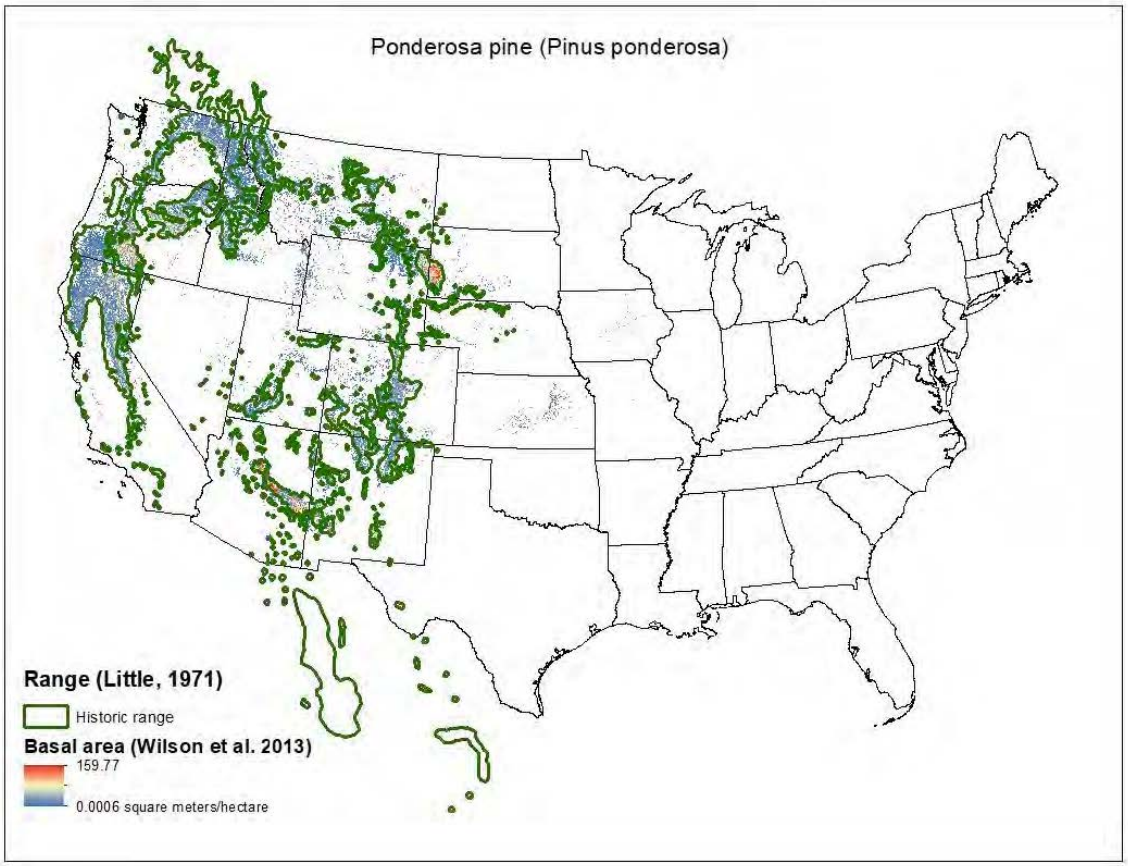



Table 4B-6. Distribution of ponderosa pine (*Pinus ponderosa*) in the continental U.S.

**Figure 4-11. Distribution of ponderosa pine in the continental U.S. (EPA, 2022, page 4B-8).**

Lee et al. (2020) 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 $O_3$ concentrations of 0.10 ppm and higher (N100), because they may play an important role in eliciting a plant response to $O_3$ 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. $O_3$ 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 $O_3$ exposures, especially for the sensitive species.

The frequency of the occurrence elevated $O_3$ 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 $O_3$ exposures. The results of Lefohn and Foley (1992), Lefohn et al. (1992), Lefohn et al. (1997), Davis and Orendovici (2006), and Lee et al. (2022) provide important insight into the application of a unique secondary $O_3$ NAAQS that consists of the combination of (1) 3-month 12-h W126 index (e.g., 7 ppm-hrs) and (2) a specific N100 threshold value (e.g., 4 hours). The dual-parameter form of the $O_3$ 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 as the Secondary $O_3$ NAAQS

In the 2020 decision (Federal Register, 2020), the EPA (1) preferred the use of the W126, as well as peak hourly concentrations of interest, for assessing vegetation risk associated with $O_3$ exposures and (2) applied the current 8-h form of the $O_3$ NAAQS to control for those W126 values and peak hourly concentrations associated with vegetation effects. As noted previously, Lefohn et al. (1997) had identified for the most sensitive tree seedlings in the Southern Appalachian area with 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 $O_3$ 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) 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. 4-6 (Fig. 4-9 in EPA (2022, page 4-63)), the right side of the figure shows that the range of values for the current form of the $O_3$ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the $O_3$ 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 $O_3$ NAAQS each behave differently at levels of 7 ppm-hrs and below. Focusing on the N100 values, the draft PA 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 $4^{th}$ Max ≤ 70 ppb.

Therefore, at a W126 level of 7 ppm-hrs, (1) there does not appear to be a strong relationship between the annual W126 and the current form of the 8-h $O_3$ NAAQS and (2) there does not appear to be an advantage of applying the current 8-h form of the 3-year $4^{th}$ Max $\leq$ 70 ppb to control for the N100 values. Thus, a strategy of using the $O_3$ NAAQS to limit annual W126 values to 7 ppm-hrs (and below) and N100 values to zero appears to be questionable.

## 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 draft PA (EPA, 2022, 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, 2020), 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 $O_3$ concentrations of concern. The results of the air quality analyses first appeared in the Administrator's final decision. The draft PA (EPA, 2022), 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 $O_3$ concentrations which may contribute to vegetation exposures of concern (2020 PA, Appendix 2A, section 2A.2; Wells, 2020).

In response to the Court's decision, the Administrator noted in his $O_3$ NAAQS rulemaking (Federal Register, 2020) 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, 2020, 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 $O_3$ 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 $O_3$ 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 $O_3$ concentrations ≥ 100 ppb). For this additional aspect, the draft PA (EPA, 2022, 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, 2020, 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 $O_3$ NAAQS reviews, the air quality criteria for vegetation effects have emphasized the risk posed to vegetation from higher hourly average $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ on which the established E-R functions for the 11 tree seedling species are based, had hundreds of hours of $O_3$ 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 $O_3$ concentrations at or above 100 ppb (Lefohn and Foley, 1992).

The Administrator continues (Federal Register, 2020, 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 $O_3$ 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 $O_3$ 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 $O_3$ 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, 2020, page 87327), Lefohn et al. (1997) and Lefohn and Foley (1992) identified numerous occurrences of hourly average $O_3$ concentrations ≥ 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 $O_3$ concentrations. At the more moderate treatments, the exposures contained infrequent occurrences of hourly average $O_3$ concentrations (see for example, Table 4A-6 in the draft PA (EPA, 2022, pages 4A-17 – 4A-21). 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 $O_3$ 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 $O_3$ exposures. The Administrator also noted that he preferred to apply the current 8-h form of the $O_3$ NAAQS to control for those W126 and hourly average $O_3$ concentrations ≥ 100 ppb values of concern for the protection of vegetation from $O_3$ exposures.

The draft PA (EPA, 2022, 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 $O_3$ 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 $O_3$ 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) 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 draft PA (EPA, 2022, page 4D-10) comments on the variation of the annual W126 index from the 3-year (2018-2020) W126 index. The draft PA 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. 4-12 (Fig. 4D-6 in the draft PA) and 4-13 (Fig. 4D-7 in the draft PA) illustrate the variation of the annual W126 values from the 2018-2020 average W126 metric values. Both figures illustrate that at the 7 ppm-hrs level for the annual W126 index the deviation can be large relative to the absolute value of the W126 index.

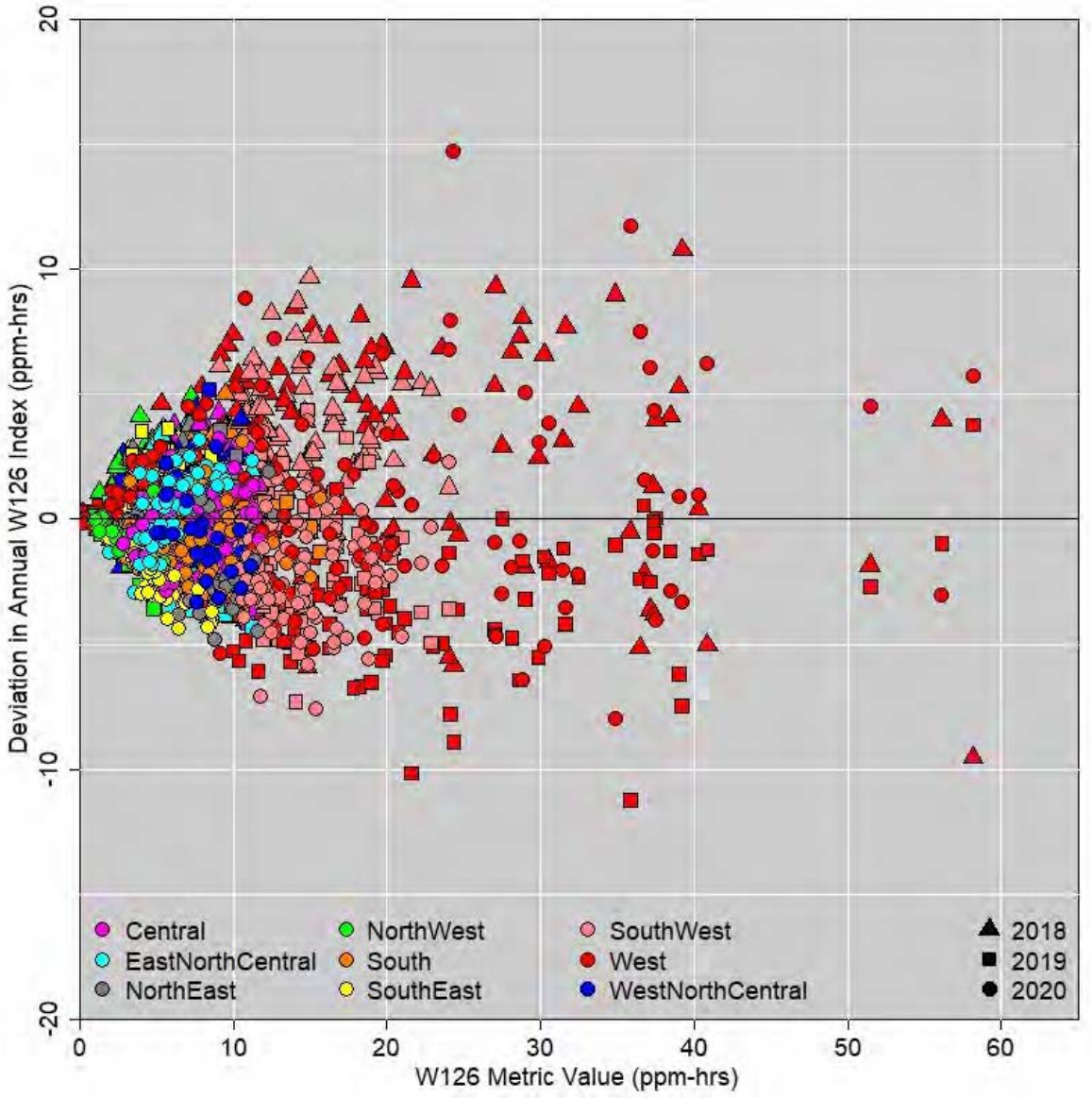



**Figure 4D-6. Deviation in annual W126 index values from their respective 3-year averages for all U.S. monitoring sites in 2018-2020.**

**Figure 4-12. Deviation in annual W126 index value from their respective 3-year averages for all U.S. monitoring sites in 2018-2020. (EPA, 2022, page 4D-14).**

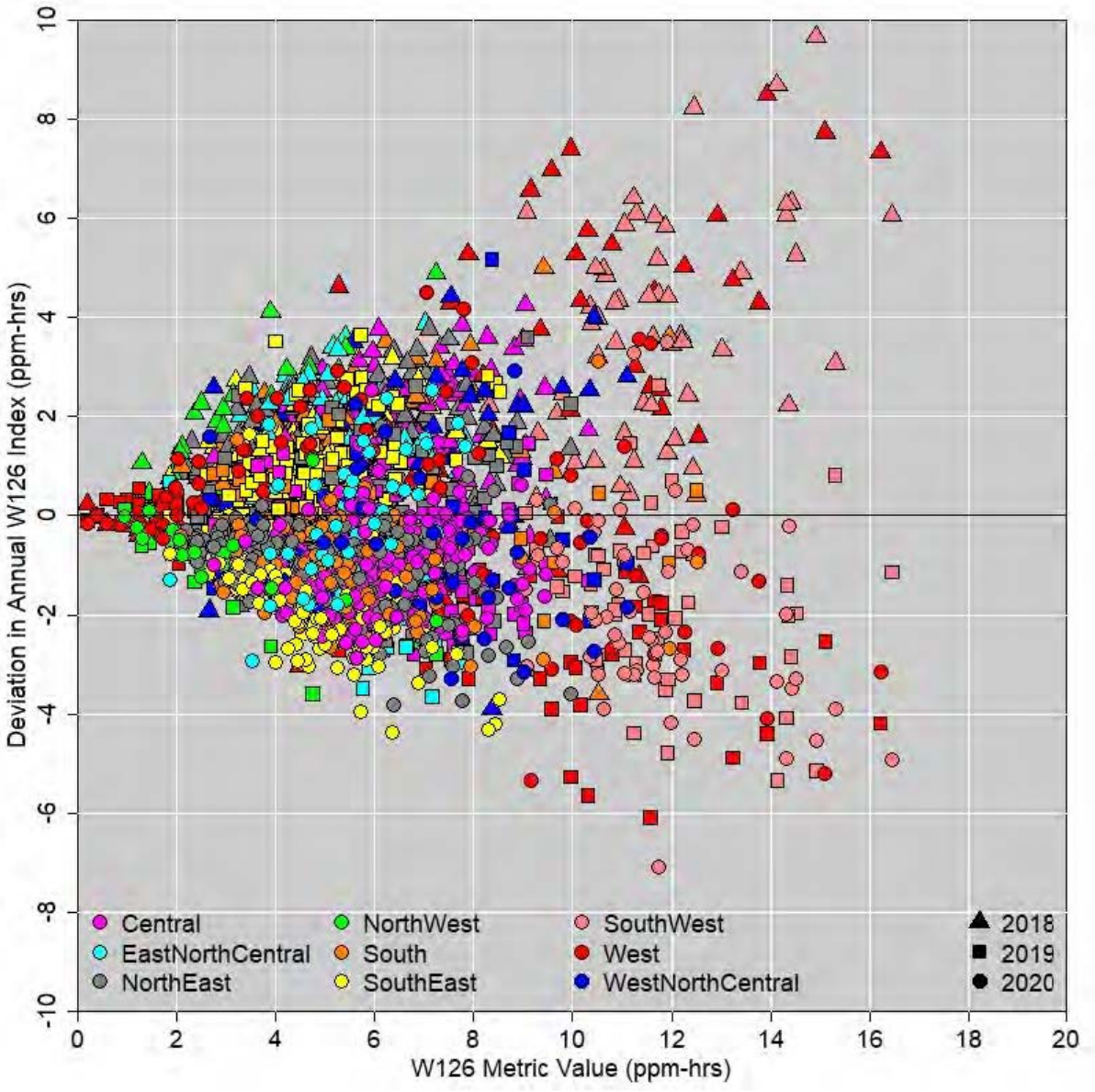



**Figure 4D-7. Deviation in annual W126 index values from their respective 3-year averages for all U.S. monitoring sites with 4th max metric values at or below 70 ppb in 2018-2020.**

**Figure 4-13. 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, 2022, page 4D-15).**

In addition to the deviation analyses for supporting the use of the 3-year versus the annual W126 index, the draft PA (EPA, 2022, page 4-52) refers to 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 $O_3$-related biomass impacts estimated using the established E-R functions for aspen. The draft PA (EPA, 2022, page 4-93

notes that while the E-R functions were based on strong evidence of cumulative seasonal $O_3$ exposure reducing tree growth, and while they provide for quantitative characterization of the extent of such effects across cumulative seasonal $O_3$ exposure levels of appreciable magnitude, there is uncertainty associated with the resulting RBL predictions that might be described as an imprecision or inexactitude. The draft PA notes that the EPA's analyses of the King et al. (2005) study did 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 multi-year average. In the draft PA (EPA, 2022, page 4A-26), the authors note

> Additionally, this analysis is based on aspen, and the specific pattern of differences between the two scenarios might be expected to vary for species with different biomass growth rates (and E-R functions). However, while many multi-year tree growth studies may exist, datasets of tree growth that investigate the impact of $O_3$ across multiple-year periods (providing annual growth measurements and also detailed records of hourly concentrations that support derivation of W126 index metrics) such as that available for aspen in the study by King et al. (2005) are not prevalent.

Thus, using the Agency's analyses of the King et al. (2005) data, the draft PA (EPA, 2022) notes that the evidence did 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 multi-year average. However, the draft PA (EPA, 2022, page 4A-26) notes that datasets of tree growth that investigate the impact of $O_3$ across multiple-year periods, such as that available for aspen in the King et al. (2005) study, are not prevalent. It is suggested that caution be used in drawing a broad interpretation of EPA's use of the King et al. (2005) data. It may be premature to use the conclusions derived from the EPA's analyses of data from the King et al. (2005) study to support the Agency's decision to apply a 3-year average of the W126 index.

As noted above, in the Administrator's 2020 decision (Federal Register, 2020), 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.

At the 7 ppm-hrs annual W126 level, Fig. 4-6 (Fig. 4-9 in the draft PA (EPA (2022, page 4-63)), which was previously presented, the right side of the figure shows that the range of values for the current form of the $O_3$ NAAQS varies from approximately 55 ppb to 80 ppb. For those sites meeting the $O_3$ 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 $O_3$ NAAQS each behave differently at levels of 7 ppm-hrs and below. Focusing on the N100 values, the draft PA (EPA, 2022, page 4-69, 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 $4^{th}$ Max ≤ 70 ppb. 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 $4^{th}$ Max to control for the N100 values. Thus, a strategy of using the $O_3$ 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 $O_3$ 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 $O_3$ 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 (EPA, 2022, 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 $O_3$ NAAQS could control for the occurrences of elevated hourly average $O_3$ 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."

However, 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 $O_3$ NAAQS from elevated $O_3$ hourly average

concentrations. 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 $O_3$ concentrations ≥ 100 ppb (N100) ranging from 0 to 7 at those W126 exposures. In its preliminary conclusion, the draft PA (EPA, 2022, page 4-120) notes

> In summary, the evidence characterized in the 2020 ISA is consistent with that available in the 2015 review for the principal effects for which the evidence is strongest (e.g., plant growth, reproduction, and related larger-scale effects, as well as visible foliar injury) and for key aspects of the current standard. The evidence regarding RBL and air quality in areas meeting the current standard does not appear to call into question the adequacy of public welfare protection afforded by the standard. With regard to visible foliar injury, the currently available evidence for forested locations across the U.S., such as studies of USFS biosites, does not indicate an incidence of significant visible foliar injury that might reasonably be concluded to be adverse to the public welfare under air quality conditions meeting the current standard. For the insect-related effects that the ISA newly concludes likely to be causally related to $O_3$, the new information does not support an understanding of the potential for the occurrence of such effects in areas that meet the current standard to an extent that they might reasonably be judged significant to public welfare. Thus, we do not find the current information for these newly identified categories to call into question the adequacy of the current standard. Similarly, key uncertainties recognized in the 2015 review remain in the evidence for $O_3$ contribution to radiative forcing or effects on temperature, precipitation and related climate variables, including specifically uncertainties that limit quantitative evaluations that might inform consideration of these effects (as discussed above). Based on all of the above considerations, we conclude that the currently available evidence and quantitative exposure/risk information does not call into question the protection afforded by the current secondary standard, such that it is appropriate to consider retaining the current standard without revision. In light of this conclusion, we have not identified any potential alternative standards for consideration.

In summary, the draft PA (EPA, 2022, page 4-69, Table 4-1) notes for the 2018 to 2020 period that at an annual 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 $4^{th}$ Max ≤ 70 ppb. 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 $4^{th}$ Max to control for controlling the N100 values. Thus, at an annual W126 of 7 ppm-hrs and below, the evidence for applying a 3-year average period, either for the W126 index or the 3-year average of the $4^{th}$ Max, is not strong.

In Section 4.3, it was noted that the draft PA (EPA, 2022) does not include the results of Lee et al. (2022) because the authors' paper was not published at the time the draft PA was prepared. Using the data from the original 11 tree species summarized in the draft PA (EPA, 2022), as well as data from 5 additional tree species, Lee et al. (2022) discuss the importance of the sensitive tree species, which exhibited 5% biomass loss experiencing three month 12-h W126

exposures in the range of 2.5-9.2 ppm-hrs with N100 values ranging from 0 to 7 occurrences. The Lee et al. (2022) study is a potentially important contribution that supplements the current information contained within the draft PA on the appropriate form and level of the secondary $O_3$ NAAQS. It is recommended that EPA include the authors' findings when the Agency revises the current draft PA (EPA, 2022).

## 5. Acute and Chronic Ozone Effects

The Administrator noted (Federal Register, 2020 - page 87266) that in the Agency's 2015 review of the $O_3$ 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 $O_3$ and that it could effectively limit health effects attributable to both short- and long-term $O_3$ exposures (80 FR 65348, October 26, 2015).

In its review of the 2015 $O_3$ 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 $O_3$. This point was also highlighted by some commenters who advocated for a level of 60 ppb, based on the discussion of $O_3$-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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$, 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 $O_3$ 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 $O_3$ standard.

### 6. Margin of Safety Considerations

As noted in the Administrator's Review of the Ozone National Ambient Air Quality Standards document (Federal Register, 2020 – 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, 2020 – 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, 2020 - 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 4$^{th}$ highest daily maximum 8-hour average $O_3$ concentrations, was sufficient to protect public health, including the health of at-risk populations, with an adequate margin of safety (Federal Register, 2020 - 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 uncertainties is background $O_3$. According to a Court decision in 2019, the current rulemaking background $O_3$ should not directly influence the setting of the level of the NAAQS. The Administrator notes (Federal Register, 2020 - page 87261) that in the decision on subsequent litigation on the 2015 $O_3$ 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 $O_3$ concentrations, and socioeconomic and energy impacts. Regarding background $O_3$, the Court rejected the argument that the EPA was required to take background $O_3$ 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 $O_3$ levels in any part of the country exceeded the level of $O_3$ 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 $O_3$.

While background $O_3$ currently, as per the Court's August 19, 2019 decision, is not a direct consideration in the setting of the level of the $O_3$ standard, background $O_3$ 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 $O_3$, in an indirect manner, could influence the level of the $O_3$ standard. Background $O_3$ concentrations in the low- and mid-level part of the distribution of concentrations make up a large fraction of the total ambient $O_3$ 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 $O_3$ concentrations are estimated to range from 20−50 ppb (EPA, 2020a - page 1-1). This means that over an entire $O_3$ season, hourly average background $O_3$ 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 $O_3$ along the full range of $O_3$ concentrations (page 9-30 of the 2014 Health REA (EPA, 2014b)), including the low-level values associated with background $O_3$. For lung function probabilistic population-based Exposure-Response (E-R) functions risk assessments, the lower concentrations, which are at background $O_3$ 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 $O_3$ 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 $O_3$ 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) in the draft PA (EPA, 2022). However, quantification of background $O_3$ for the 8 cities was not performed. Therefore, no information was provided in the current $O_3$ NAAQS reconsideration rulemaking process concerning the contribution of background $O_3$ to the human health effects risk assessments that provided valuable information on the margin of safety.

## 7. Future Research

### 7.1 Health

Areas of future health research are discussed in the draft PA (EPA, 2022, page 3-102). 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 $O_3$ 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 $O_3$-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 $O_3$ in the range of 40 to 80 ppb, and lower, for 6.6 hours while engaged in moderate exertion.**

As noted in the draft PA (EPA, 2022), the lower hourly average $O_3$ 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 $O_3$ becoming more and more important in the lower MDA8 concentration range. Thus, suggesting that researchers utilize $O_3$ concentrations in the background concentration part of the distribution 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 $O_3$ concentrations elicit a greater effect than the middle and lower values. Unless the EPA can justify the use of constant concentration exposure regimes that mimic actual ambient data patterns, it is recommended that one should use $O_3$ exposure regimes that are experienced under actual conditions.

2. **Epidemiologic studies assessing the influence of "long-term" or "short-term" $O_3$ 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 $O_3$ health effects requires a short-term standard, such as the 4$^{th}$ highest 8-h daily maximum exposure metric and that a long-term average concentration standard is required to reduce chronic $O_3$ 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" $O_3$ 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 $O_3$-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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ 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 $O_3$ necessitate a standard level below 70 ppb.

Thus, given EPA's conclusions reached in the 2015 $O_3$ 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 draft PA (EPA, 2022) that for sites experiencing emission reductions, exposure metrics that use averaging over longer time periods of hourly $O_3$ measurements, such as the 6-month (April-September) average of daytime (8am-7pm) $O_3$ 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 $O_3$ concentrations showing increasing trends, while the 4$^{th}$ 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 $O_3$ exposures. Both annual average and other long-term average metrics are influenced by the titration of $O_3$ by NO as $NO_x$ emissions are reduced to protect the public's health and welfare.

Rather, 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 was described by Lefohn, Hazucha, Shadwick, and Adams (2010). The authors described a sigmoidal weighting scheme for hourly average $O_3$ concentrations. The weighting scheme addresses the nonlinearity of response (i.e., the greater effect of higher $O_3$ concentrations over the mid- and low-range values) on an hourly basis. The weighting scheme focused on the use of daily $O_3$ exposures that were integrated over time. The authors described a W90 exposure index for use in assessing $FEV_1$ decrements. The scheme is shown in Fig. 7-1 below. The form of the W90 index is $\Sigma\, w_i \times C_i$ 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 $O_3$ concentration in units of ppb. The W90 index has units of ppb-hrs.

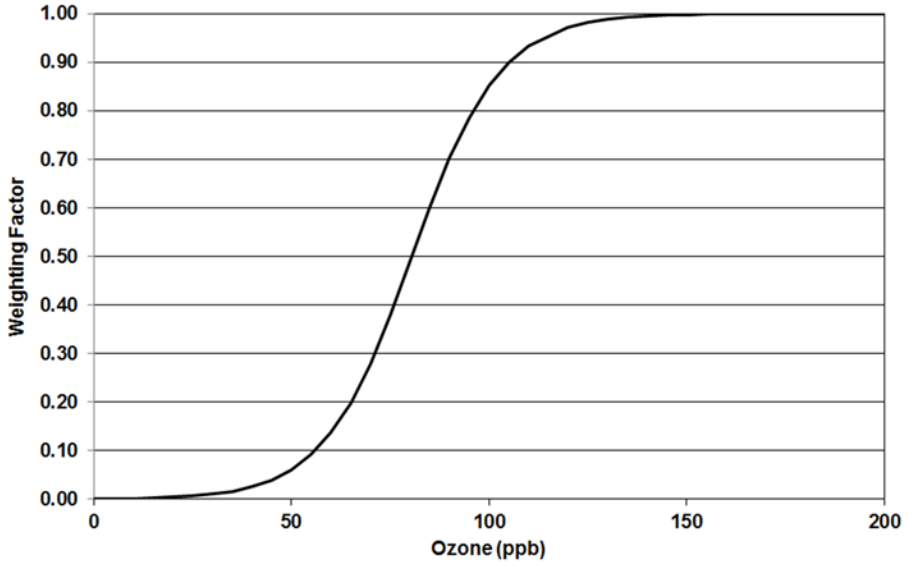



**Figure 7-1. The weighting applied to hourly average ozone values for the calculation of the W90 exposure index (see Lefohn et al., 2010).**

## 7.2 Vegetation

Areas of future health research are discussed in the draft PA (EPA, 2022, page 4-121).

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 $O_3$ concentration data over the exposure would reduce uncertainty in estimates of effects across multiple-year periods and at the $O_3$ 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 Table 4A-6 in the draft PA (EPA, 2022, pages 4A-17 – 4A-21), additional experiments are needed using hour-by-hour exposures that mimic ambient exposures. 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 on the lower range of $O_3$ exposures that include a low number or 0 occurrences of hourly average $O_3$ concentrations ≥ 100 ppb. In other words, a better definition of the lower part of the effects curve is needed.

## 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/.

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 lnJury 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.

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 Adminstrator, Science Advisory Board Washington, DC. Available at: https://yosemite.epa.gov/sab/sabproduct.nsf/264cb1227d55e02c85257402007446a4/4713D217BC0710348525851500 6359BA/$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., Roccato, F., Venzac, H., Vuillermoz, E., Bonasoni, P. 2010. Tropospheric ozone variations at the Nepal Climate Observatory-Pyramid (Himalayas,

5079ma.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., Arbillaa, 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 NOx 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.

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.

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, p. 3051.

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. 251 / Friday, December 31, 2020. 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 $NO_2$ 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.

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.

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 O-3 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 $O_3$ 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, nwaa137, 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 $O_3$ 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 $O_3$ compromises net primary production in young stands of trembling aspen, paper birch and sugar maple in response to elevated atmospheric $CO_2$. 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 03 -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 03 crop reduction mathematical model. J. Air Pollut. Control Assoc. 34: 1023-1034.

Laughner, J., Cohen, R. 2019. Direct observation of changing NOx 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. 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., 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., Shadwick, D.S., Somerville, M.C., Chappelka, A.H., Lockaby, B.G., Meldahl, R.S. 1992. The Characterization and Comparison of Ozone Exposure Indices Used in Assessing the Response of Loblolly Pine to Ozone. Atmospheric Environment. 26A(2):287-298.

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.

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. 2010. An alternative form and level of the human health ozone standard. Inhalation Toxicology 22:999-1011.

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.

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 $S0_2$ 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.

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 $S0_2$ 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., Holopaninen, 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, DW., Thompson, AM., 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?Dockey=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.

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 LJ, Barnard JW, Shephard RJ. 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 NOx and VOC emissions. *Environ Sci Technol* **49**: 186-195. dx.doi.org/10.1021/es504514z.

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., Pfhal, 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 NOy 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.

US 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.

US. 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.

US. 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

US 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.

US 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.

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.

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.

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° × 2/3° horizontal resolution over North America. Atmospheric Environment 45, 6769-6776.