Wildfire Exceptional Events Demonstration for Ground-Level Ozone in Western Michigan 2015 Ozone Nonattainment Areas of Allegan, Berrien, and Muskegon Counties June 17-20, 2020 Episode

Draft Demonstration Document

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Prepared by:

Alpine Geophysics, LLC
387 Pollard Mine Road
Burnsville, NC 28714

For

Michigan Department of Environment, Great Lakes, and Energy

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Introduction

The following is Michigan's Exceptional Event Demonstration, which clearly establishes that plumes from Arizona wildfires adversely affected ozone data in a regulatorily significant way at ozone monitors in Allegan, Berrien, and Muskegon Counties in Michigan during a June 17-20, 2020 episode. Wildfires occurred across Arizona throughout June 2020 with meteorological conditions (at the surface and aloft) favorable for transport of smoke from the wildfires into the region, including Michigan, during June 17-20, 2020. Maximum daily 8-hour average ozone (MDA8) concentrations across the region on June 17, 18, 19, and 20, exceeded the 2015 ozone National Ambient Air Quality Standards (NAAQS) after the precursor-rich smoke plume subsided to the surface (Figure 1). Western Michigan MDA8 concentrations reached 83 ppb at the peak of the event at the Muskegon monitor and 79 ppb at both the Holland (Allegan) and Coloma (Berrien) monitors and were among the highest MDA8s during the 2020 ozone season for those sites.

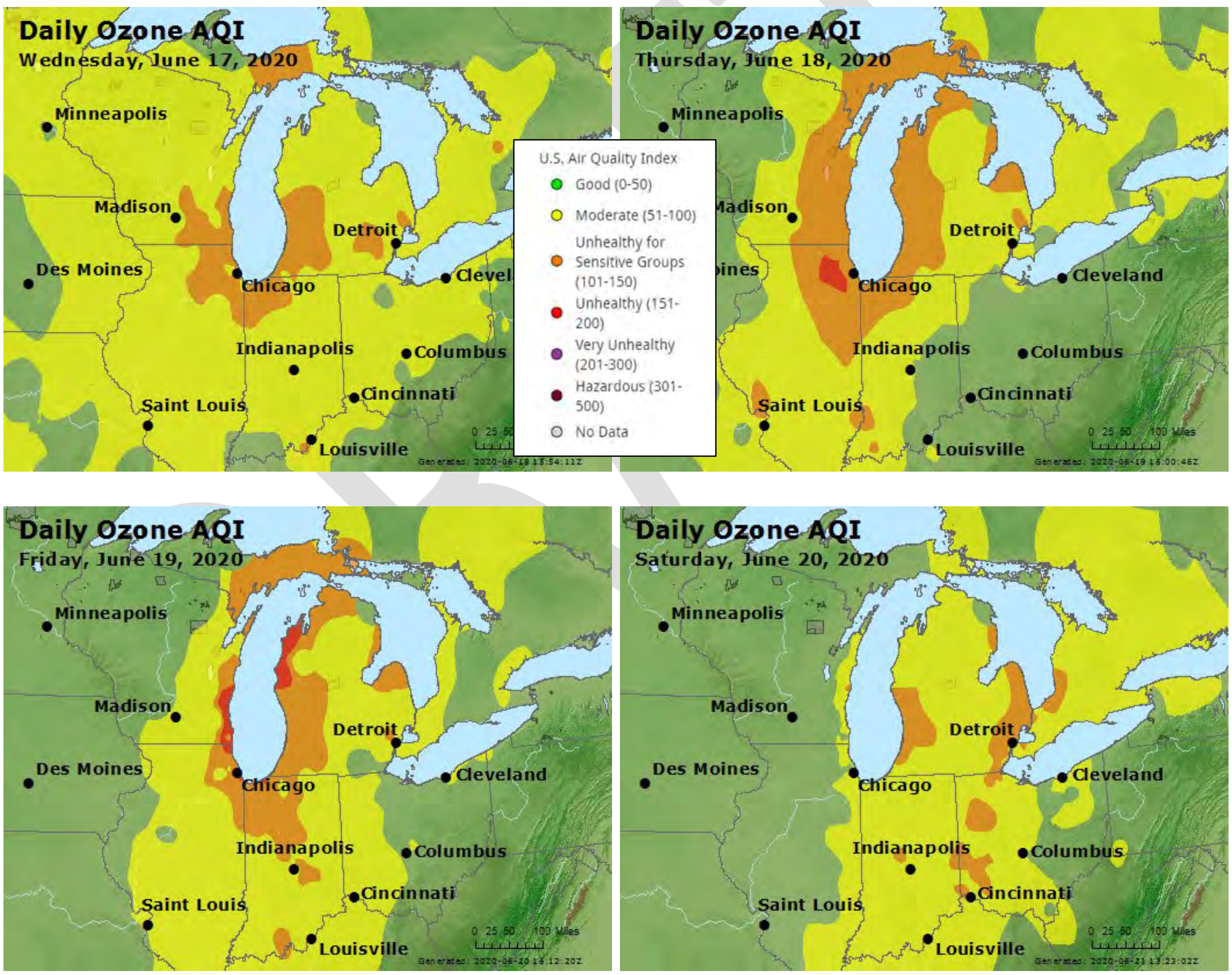

Figure 1. Ozone Air Quality Index (AQI) maps from June 17 (top left), 18 (top right), 19 (bottom left), and 20 (bottom right), 2020.

Table 1 presents the MDA8 ozone concentrations for the episode days of June 17-20, 2020 with an indicator (in parentheses) of the rank of that day's value compared to 2020 at each monitor.

Table 1. MDA8 ozone concentrations and ranks on June 17-20, 2020 for western Michigan monitors.

		MDA8 [ppb] (rank)				Preliminary 2020 (ppb)	
Site Name	**Monitor ID**	**6/17**	**6/18**	**6/19**	**6/20**	**4th High**	**DV**
Holland	26-005-0003	70 (7)	76 (4)	79 (2)	72 (5)	76	73
Coloma	26-021-0014	73 (5)	79 (3)	78 (4)	72 (6)	78	72
Muskegon	26-121-0039	69 (9)	76 (5)	80 (4)	83 (1)	80	76

Table 2 identifies the Michigan monitors that were affected by smoke transported from Arizona wildfires on June 17-20, 2020 such that the data should be excluded from regulatory determinations.

Table 2. Ozone Data Requested for Exclusion

Nonattainment Area	Monitor ID	Site Name	Dates
Allegan	26-005-0003	Holland	June 17-20, 2020
Berrien	26-021-0014	Coloma	June 17-20, 2020
Muskegon	26-121-0039	Muskegon	June 18-20, 2020

40 CFR 50.14 establishes the procedures for submitting an Exceptional Event Demonstration. Specifically, 40 CFR 50.14(c)(3)(iv) states: "The demonstration to justify data exclusion must include:

A. A narrative conceptual model that describes the event(s) causing the exceedance or violation and a discussion of how emissions from the event(s) led to the exceedance or violation at the affected monitor(s);
B. A demonstration that the event affected air quality in such a way that there exists a clear causal relationship between the specific event and the monitored exceedance or violation;
C. Analyses comparing the claimed event-influenced concentration(s) to concentrations at the same monitoring site at other times to support the requirement at paragraph (c)(3)(iv)(B) of this section. The Administrator shall not require a State to prove a specific percentile point in the distribution of data;
D. A demonstration that the event was both not reasonably controllable and not reasonably preventable; and
E. A demonstration that the event was a human activity that is unlikely to recur at a particular location or was a natural event."

The following demonstration was prepared in accordance with 40 CFR 50.14, U.S. Environmental Protection Agency's (EPA) September 16, 2016, "Guidance on the Preparation of Exceptional Events Demonstrations for Wildfire Events that May Influence Ozone Concentrations"[1] (herein referred to as Exceptional Events Guidance), and U.S. EPA's "EPA Review Technical Support Document Template for Wildfire/Ozone Events."[2]

[1] https://www.epa.gov/sites/production/files/2016-09/documents/exceptional_events_guidance_9-16-16_final.pdf
[2] https://www.epa.gov/sites/production/files/2017-06/documents/tsd_template_ozone_wildfire_ee_2017_0606.pdf

Summary of Findings

This report:

1. Contains the required narrative conceptual model describing the Arizona wildfire event that caused the violation at the Holland, Coloma, and Muskegon ozone monitors, and how emissions from those events reached the affected monitor, leading to elevated measured ozone concentrations on the specific days in question.
2. Demonstrates that there was a clear causal relationship between smoke and the maximum daily average 8-hour (MDA8) ozone exceedances.
3. Contains analyses comparing the ozone concentrations during the event-influenced days to concentrations at the same monitor at other times on days with similar meteorological conditions.
4. Demonstrates that the wildfires causing smoke were not reasonably controllable or preventable and are unlikely to recur, and that they were considered natural events.

Key findings and evidence supporting these assertions include the following:

1. Considerable ozone was created upstream of Michigan due to the presence of wildfire smoke generated during one of Arizona's largest recorded wildfire years, which was then transported into Michigan over several days in June 2020.
2. Meteorological conditions (at the surface and aloft) were favorable for transport of smoke from the wildfire in Arizona into the region, including Michigan, during June 2020.
3. Ozone concentrations during the June 17-20 episode at the Muskegon, Holland, and Coloma monitors were measured above the 99th percentile of the 5-year distribution of ozone monitoring data at the sites.
4. Satellite images captured visual smoke plumes that were transported into the Lake Michigan region on days when the ozone concentrations were highest.
5. Analysis of the National Oceanic and Atmospheric Administration's (NOAA) Hazard Mapping System (HMS) smoke product and Ozone Air Quality Index (AQI) shows an enhanced ozone concentration impact at monitors along the wildfire smoke transport path that eventually culminates with excess ozone observations in western Michigan.
6. Cloud-Aerosol Lidar and Infrared Pathfinder Satellite Observation (CALIPSO) retrievals identified smoke among the classified aerosols at the surface in the region during the June 17-20, 2020, episode.
7. Regional upwind measurements identify multiple monitors with unusually high ozone concentrations during the dates when the transported smoke plume passes through the region prior to the June 17-20, 2020 episode event.
8. Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model forward and backward trajectory analyses demonstrate that wildfire smoke was transported into the region and was then transported into the western Michigan area during the June 2020 event.
9. Additional satellite retrievals demonstrate the transport of wildfire smoke into the region and provide additional evidence that the smoke plume and associated ozone precursor emissions were present during the June 17-20 episode.
10. Fine particulate matter (PM2.5), was elevated during the event, consistent with a wildfire smoke plume.
11. PM2.5 speciated data (organic carbon and potassium ion) showed elevated wildfire attributable concentrations during the June 17-20, 2020 event.

12. Comparable meteorological and typical non-event ozone exceedance day analyses suggests that the June 17-20, 2020 exceedance events were influenced by factors not explained by meteorology alone and indicated a wide, regional impact beyond just the western Michigan NAAs, lending support to the conclusion that the influence of wildfire smoke created the ozone exceedances on June 17-20, 2020.
13. A multi-day buildup of both wildfire smoke and ozone precursor concentrations in the Michigan area enhanced ozone concentrations in the days building up to the June 17-20, 2020 episode days.
14. A screening analysis of average standardized log-transformed timeseries concentrations of key pollutants provides supporting evidence for smoke influence in the western Michigan region during the June 17-20, 2020 episode.
15. Q/d analyses, while not meeting specific U.S. EPA thresholds for clear causal influence, are consistent with other previous long-range smoke and ozone transport events approved by U.S. EPA.

Several analytical methods were used to develop a weight of evidence demonstration that the 8-hour ozone concentrations above 70 parts per billion (ppb) in the June 17-20, 2020 event meet the rules for data exclusion as an Exceptional Event. In summary, satellite images and data, meteorological data, trajectory analysis, screening tools, and speciated PM2.5 data were used to assess whether conditions were favorable for transport of smoke from the Arizona wildfires to monitors that showed 8-hour ozone concentrations above 70 ppb. The data also showed that during the June 2020 episode, the transported smoke degraded air quality upstream of Michigan first, then this photochemically aging airmass was transported northeastward, creating a prolonged period (June 17-20, 2020) of enhanced ozone from the Mississippi River into the region, including Michigan.

Michigan’s analysis strongly supports that the Muskegon, Holland, and Coloma monitors were impacted by smoke, that concentrations on June 17-20, 2020, meet the rules as Exceptional Events, and that the Muskegon, Holland, and Coloma monitors and associated ozone observations on these days should be excluded from design value calculations.

Exceptional Event Demonstration

A. Regulatory Significance

The Exceptional Events rule applies to data showing an exceedance of a standard which may affect regulatory determinations regarding attainment designation status or other actions by the Administrator.

Exclusion of the June 17-20, 2020 data, alone or in combination with a separate August 26, 2020 exceptional event, may allow the Allegan and Berrien 2015 ozone nonattainment areas (NAAs) to be eligible for redesignation to attainment for the 2015 ozone National Ambient Air Quality Standard (NAAQS). Table 3 compares preliminary 2018-2020 design values calculated with and without the inclusion of data from the events. The 2020 design value data are preliminary and based on data reported through December 8, 2020, which have not yet been certified.

Exclusion of the June 17-20 and August 26, 2020, ozone data would reduce the preliminary 2018-2020 design value for the Holland monitor (26-005-0003) from 73 ppb (nonattainment) to 70 ppb (attainment) thereby bringing the Allegan NAA into attainment with the 2015 ozone NAAQS.

Exclusion of the June 17-20, 2020, ozone data would reduce the preliminary 2018-2020 design value for the Coloma monitor (26-021-0014) from 72 ppb (nonattainment) to 69 ppb (attainment) thereby bringing the Berrien NAA into attainment with the 2015 ozone NAAQS.

Exclusion of the June 18-20 and August 26, 2020, data may allow the Muskegon 2015 ozone nonattainment area to be eligible for one-year extension of the attainment date for the 2015 ozone NAAQS. Table 3 presents the preliminary 2020 maximum daily 8-hour average (MDA8) observations calculated with and without the inclusion of data from the event. The 2020 data are preliminary and based on data reported through December 8, 2020, which have not yet been certified.

Exclusion of the June 18-20 and August 26, 2020 ozone data would reduce the preliminary 2020 4th high concentration for the Muskegon monitor (26-121-0039) from 80 ppb to 70 ppb thereby bringing the NAA below the threshold necessary to meet the criteria, as provided in CAA section 181(a)(5) and 40 Code of Federal Regulations (CFR) 51.1107, to qualify for a 1-year attainment date extension for the 2015 ozone NAAQS even though the monitor did not attain the NAAQS by the applicable deadline. Depending on 2021 or 2022 data, exclusion of June 17-20 and August 26, 2020 data may have regulatory significance for other actions by the Administrator, including future clean data determinations, redesignations, 2015 ozone NAAQS, or violations of the 2015 ozone NAAQS.

As a result, Michigan has decided to focus this demonstration on only the dates necessary to demonstrate attainment or reach 4th high MDA8 levels for eligibility for a one-year extension of the 2015 ozone NAAQS attainment date and notes that if future assessments of attainment status based on inclusion of sites and dates provide lower controlling critical differences, then Michigan will revisit this analysis.

Table 3. Ozone monitors at which Michigan is seeking EPA concurrence to exclude data.

		Preliminary 2020 Ozone									
		MDA8 [ppb] (rank)				4th High [ppb]			DV [ppb]		
Site Name	Monitor ID	6/17	6/18	6/19	6/20	Including	Excluding	Excluding June & Aug	Including	Excluding	Excluding June & Aug
Holland	26-005-0003	70 (7)	76 (4)	79 (2)	72 (5)	76	68	67	73	71	70
Coloma	26-021-0014	73 (5)	79 (3)	78 (4)	72 (6)	78	70	70*	72	69	69*
Muskegon	26-121-0039	-	76 (5)	80 (4)	83 (1)	80	70	70	76	72	72

*No change in these values as the Coloma monitor is not part of the August 26, 2020 exceptional event demonstration.

B. Narrative Conceptual Model

Area Descriptions

As shown in Figure 2, Michigan has four 2015 ozone NAAs, Muskegon County, Allegan County, Berrien County, and Detroit. This document has been prepared to address exceptional events that impacted the three NAAs on the western side of the state, Muskegon, Allegan, and Berrien NAAs.

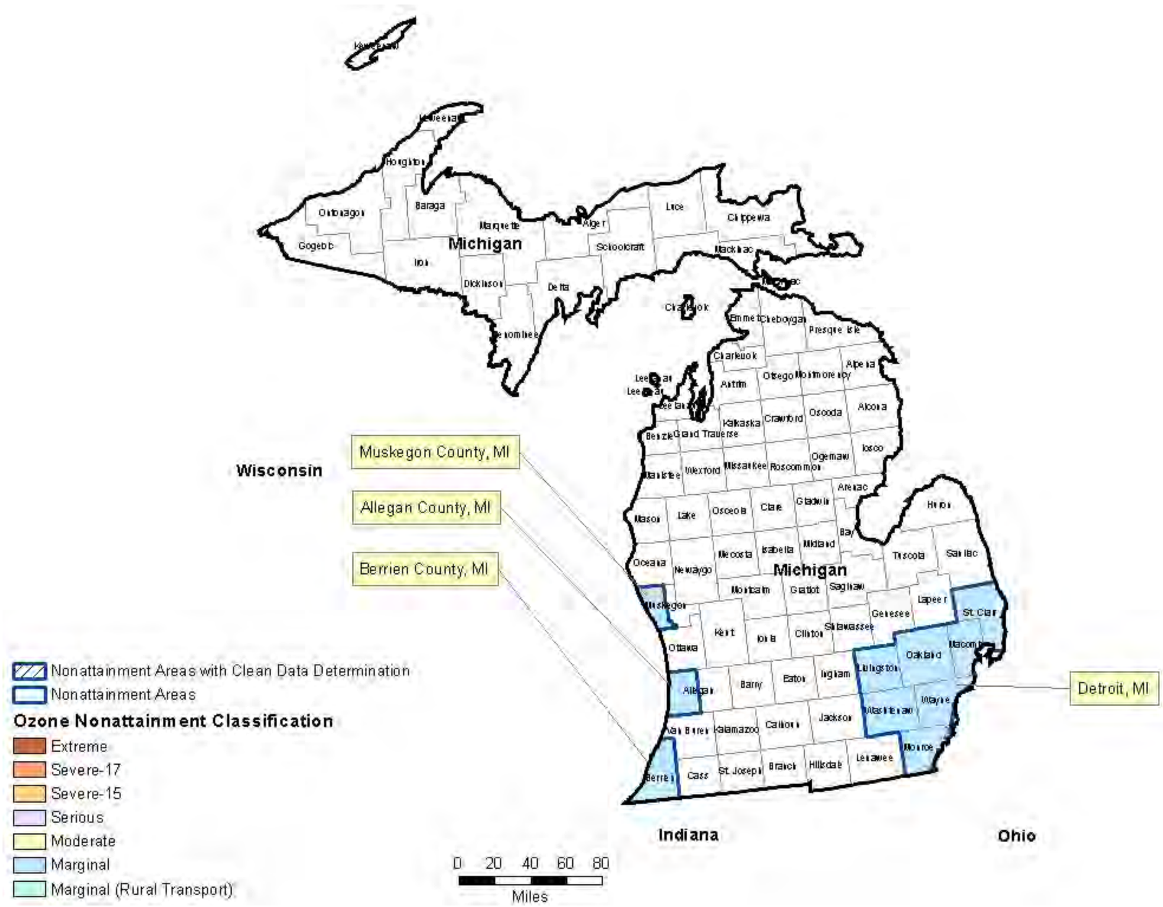

Figure 2. Michigan 8-hour Ozone Nonattainment Areas (2015 NAAQS).

On June 4, 2018, U.S. EPA designated Berrien County and portions of Allegan and Muskegon Counties in western Michigan as "marginal" ozone NAAs based on monitoring data from 2014-2016[3]. The attainment deadline for marginal NAAs to meet the 2015 ozone NAAQS is August 3, 2021.

These and the Detroit NAA are classified as marginal nonattainment, which is the lowest level of classification and means that ozone concentrations are less than 10 parts per billion (ppb) above the standard. A redesignation request was submitted for Berrien County in January of 2020 and is available on the Recent Air Quality Planning Actions and Documents webpage[4]. This request has not yet been acted on by U.S. EPA.

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[3] 83 FR 25776

[4] https://www.michigan.gov/documents/egle/aqd-aqe-sip-Berrien_County_Redesignation_Request_680643_7.pdf

As a result of the action for the NAAs described above, the state of Michigan must submit a State Implementation Plan (SIP) that meet the requirements applicable to marginal ozone NAAs.

Ozone has significantly decreased in the three western Michigan NAAs due to sizeable and sustained reductions in ozone precursor emissions. This is evident in Figure 3, Figure 4, and Figure 5 below, showing the number of days in each year since 2000 exceeding the 70 ppb NAAQS for ozone for Allegan County, Berrien County, and Muskegon County, respectively.

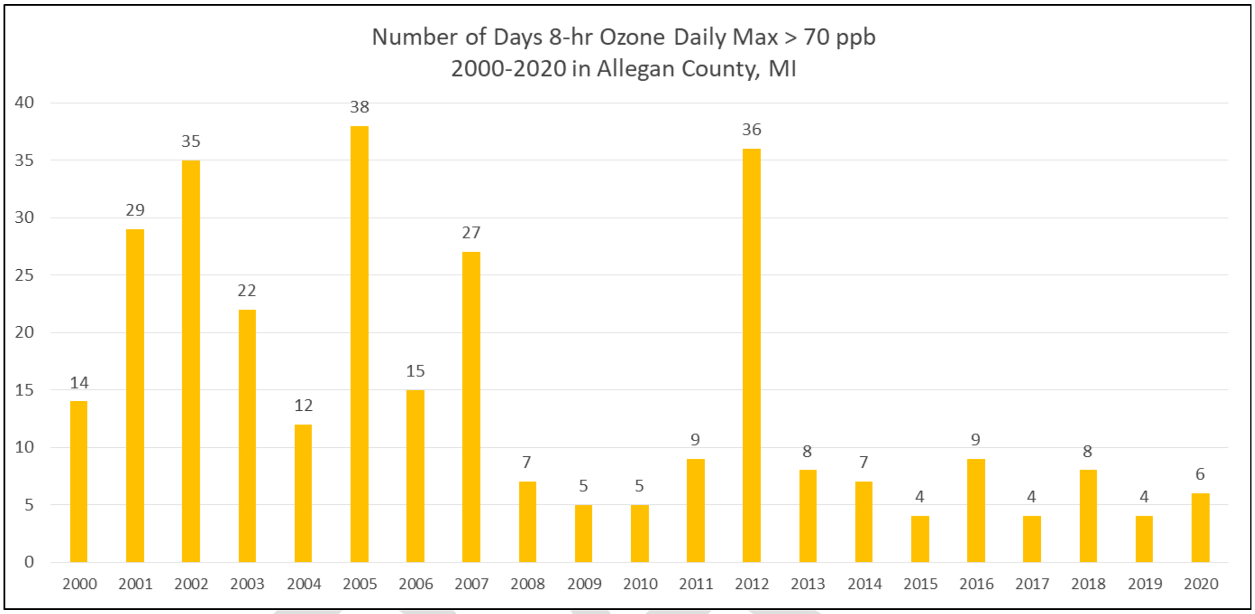

Figure 3. Number of Days Exceeding 2015 Ozone NAAQS Level of 70 ppb in Allegan NAA.

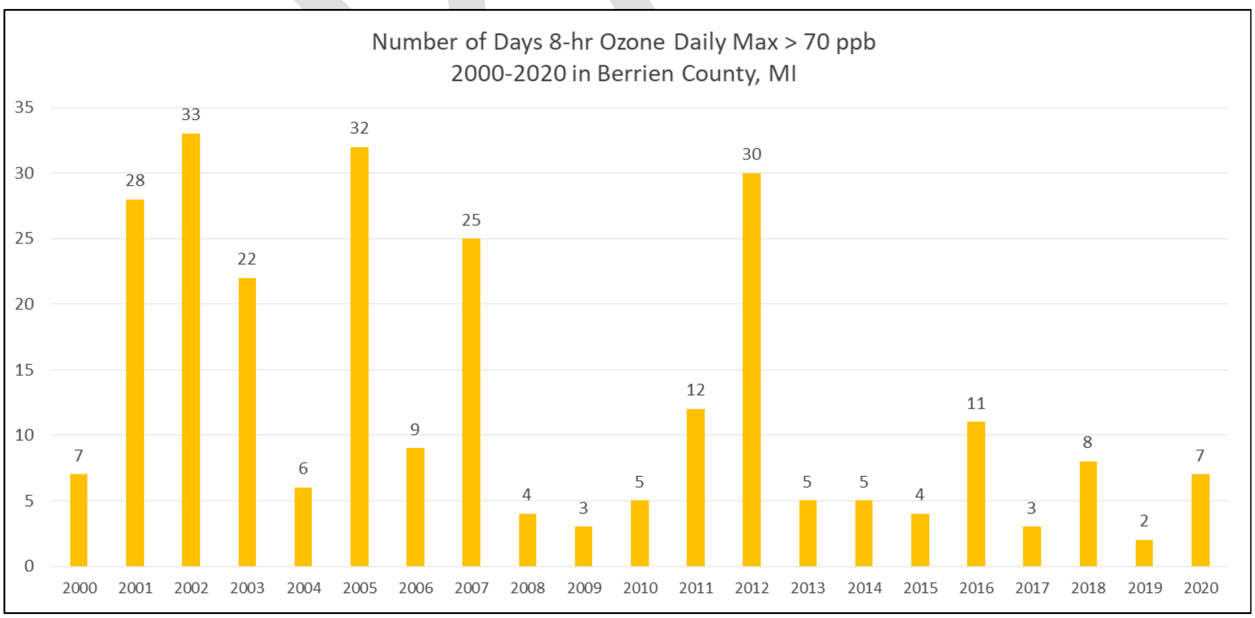

Figure 4. Number of Days Exceeding 2015 Ozone NAAQS Level of 70 ppb in Berrien NAA.

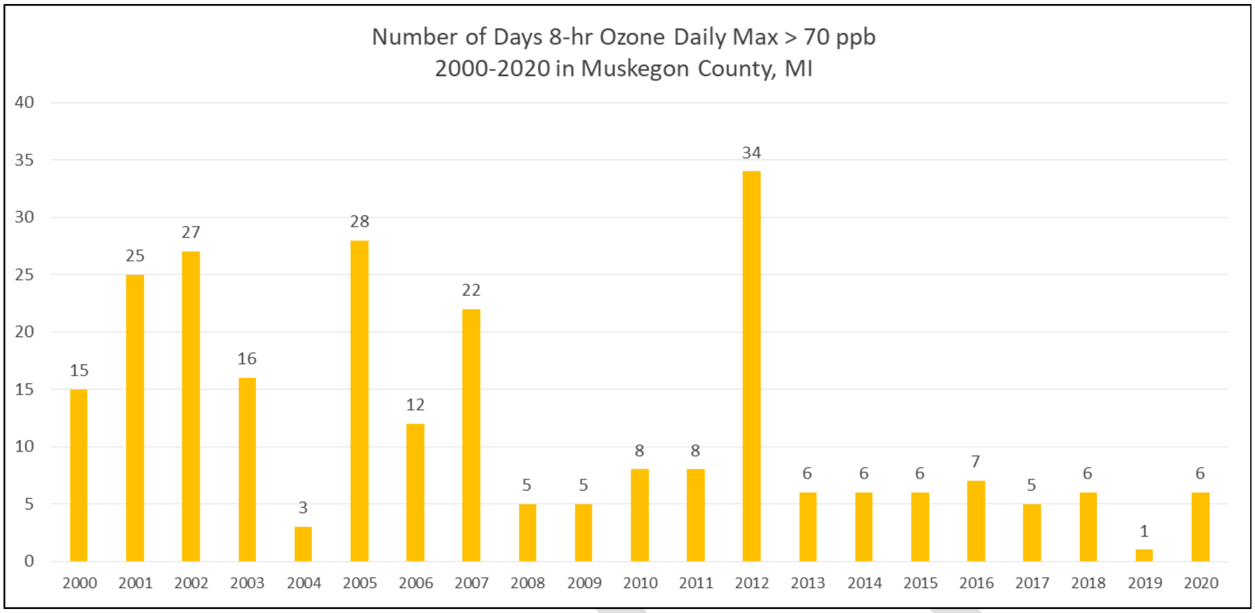

Figure 5. Number of Days Exceeding 2015 Ozone NAAQS Level of 70 ppb in Muskegon NAA.

The Holland site (Allegan County; Monitor ID 26-005-0003; Lat 42.7678, Lon -86.14861) is an urban scale site purposed to monitor maximum concentrations. It is a State and Local Air Monitoring Station (SLAMS) station located approximately 3 miles east of Holland State Park on Lake Michigan monitoring ozone and continuous PM2.5.

The Coloma site (Berrien County; Monitor ID 26-021-0014; Lat 42.1978, Lon -86.30972) is a regional scale site located in a residential area in Coloma Charter Township purposed to measure maximum concentrations. It also is a SLAMS station located at 4689 Defield Road at the Paw Paw Lake Wastewater plant, approximately 4.7 miles east of Lake Michigan and monitors ozone.

The Muskegon site (Muskegon County; Monitor ID 26-121-0039; Lat 43.2781, Lon -86.31111) is a regional scale site located in a residential area in Laketon Township. It is a SLAMS station located at 1340 Green Creek Road, at the Laketon Township Hall approximately 3 miles east of Lake Michigan and monitors ozone.

Figure 6 presents the location of each of these monitors.

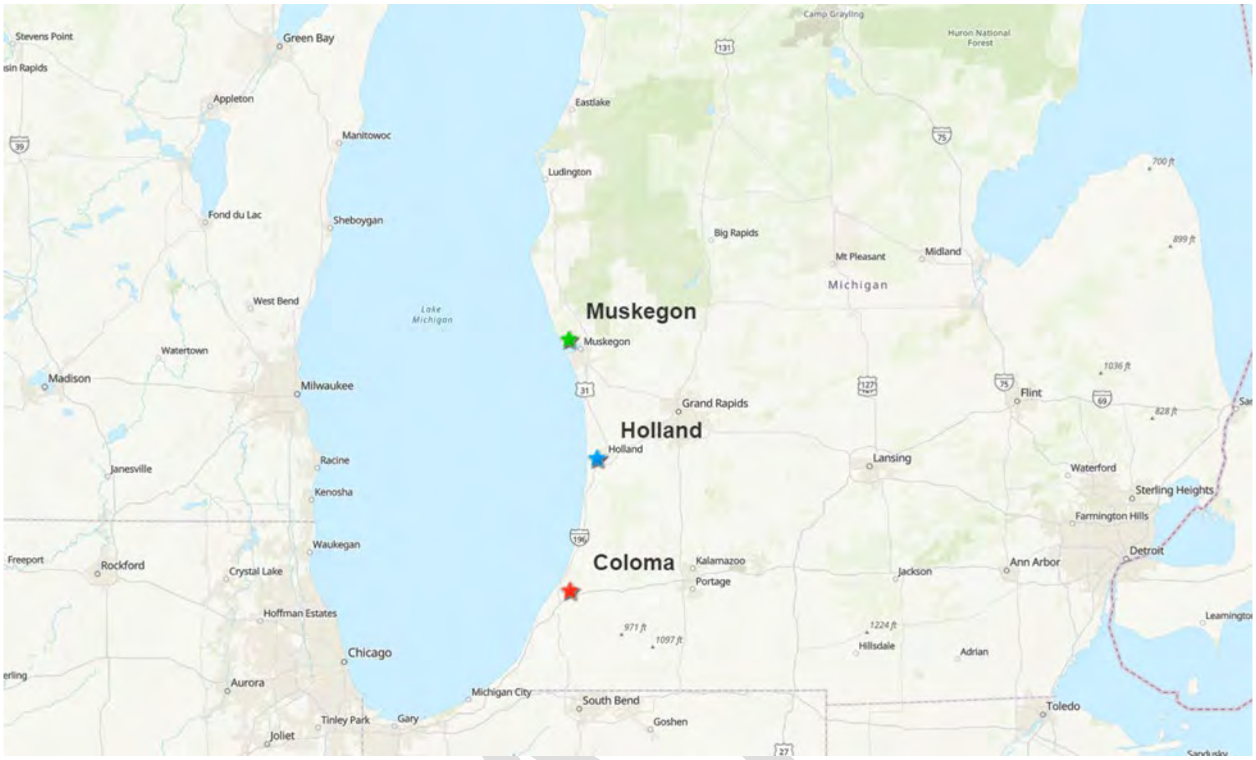

Figure 6. Western Michigan NAA monitors.

Characteristics of Non-Exceptional Event (Typical) Ozone Formation

The following conceptual model of typical ozone formation characteristics is adopted from Lake Michigan Air Directors Consortium's (LADCO) November 19, 2020, "Attainment Demonstration Modeling for the 2008 Ozone National Ambient Air Quality Standard Technical Support Document."[5] This regional description is applicable to the western Michigan area.

Based on the data and analyses presented in the LADCO report and previous conceptual models and technical support documents developed for the Lake Michigan region, a conceptual model of the behavior, meteorological influences, and causes of high ozone in the western Michigan NAAs is summarized below:

- Monitoring data show that, as of 2019, only the Coloma ozone monitoring site in Berrien County was meeting the 2015 8-hour ozone NAAQS across all three of the analyzed NAAs. As shown in Figure 3, Figure 4, and Figure 5 above, historical ozone data show year to year variation with an overall downward trend over the past 19 years, due likely to federal and state emission control programs.
- Ozone concentrations are strongly influenced by meteorological conditions, with high ozone days and higher ozone levels occurring more frequently during summers with above-normal temperatures. Nevertheless, meteorologically-adjusted trends at the controlling monitors show that concentrations have declined, even on hot days, which provides strong evidence that emission reductions of ozone precursors have been effective.
- The presence of Lake Michigan influences the formation, transport, and duration of elevated ozone concentrations along its shoreline. Depending on large-scale synoptic winds and local-scale lake breezes, different parts of the area experience high ozone concentrations. For example, under southerly flow, high surface ozone concentrations can occur in eastern Wisconsin, and under southwesterly flow, high surface ozone can occur in western Michigan.
- A natural lake-land breeze circulation pattern is a major cause of the high ozone concentrations observed along the lakeshore. This pattern is driven by surface temperature gradients between the lake and the land. At night and during the early morning hours, when the lake surface is warmer than the land surface, a land breeze forms (surface winds travel from the land to the lake). The land breeze transports ozone precursors from industrial and mobile sources on land to the area over the lake. When the sun rises, the ozone precursors over the lake begin to rapidly react to form ozone. The lake breeze transports the ozone precursors and the concentrated ozone that has formed above the lake surface and precursors from the lake, inland to a narrow band along the lake shore. The ozone concentrations observed along the lakeshore that violate the NAAQS are often associated with lake-land breeze patterns.
- Areas in closer proximity to the lake shoreline display the most frequent and most elevated ozone concentrations.

Transport of ozone and its precursors is a significant factor and occurs on several spatial scales. Regionally, over a multi-day period, somewhat stagnant summertime conditions can lead to the build-up in ozone and ozone precursor concentrations over a large spatial area. This polluted air mass can be transported long distances, resulting in elevated ozone levels in locations far downwind.

[5] https://www.ladco.org/wp-content/uploads/Documents/Reports/TSDs/O3/LADCO_2008O3_SeriousNAASIP_TSD_19Nov2020.pdf

Locally, emissions from urban areas add to the regional background leading to ozone concentration hot spots downwind. Depending on the synoptic wind patterns (and local land-lake breezes), different downwind areas are affected.

Electric generating units (EGUs) are a major source of ozone precursors. EGUs can produce large amounts of emissions over a short duration and generally emit at stack elevations conducive to transport. During hot days many of the less frequently used high-emitting EGUs come online to supply the high electric demand of air conditioning and refrigeration along with base load units operating at full capacity. To examine if the high observed ozone concentrations on June 17-20, 2020 were a result of high EGU NOx emissions, an analysis was conducted to examine the correlation of EGU emissions and ozone concentrations during the 2020 ozone season.

U.S. EPA's preliminary transport modeling for the 2015 ozone standard[6] shows that ozone concentrations at these monitors are most influenced by emissions from Illinois, Indiana, Kentucky, and Michigan, in addition to Ohio's own emissions. Figure 7 and Figure 8 show EGU NOx emissions[7] from these states during the ozone season and during June has significantly decreased from 2016 to 2020.

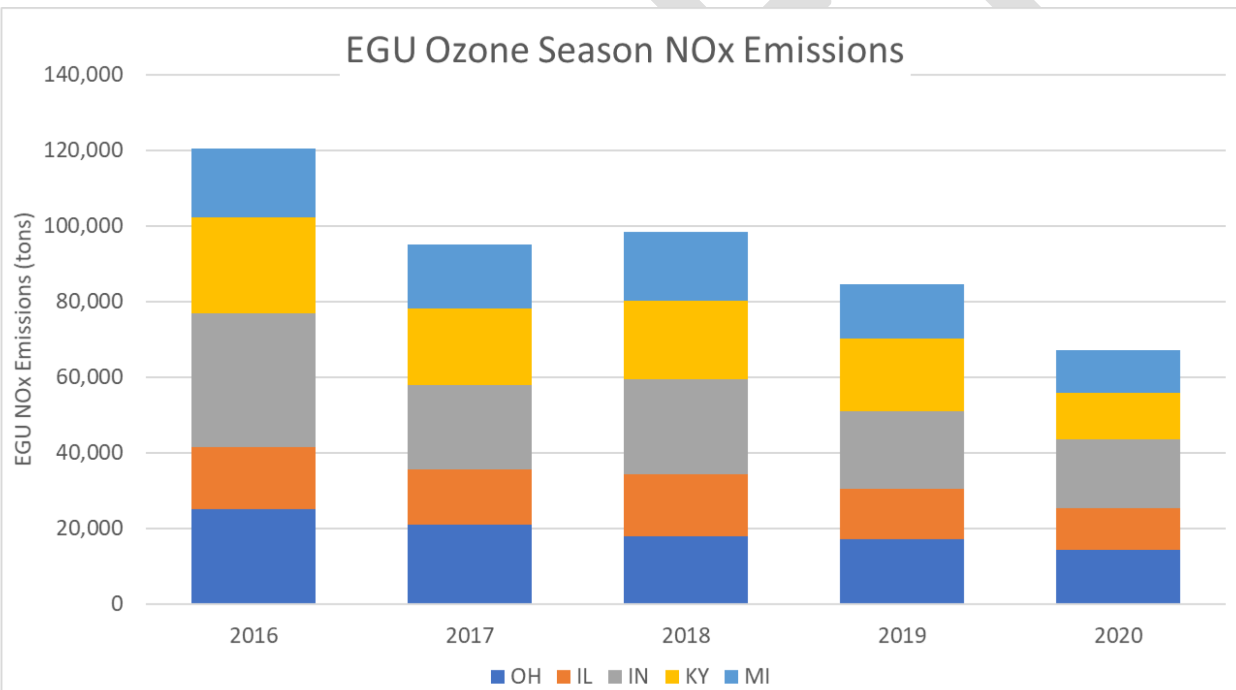

Figure 7. 2016-2020 EGU NOx Emissions (OH, IL, IN, KY, MI) – Ozone Season

[6] https://www.epa.gov/sites/production/files/2017-01/documents/aq_modeling_tsd_2015_o3_naaqs_preliminary_interstate_transport_assessmen.pdf
[7] Data obtained from U.S. EPA's Clean Air Markets Division (CAMD) at https://ampd.epa.gov/ampd/

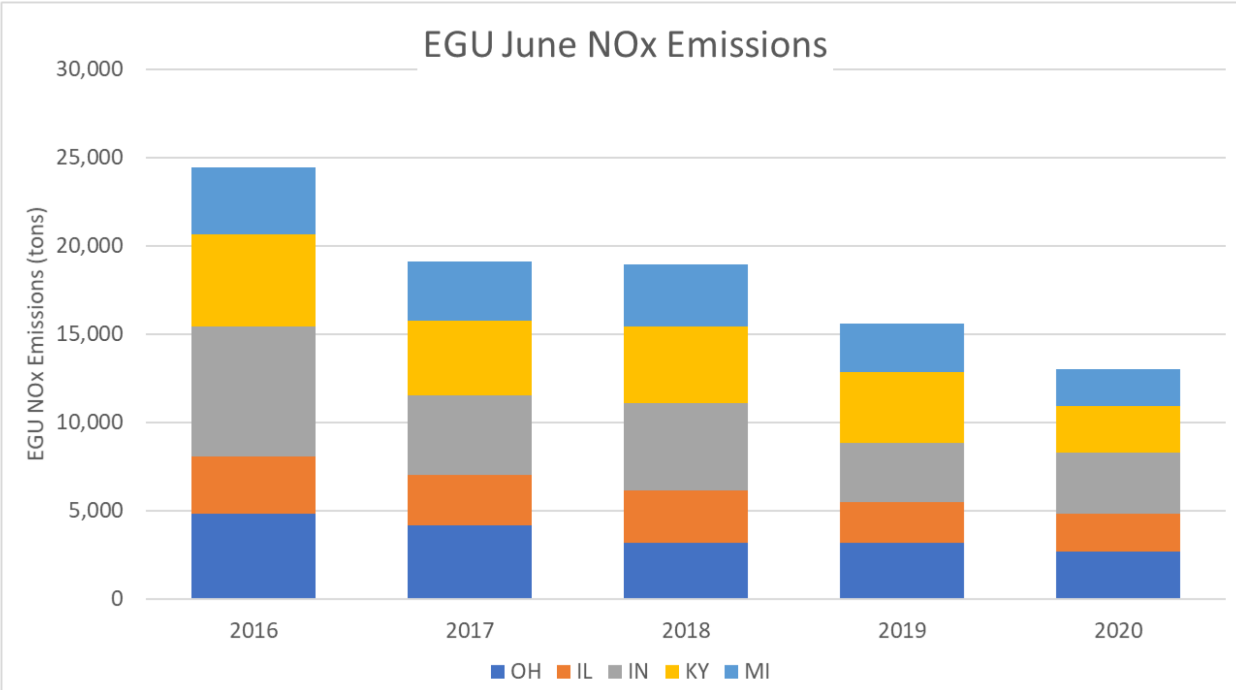

Figure 8. 2016-2020 EGU NOx Emissions (OH, IL, IN, KY, MI) – June

Since most other anthropogenic source emissions are relatively constant between workdays, emissions from sources such as EGUs must not change significantly between days to conclude that the smoke played the integral part in these exceedances. To further evaluate whether the increased ozone observed was carried within the smoke plume and was not simply from upstream EGUs, an EGU NOx to ozone ratio for the ozone season was developed. If the EGU NOx emissions were insignificant to ozone production across the region as compared to the contribution to ozone supplied by the smoke, the EGU NOx emissions to ozone concentration ratio should be quite low. If ozone production were dependent on EGU NOx output, the ratio would remain constant and/or high.

Under typical conditions, less NOx means less ozone. Thus, an airmass characterized by an abundance of ozone but also impacted by copious EGU NOx emissions will maintain a high or constant EGU NOx emissions to ozone concentration ratio. A reduction in emissions leading to reductions in ozone and again the ratio would remain relatively constant. However, an airmass producing abundant ozone without substantial increases in anthropogenic NOx emissions produces a low ratio and indicates a highly efficient, ozone-productive airmass composition. Such a scenario indicates additional influences on the airmass composition.

Figure 9, Figure 10, and Figure 11 provide daily state-level EGU NOx emissions from states determined by EPA to be contributors to ozone formation in the western Michigan region compared to observed MDA8 ozone concentrations at the Muskegon, Holland, and Coloma monitors. Each line represents state total EGU NOx emissions as obtained from the Clean Air Markets Division[8] with red diamonds representing the MDA8 values at the monitors. The June 17-20, 2020 episode is denoted by the grey column. Of note, the figure suggests that on these exceptional event days, increased ozone levels were not the result of increased nearby EGU emissions.

[8] https://ampd.epa.gov/ampd/

If it were the case that the high observations were a result of higher EGU emissions in the region, the peaks of the representative state total lines would correlate to high values represented by the red diamonds. During the June 17-20, 2020 episode, EGU NOx emissions from Illinois, Indiana, Kentucky, Michigan, and Ohio are relatively low compared to the rest of the ozone season, although the observed MDA8 values are among the highest recorded for the year.

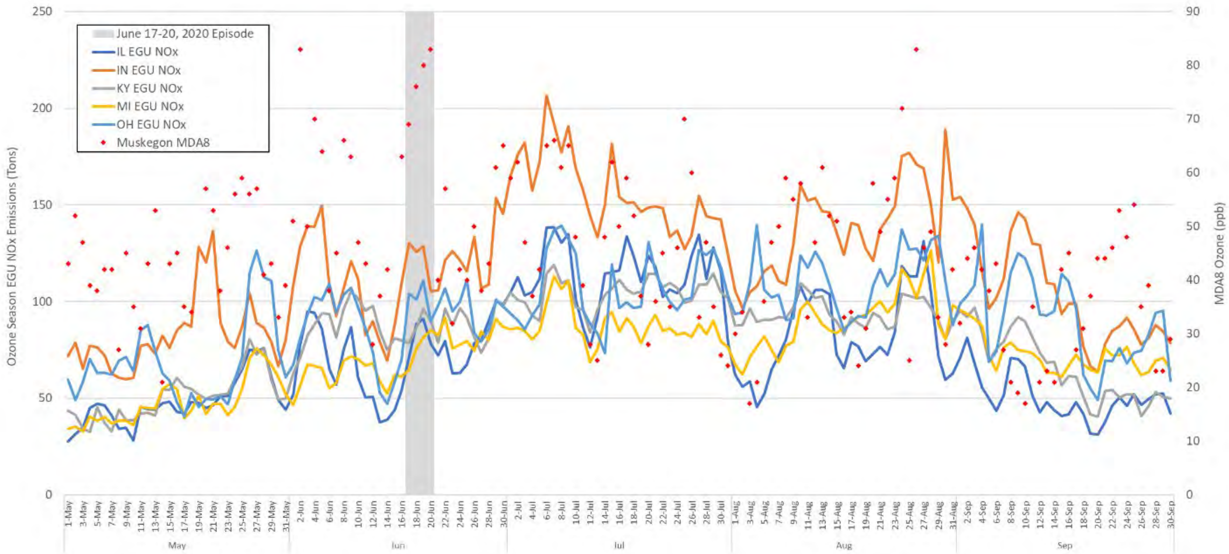

Figure 9. Daily 2020 Ozone Season EGU NOx Emissions and Muskegon Monitor MDA8 Ozone

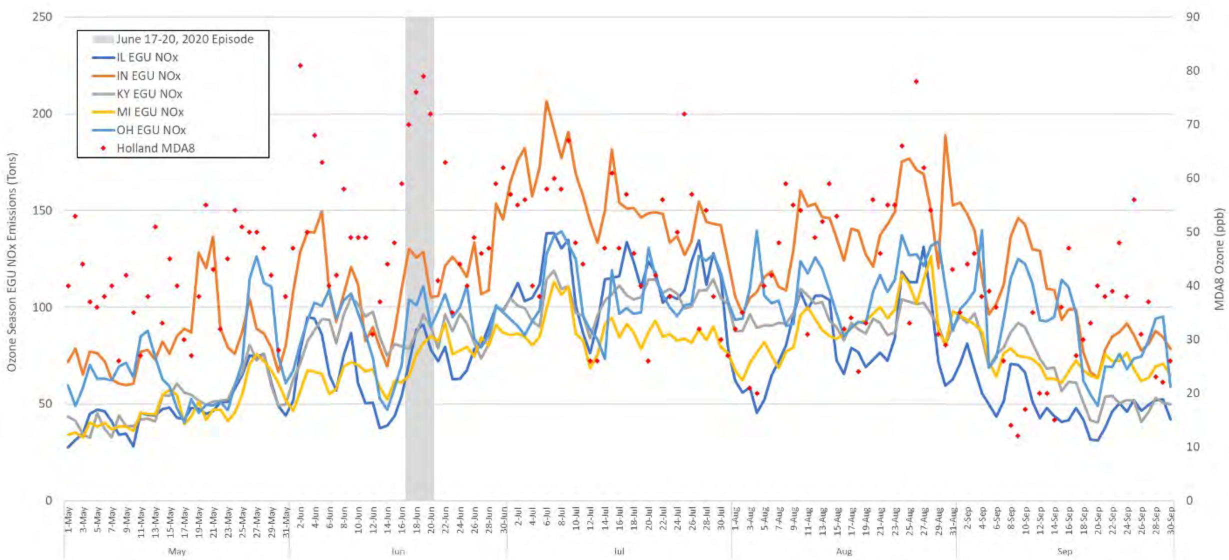

Figure 10. Daily 2020 Ozone Season EGU NOx Emissions and Holland Monitor MDA8 Ozone

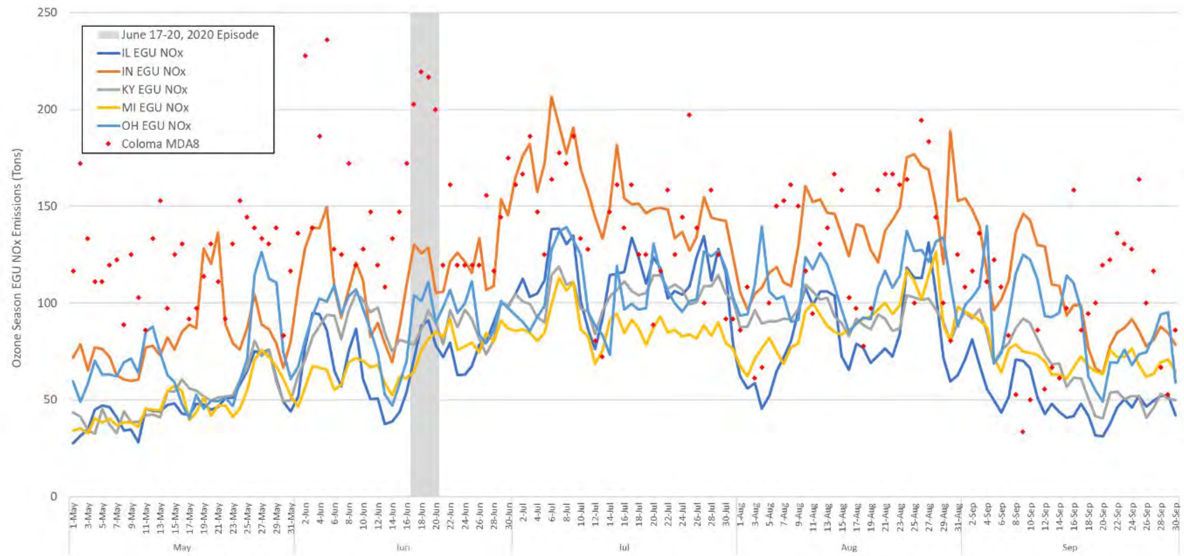

Figure 11. Daily 2020 Ozone Season EGU NOx Emissions and Coloma Monitor MDA8 Ozone

Daily EGU NOx/ozone ratios were also calculated to show that the high ozone levels were due to smoke-influenced ozone production, not emissions from upstream EGUs. Figure 12, Figure 13, and Figure 14 demonstrate that on the days of the selected episode (grey column representing June 17-20, 2020), the ratio of EGU NOx emissions compared to the MDA8 measurement at the three monitors was among the lowest measured during the 2020 ozone season (red bars). A high ratio of EGU NOx to MDA8 would be an indicator that EGU anthropogenic emissions sources played a significant role in ozone formation on that day. These figures provide clear indication that despite the relatively low EGU NOx emissions observed in the days leading up to the event days, ozone production rose uncharacteristically high. Therefore, the high ozone concentration on June 17-20, 2020 are not due to emissions produced by high or abnormal EGU emissions and may be attributed to smoke influenced ozone production.

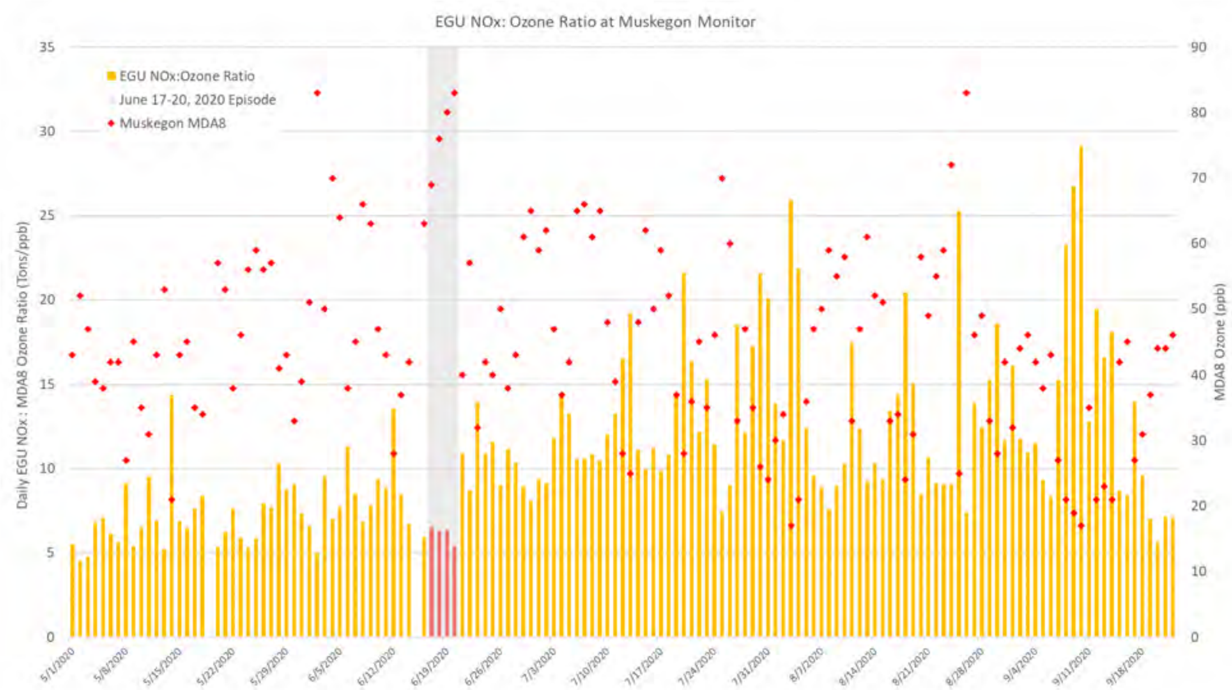

Figure 12. Daily 2020 Ozone Season EGU NOx Emission: Muskegon MDA8 Ozone Ratios

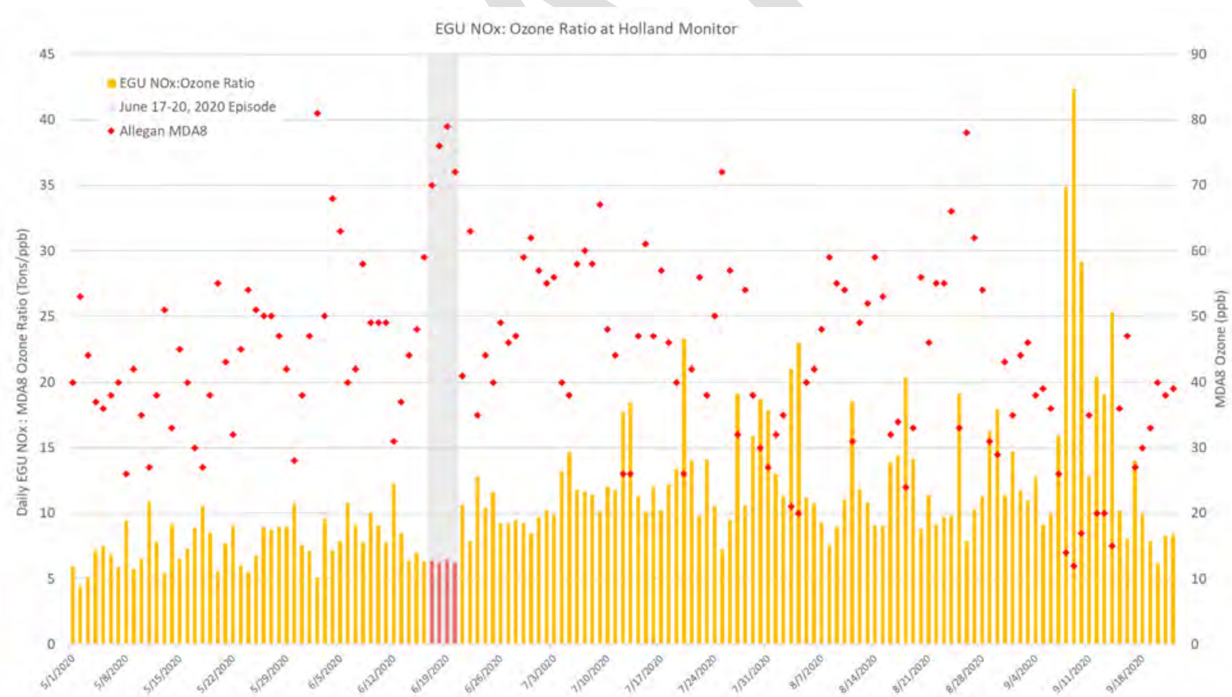

Figure 13. Daily 2020 Ozone Season EGU NOx Emission: Holland MDA8 Ozone Ratios

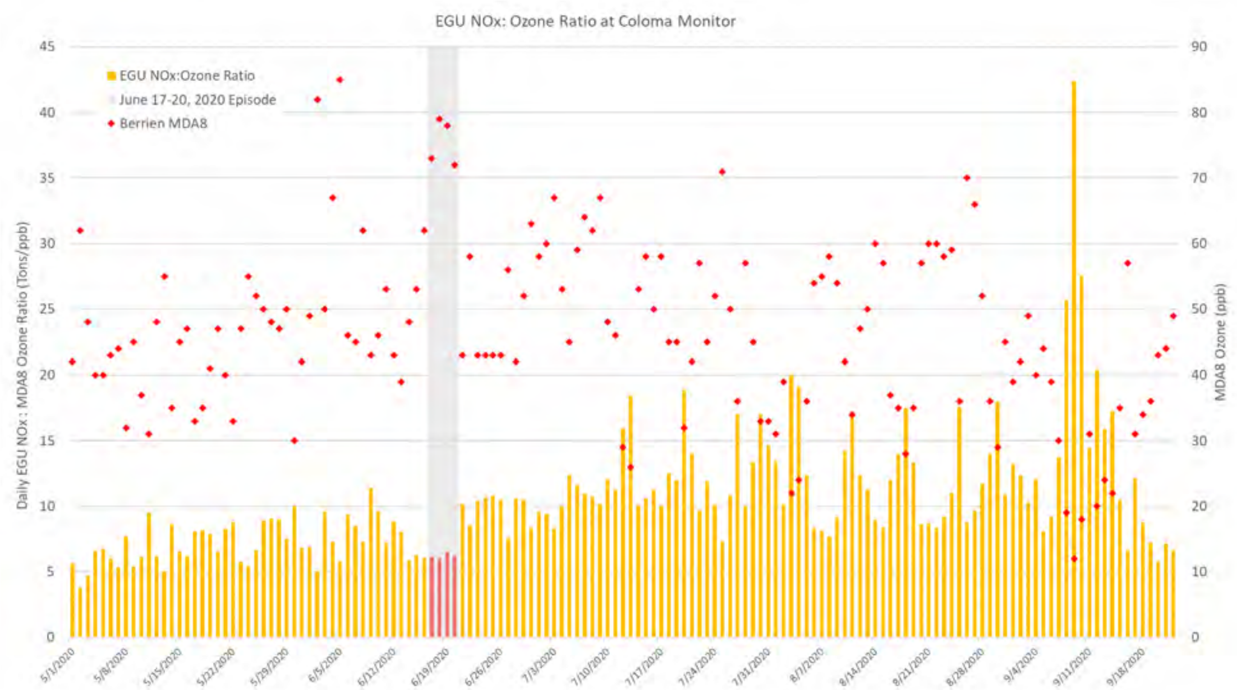

Figure 14. Daily 2020 Ozone Season EGU NOx Emission: Coloma MDA8 Ozone Ratios

Wildfire Descriptions

The Arizona 2020 wildfire season has burned just under 955,000 acres as of mid-November 2020, according to the Arizona Department of Forestry and Fire Management[9]. That value is almost double the 520,000 acres that burned in 3,627 fires over the previous *two* years combined (approximately 165,000 acres in 2018 and 385,000 acres in 2019)[10]. Figure 15 presents 2020 wildfires in Arizona with fires larger than 1,000 acres labeled individually. The Bush, Bighorn, and Mangum wildfires are circled.

Fire and smoke maps show large plumes of wildfire smoke emanating from a southwest Arizona wildfire complex burning June 5 – July 6, 2020. During June 11-15, 2020, this smoke was initially transported in an easterly flow across much of the southeastern United States. Then residual smoke and newly generated wildfire smoke were both transported to the Lake Michigan region during a period that included June 17-20, 2020. Three wildfires of note from this complex, the Mangum, Bush, and Bighorn wildfires, were among the largest in Arizona wildfire history with the Bush fire listed as the fifth-largest recorded in the state.

The Bush, Mangum, and Bighorn fires occurred following an unusually hot dry spring season. In April 2020, the Arizona Department of Forestry and Fire Management expected a "potentially active" fire season reminiscent of the 2019 season. Increased grass load from a wet winter was expected to contribute to an elevated risk of fire in the central Arizona deserts. Southwest Coordination Center Predictive Services forecasted an Above Normal risk for significant wildland fires from May through July for most of Arizona. They cited above-normal fine fuel loading in southern Arizona deserts and an active weather pattern through mid-June to support this risk.

[9] https://dffm.az.gov/
[10] National Interagency Coordination Center and the National Fire and Aviation Management Web Applications

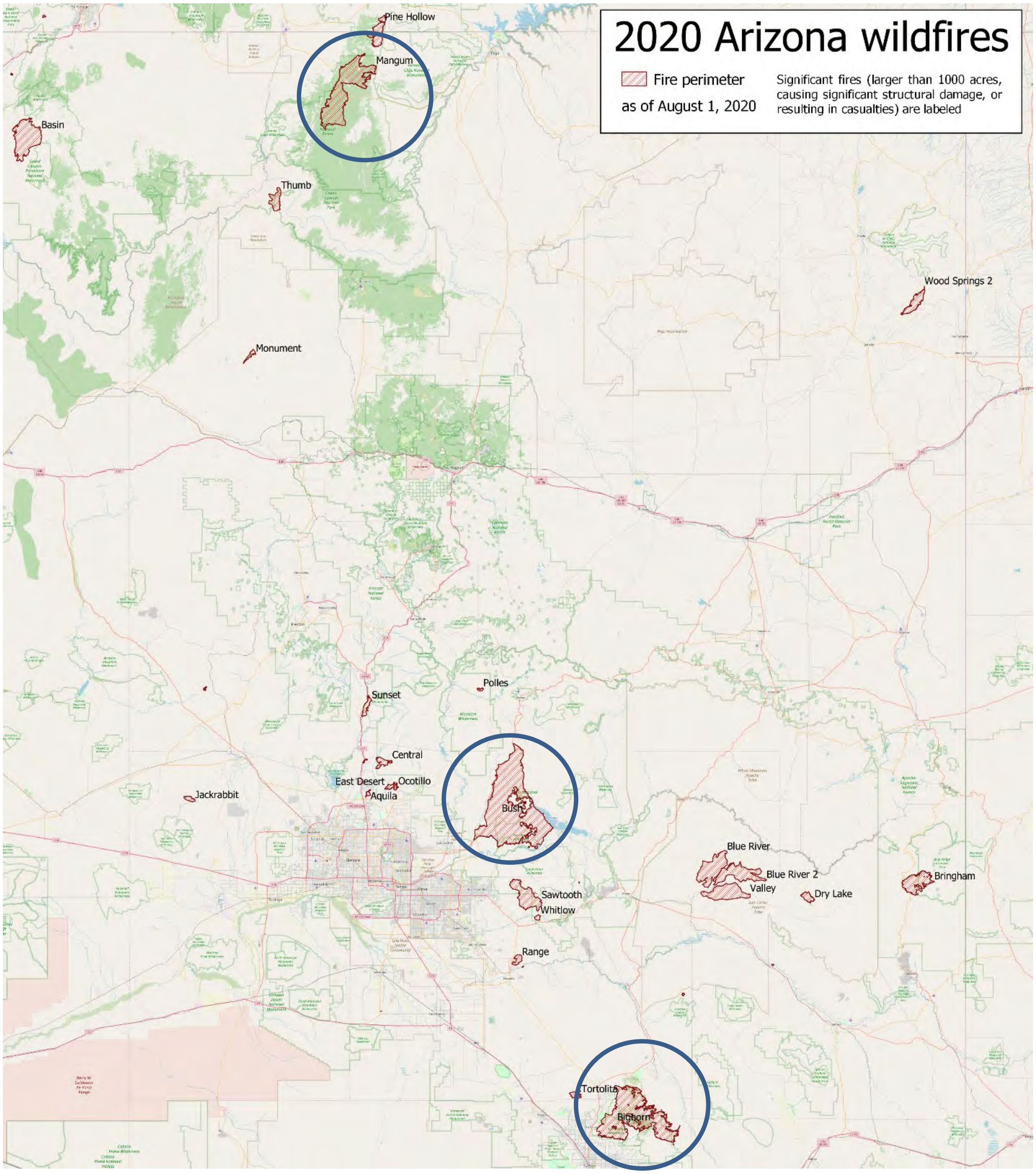

Figure 15. Perimeters of fires in the state of Arizona during 2020[11]. Fires larger than 1000 acres or otherwise significant are labeled. Mangum (top), Bush (middle), and Bighorn (bottom) wildfires are circled.

[11] Map data (c) OpenStreetMap (and) contributors, CC-BY-SA, Atomic7732.

The Bush Fire[12] was a human-caused wildfire that started in the Tonto National Forest northeast of Phoenix, Arizona. It burned 193,455 acres. The fire started on June 13, 2020, near the intersection of Bush Highway and SR 87 and was not fully contained until July 6, 2020.

Figure 16. Bush Fire in the Mazatzal Mountains as seen from Fort McDowell, AZ, on June 16, 2020[13].

[12] https://inciweb.nwcg.gov/incident/6773/
[13] CC BY-SA 4.0; Atomic7732

The Bighorn Fire[14] was a wildfire in the Santa Catalina Mountains north of Tucson, Arizona on the Coronado National Forest. It burned 119,987 acres until it was finally put out on July 23, 2020. A lightning strike from a storm on the evening of June 5, 2020, caused the fire.

Figure 17. The Bighorn Fire on the slopes of the Catalina Mountains as seen on June 12, 2020[15].

[14] https://inciweb.nwcg.gov/incident/6741/

[15] Kelly Presnell/Arizona Daily Star via AP

The Mangum Fire[16] was a wildfire that burned in Kaibab National Forest in Arizona. The fire, which started on June 8, 2020, was approximately 16 miles north of the North Rim of Grand Canyon National Park and burned a total of 71,450 acres. It was not fully contained until July 27, 2020. The exact cause of the fire remains under investigation; however, fire officials have confirmed it was human caused.

Figure 18. Smoke from the Mangum Fire as seen on June 12, 2020[17].

[16] https://inciweb.nwcg.gov/incident/6748/
[17] https://inciweb.nwcg.gov/incident/photographs/6748/24/

This news article describing the Bush, Bighorn, and Mangum fires provides additional information about the scale and impact:

Arizona reels as three of the biggest wildfires in its history ravage state – The Guardian, 2 July 2020[18]

> *"But on 5 June lightning ignited a wildfire that has grown to engulf over 118,000 acres. The fires are still only 58% contained. Called the Bighorn fire, it is the eighth-biggest in state history, and it has transformed the Catalinas into a hub for the study of the impacts of climate change. Nasa satellite photos show large scar marks left by the fire.*
>
> *At night you can see basically the outline of the fire on the mountain," said Courtney Slanaker, the executive director for the American Red Cross Southern Arizona, "and then during the daytime you're seeing that heavy smoke as it moves through different fuel sources on the mountain.*
>
> *And yet, Bighorn is just one of three fires that sit in the top 10 biggest wildfires in Arizona history.*
>
> *The Bush fire in the Tonto national forest, about 30 miles from Phoenix, now covers 193,000 acres and 98% is contained. It is the fifth-biggest in state history. Meanwhile, the Mangum fire burning in the Kaibab national forest now covers over 71,000 acres and 67% contained. The trio of fires are bigger than Washington DC, San Francisco, Baltimore, Chicago, Miami, Minneapolis and Manhattan combined."*

[18] https://www.theguardian.com/environment/2020/jul/02/arizona-wildfires

Conceptual Model of Ozone Formation and Transport from Wildfires

Wildfire smoke plumes contain gases including non-methane hydrocarbons (NMHCs), CO, NOx, and aerosols, which are all important precursors to photochemical production of tropospheric ozone and can travel thousands of kilometers. Smoke plume transport may cause areas far downwind of the fires to see greater enrichment of ozone compared to areas closer to the wildfires. Upper-level winds at the western Michigan monitors during the June 17-20, 2020 event originated from a south-southwestern direction, bringing with it smoke plumes attributed to multiple Arizona wildfire complexes.

Many variables, such as type of fuel or forest burned, plume path, and acreage burned, affect the intensity of the fire and ability of a plume to enhance downwind ozone production. High elevation desert coniferous forests, like those associated with Arizona wildfires, sustain high levels of invasive annual grasses whose abundance and continuity of fine fuels have been increasing with recent warmer, drier weather patterns and result in more frequent and larger fires.[19] Structurally, these woodlands (pinyon–juniper) are rather simple, characterized by being an open forest dominated by low, bushy, evergreen junipers[20].

The impact of wildfires on regional-scale atmospheric chemistry depends on the physical and chemical transformations that take place as fire emissions are transported, diluted, and exposed to chemical oxidants. Ozone and other oxidants can be formed along the way, and particle mass-loadings can grow or shrink[21]. Not all the factors that regulate these processes are well understood and individual fire plumes can have different behaviors.

The reasons for these complexities may have to do with how fast the plume was lofted and cooled or how efficiently NOx was converted to products such as peroxyacetyl nitrate (PAN). When mixed with urban emissions[22], it is clear that fire emissions often have broad-scale impacts on ozone formation[23] and can be decisive factors in triggering air quality exceedances.

Photo oxidation of the NOx and volatile organic compounds (VOCs) emitted by fire plumes shows complex behavior, sometimes leading to production of ozone[24]. Other cases have confirmed that the maximum ozone production is often observed substantially downwind of the fire, after the smoke plumes have aged for several days. Dreessen *et al*[25] noted in their analysis of a June 2015 wildfire that at peak smoke concentrations in Maryland, wildfire-attributable VOCs more than doubled, while non-NOx oxides of nitrogen (NOz) tripled. These findings suggest the long-range transport of NOx within the smoke plume. They also noted that ozone peaks a few days after the maximum smoke plume due to

---

[19] Balch, Jennifer K.; Bradley, Bethany A.; D'Antonio, Carla M.; [et al.]. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). Global Change Biology. 19(1): 173–183

[20] Dick-Peddie, William A. (1999). New Mexico Vegetation: Past, Present, and Future. University of New Mexico Press. p. 280. ISBN 0-8263-2164-X.

[21] Akagi, S. K., et al. (2012), Evolution of trace gases and particles emitted by a chaparral fire in California, Atmos. Chem. Phys., 12(3), 1397-1421, doi:10.5194/acp-12-1397-2012.

[22] Singh, H. B., C. Cai, A. Kaduwela, A. Weinheimer, and A. Wisthaler (2012b), Interactions of fire emissions and urban pollution over California: Ozone formations and air quality simulations, Atmos Environ., 56, 45-51, doi:10/1016/j.atmosenv.2012.03.046.

[23] Pfister, G. G., et al. (2006), Ozone production form the 2004 North American boreal fires, J. Geophys. Res., 111, D24S07, doi:10.1029/2006JD007695.

[24] Jaffe, D.; Wigder, N. Ozone production from wildfires: A critical review. Atmos. Environ. 51, 1−10, 2012.

[25] Dreessen, J. et. Al., Observations and impacts of transported Canadian wildfire smoke on ozone and aerosol air quality in the Maryland region on June 9–12, 2015. Journal of the Air & Waste Management Association, 66(9), 842-862, 2016.

ultraviolet light attenuation, lower temperatures, and non-optimal surface layer composition. Putero *et al*[26] observed the largest increases in ozone from fires five days (120 hours) after the initial pollutants were emitted from the fire (Figure 19).

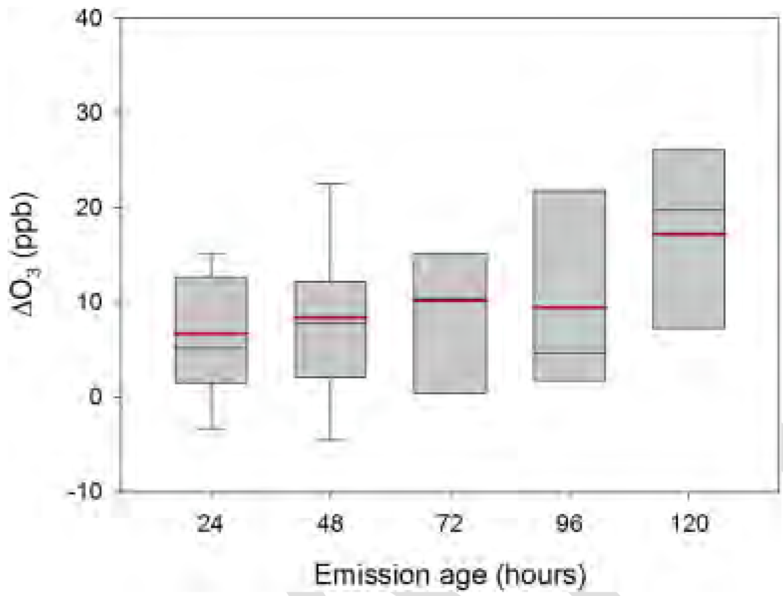

Figure 19. Ozone Enrichment by Age of Plume

[26] Putero, D. et. al., Influence of open vegetation fires on black carbon and ozone variability in the southern Himilayas, Environmental Pollution, vol 184, pp 597-604, 2014.

Meteorological Conditions Driving Smoke and Ozone Transport

Table 4 shows representative meteorological conditions[27], (between 9 AM and 9 PM ET) at the Muskegon County Airport (KMKG) in Muskegon County, from June 16-21, 2020 along with MDA8 ozone from the Muskegon monitor. Table 5 provides the same information for the West Michigan Regional Airport (KBIV) in Allegan County and Table 6 provides representative meteorological conditions for Southwest Michigan Regional Airport (KBEH) in Benton Harbor, Berrien County.

The rising temperatures and relatively low winds shown for June 17-20, 2020, are typical ozone formation characteristics in the region. The overall collective of the readings indicates a low-level high-pressure system in the region. The change in wind direction and decrease in temperatures in the days following indicate an airmass change event at this location associated with the lowering of ozone concentrations.

Table 4. Meteorological Conditions June 16 to 21, 2020 at KMKG with MDA8 from Muskegon monitor.

Variable	6/16/20	6/17/20	6/18/20	6/19/20	6/20/20	6/21/20
Maximum Temperature (F)	79	82	84	88	89	76
Surface Wind Direction (degree direction)	239	239	252	219	168	261
Avg Wind Speed (mph)	4.1	4.9	4.1	3.5	5.8	4.0
MDA8 Ozone (ppb)	63	69	76	80	83	40

Table 5. Meteorological Conditions June 16 to 21, 2020 at KBIV with MDA8 from Holland monitor.

Variable	6/16/20	6/17/20	6/18/20	6/19/20	6/20/20	6/21/20
Maximum Temperature (F)	80	83	85	89	89	77
Surface Wind Direction (degree direction)	262	265	238	257	221	271
Avg Wind Speed (mph)	3.3	3.5	4.0	3.2	5.7	3.8
MDA8 Ozone (ppb)	59	70	76	79	72	41

Table 6. Meteorological Conditions June 16 to 21, 2020 at KBEH with MDA8 from Coloma monitor.

Variable	6/16/20	6/17/20	6/18/20	6/19/20	6/20/20	6/21/20
Maximum Temperature (F)	78	83	82	88	91	78
Surface Wind Direction (degree direction)	195	238	275	292	220	270
Avg Wind Speed (mph)	3.0	3.0	2.9	2.7	5.6	3.3
MDA8 Ozone (ppb)	62	73	79	78	72	43

[27] https://mesonet.agron.iastate.edu/

Surface pressure[28], upper air 700 millibar (mb), and 850 mb height maps[29], where long range transportation can occur, for June 17-20, 2020, are shown in Figure 20 through Figure 23.

Figure 24 though Figure 34 provide daily ozone AQI and 48-hour HYSPLIT back-trajectory plots for June 17, 18, 19, and 20, 2020 of KMKG, KBIV, and KBEH, respectively. These figures demonstrate the movement of the enhanced ozone concentrations associated with the transported wildfire smoke plume and indicate the movement of air concentrations observed at the monitors on these days.

These figures also provide surface wind roses[30] and representative 12km resolution North American Mesoscale Forecast System (NAM12)-modeled surface and upper air wind roses that show the prevailing wind directions divided into sectors around the compass with due north at the top. The longer "petals" of the rose represent sectors where the wind direction is more prominent. Overlaid on these petals are color bars representing specific ranges of wind speed for each wind direction sector. The upper air wind direction and speed were informed using upper air soundings at 700 mb through 850 mb pressures from KDTX, representative of western Michigan, for 8 am EDT (12Z) on June 17, 18, 19, and 20, 2020.

Soundings from the upper air station at the Detroit/Pontiac (KDTX) station,[31] and corroborating NAM12-modeled conditions representative of KMKG, KBIV, and KBEH[32], representing western Michigan's upper air conditions on June 17-20, 2020, are provided in Figure 36 through Figure 39.

The surface maps show high-pressure system-dominated meteorological conditions during the event with the wind roses demonstrating a surface level wind direction shift from the east-northeast to the west-southwest during the days of the episode. Moving through the episode period, meteorological conditions and 48-hour back trajectories indicated the transport of air from the east-northeast until June 19 and 20, 2020 when air stagnation and light surface winds were observed and the local airmass initiated a recirculation pattern with western-drawn wildfire smoke resident in the region. Maximum temperatures rose during the event period, peaking on June 20. These conditions proved favorable to ozone formation and with the presence of wildfire smoke in the area, caused enhanced ozone concentrations.

Upper air and surface winds transported residual wildfire smoke pollutants from the Ohio Valley as well as freshly- generated wildfire plume pollutants into the upper Midwest between June 16 and June 21, 2020. On June 16 and June 17, a high-pressure region directed the latter wildfire plumes to the north of Michigan while continuing to pull smoke from the south. The movement of the high to the northeast on June 18 allowed these precursors to be transported into the area both directly from the fire to the southwest and from the recirculation region to the northeast on June 20, 2020. A low ceiling of wildfire smoke was observed, and temperature inversions capped the ozone precursor plume until conditions became favorable for a late morning mixing of atmospheric layers.

[28] Id.
[29] http://www.spc.noaa.gov/obswx/maps/
[30] https://mesonet.agron.iastate.edu/
[31] https://rucsoundings.noaa.gov/
[32] https://www.ready.noaa.gov/READYamet.php

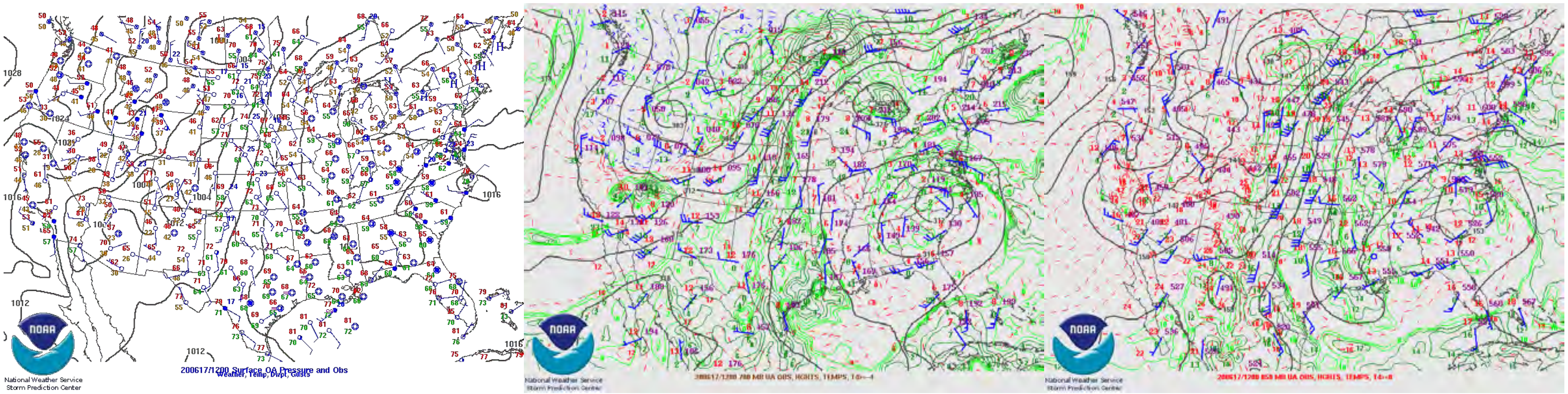

Figure 20. Surface (left), 700 mb (middle), and 850 mb (right) Pressure Patterns at 8 am EDT with Winds for June 17, 2020

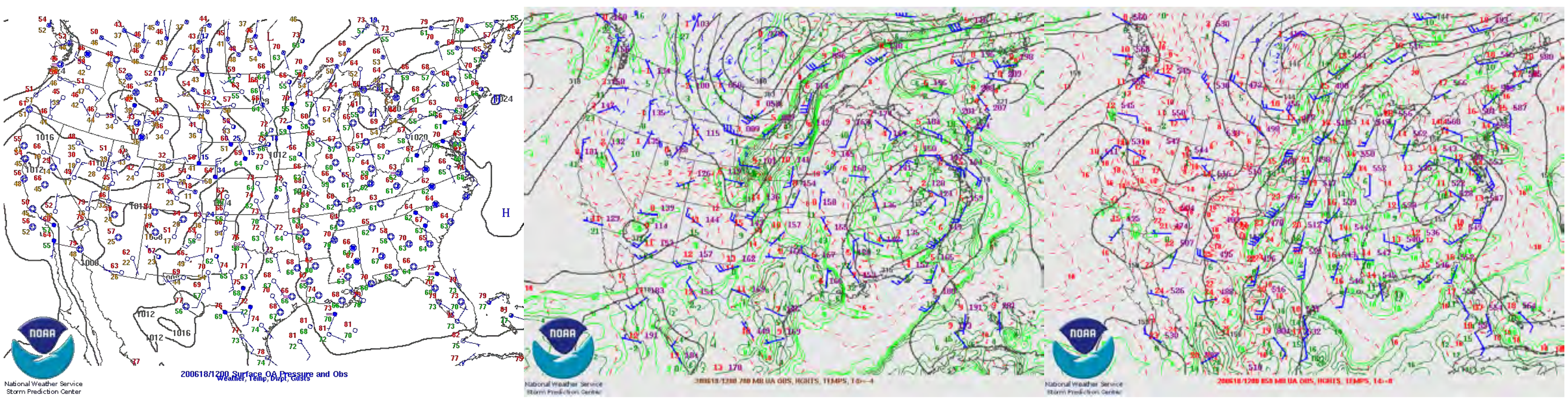

Figure 21. Surface (left), 700 mb (middle), and 850 mb (right) Pressure Patterns at 8 am EDT with Winds for June 18, 2020

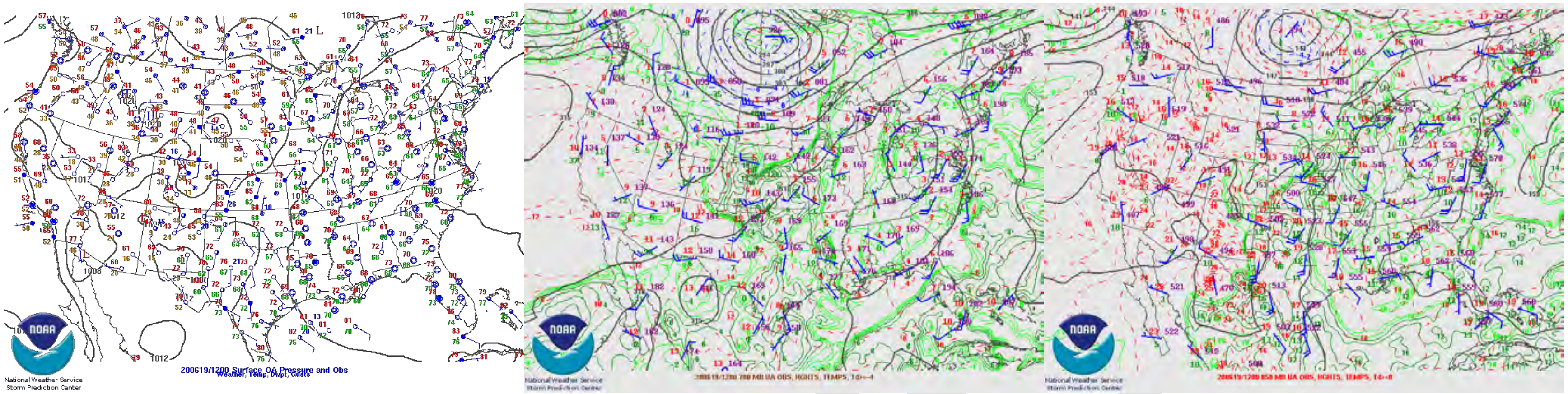

Figure 22. Surface (left), 700 mb (middle), and 850 mb (right) Pressure Patterns at 8 am EDT with Winds for June 19, 2020

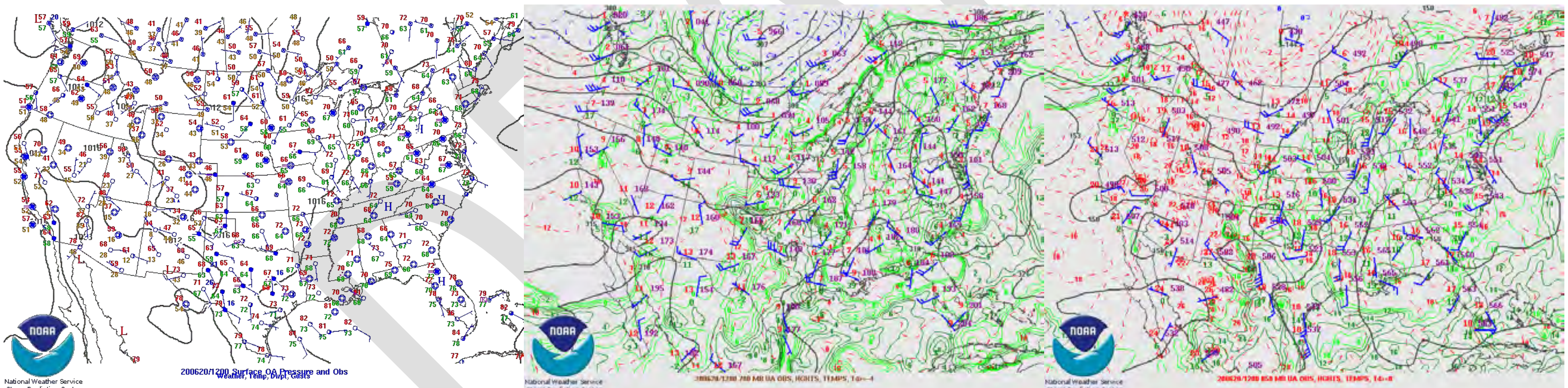

Figure 23. Surface (left), 700 mb (middle), and 850 mb (right) Pressure Patterns at 8 am EDT with Winds for June 20, 2020

As is seen at each of the three monitors starting on June 17, 2020 (Figure 24, Figure 28, and Figure 32), the AQI maps demonstrate the movement of polluted air associated with the smoke plumes from the west. This differs from June 19 and 20, 2020 (Figures 26, Figure 27, Figure 30, Figure 31, Figure 34, and Figure 35) in the fact that wind direction begins to change and wind speeds begin to decrease creating a stagnation event that recirculates the transported wildfire smoke over the western Michigan area. On each of the days of the episode, a wall of poor air quality and high ozone concentrations continues moving with the wildfire smoke plume and is seen transported from the west and south of the NAA.

These wind roses indicate the initial movement of moderate winds from the east-southeast on June 17 and 18, 2020 bringing the regionally polluted air already present from the smoke plume into the NAAs. On June 19, 2020, we see the shifting winds from the west-southwest, consistent with the continually approaching wildfire smoke plume until on June 20, 2020, winds at the surface begin to stagnate and develop a recirculation pattern which the elevated winds increase drawing air from the smoke plume transported over Lake Michigan into the NAAs.

The sounding plots for June 17-20, 2020 provided below show each day's modeled morning (left side of Figure 37, Figure 38, and Figure 39) and evening (right side of figures) temperature profiles. The vertical temperature profiles for the mornings showed temperature inversions at about 600 meters (m) above ground level (circled in Figures), the same general altitude as the smoke plume. Evening temperature profiles for June 17-20, 2020 indicate that more vigorous vertical mixing occurred up to the height of the cap. This further supports mixing of the smoke plume to the surface. Smoke that was transported in the upper layer winds and arrived from the west would have been mixed with surface layer air and would have impacted ozone observations on June 17-20, 2020.

The mixing presented in these soundings is further corroborated with National Weather Service (NWS) Fire Weather Planning Forecasts which provide twice daily predictions of mixing heights and smoke dispersion factors for the episode period of June 17-20, 2020. In the reporting from NWS, predictions of mixing heights of up to 1200 m AGL mixed to ground level are presented in Figure 40 and Figure 41. This is consistent with observed smoke aerosols in the 1000-2000 m AGL altitude presented later in this document.

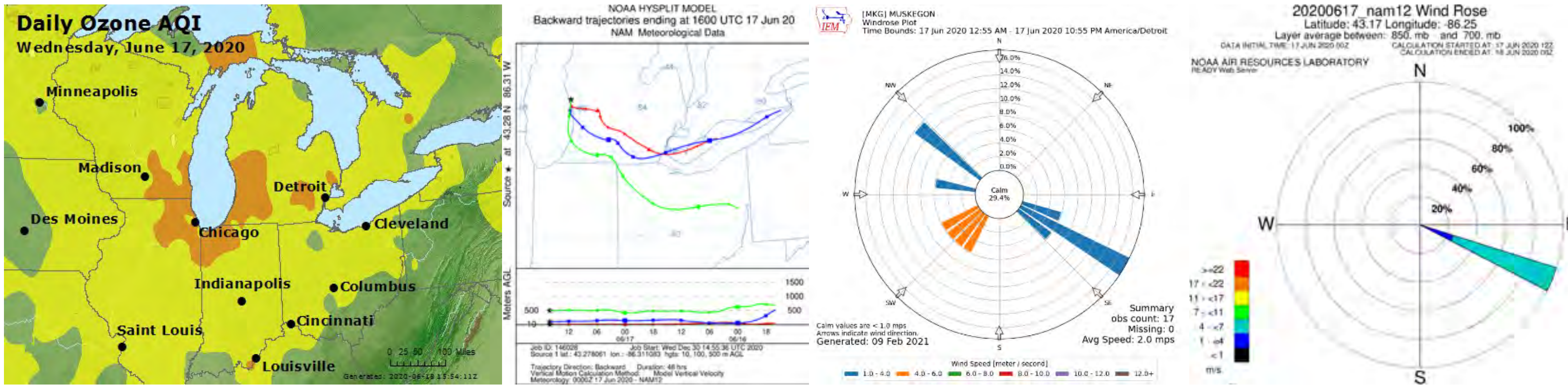

Figure 24. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Muskegon NAA on June 17, 2020.

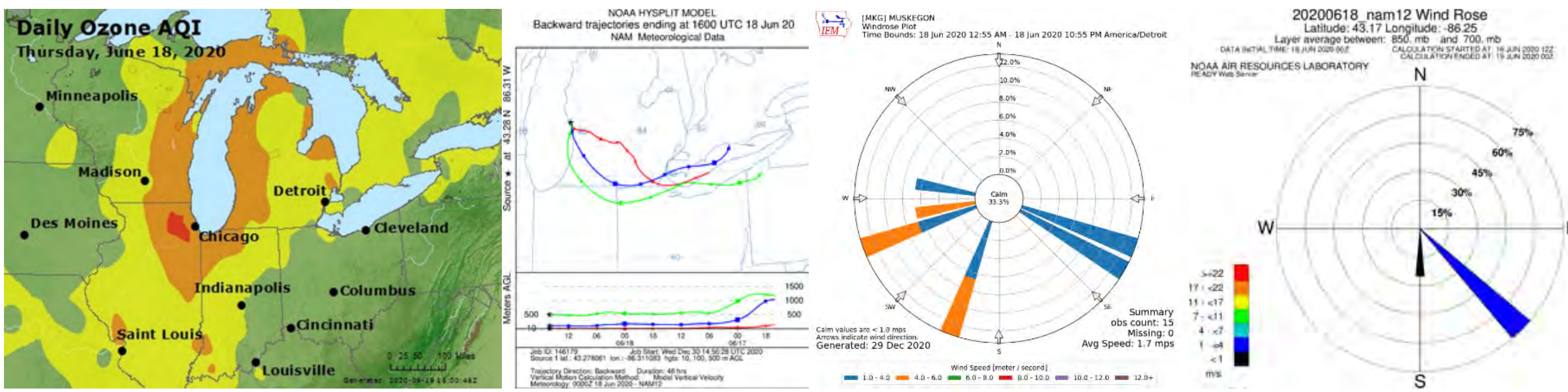

Figure 25. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Muskegon NAA on June 18, 2020.

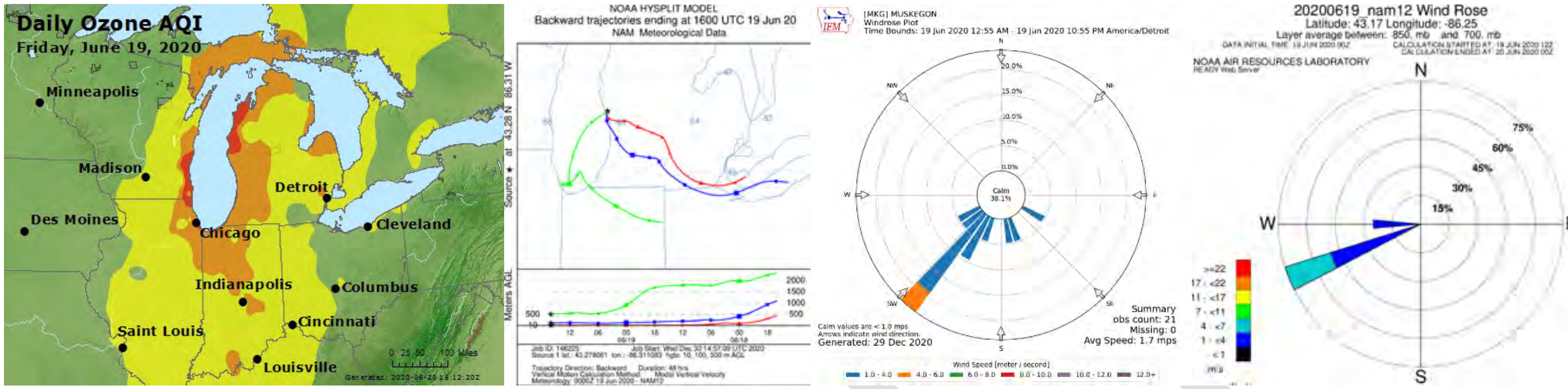

Figure 26. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Muskegon NAA on June 19, 2020.

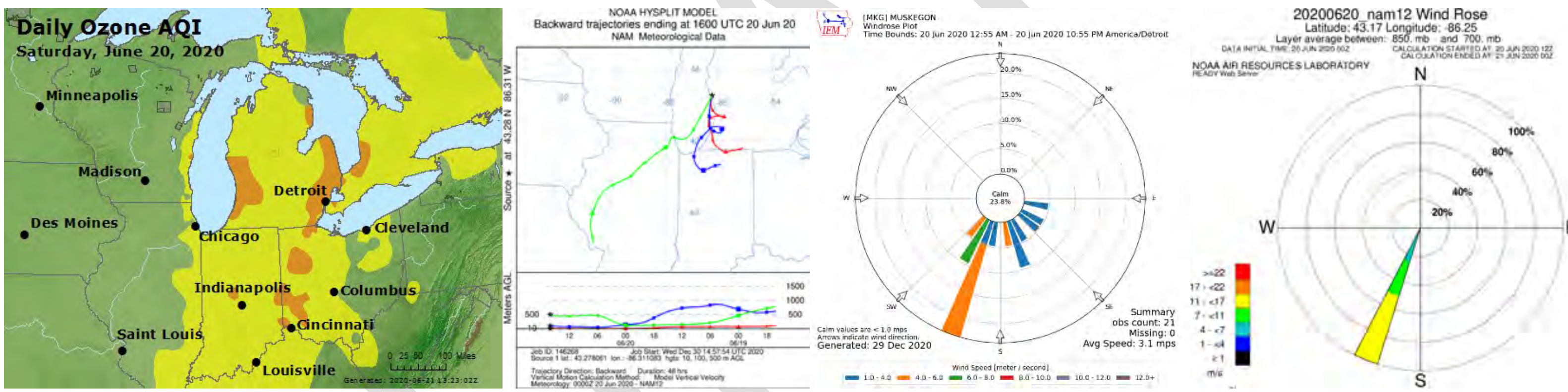

Figure 27. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Muskegon NAA on June 20, 2020.

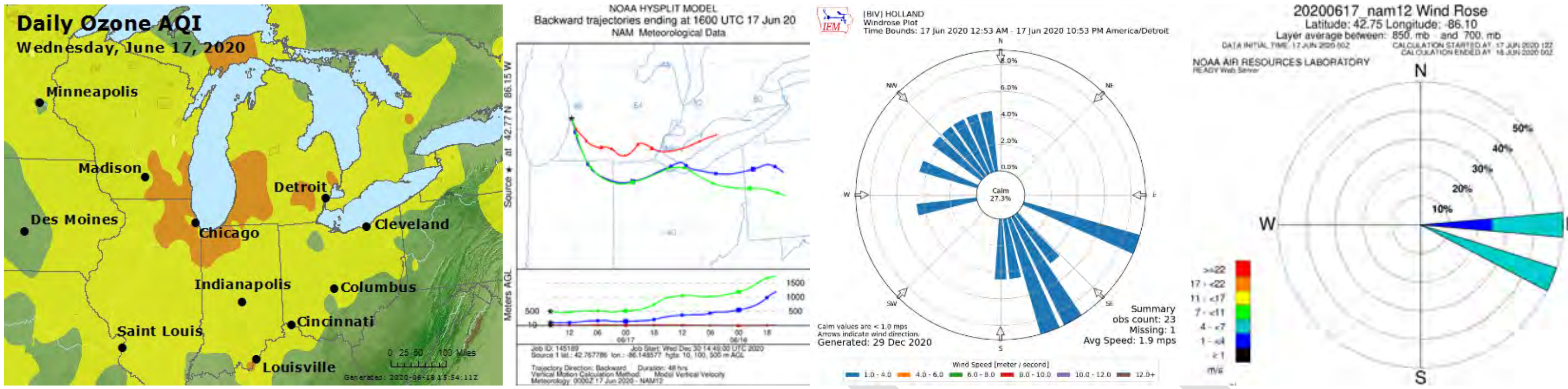

Figure 28. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Allegan NAA on June 17, 2020.

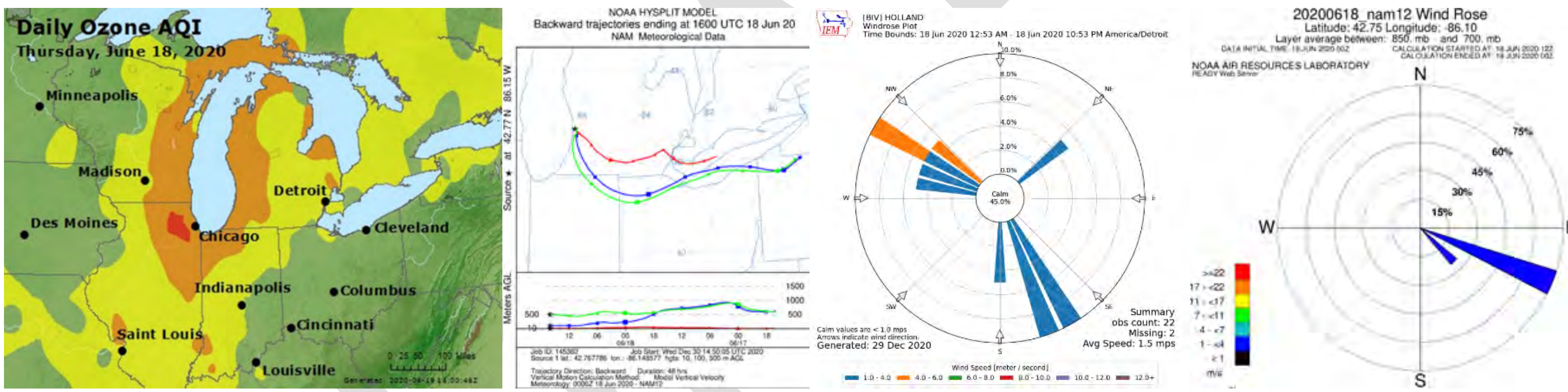

Figure 29. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Allegan NAA on June 18, 2020.

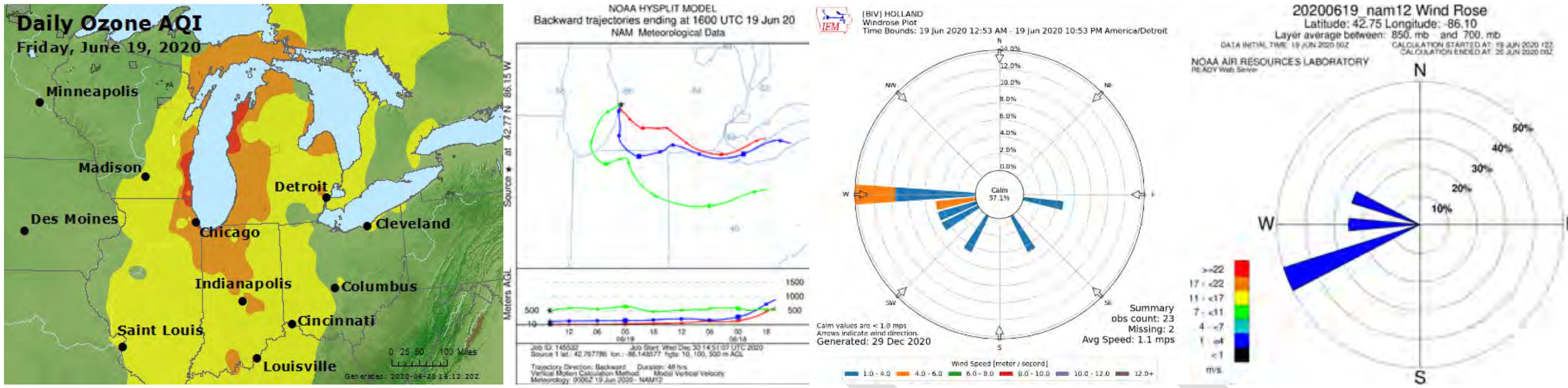

Figure 30. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Allegan NAA on June 19, 2020.

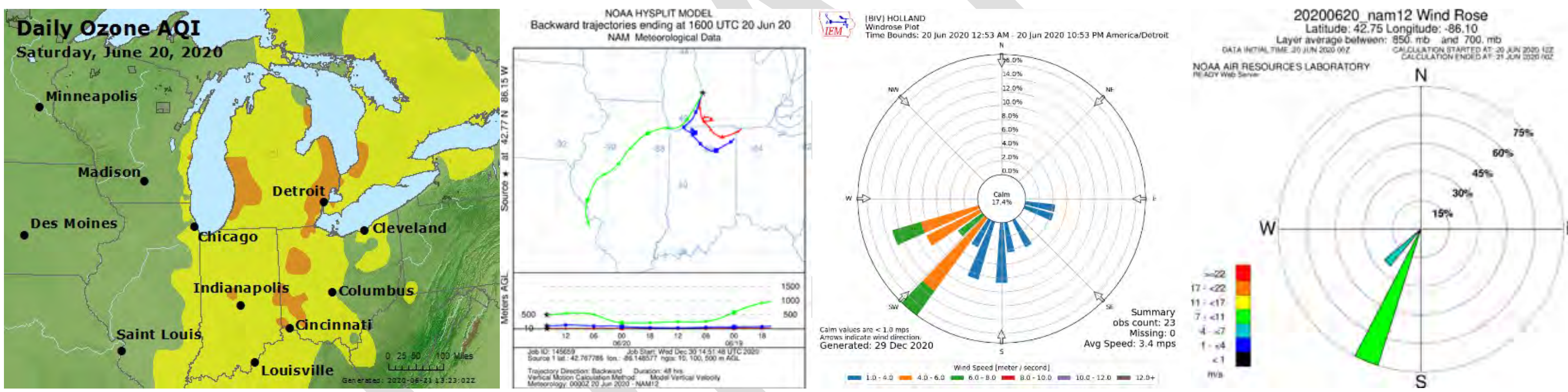

Figure 31. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Allegan NAA on June 20, 2020.

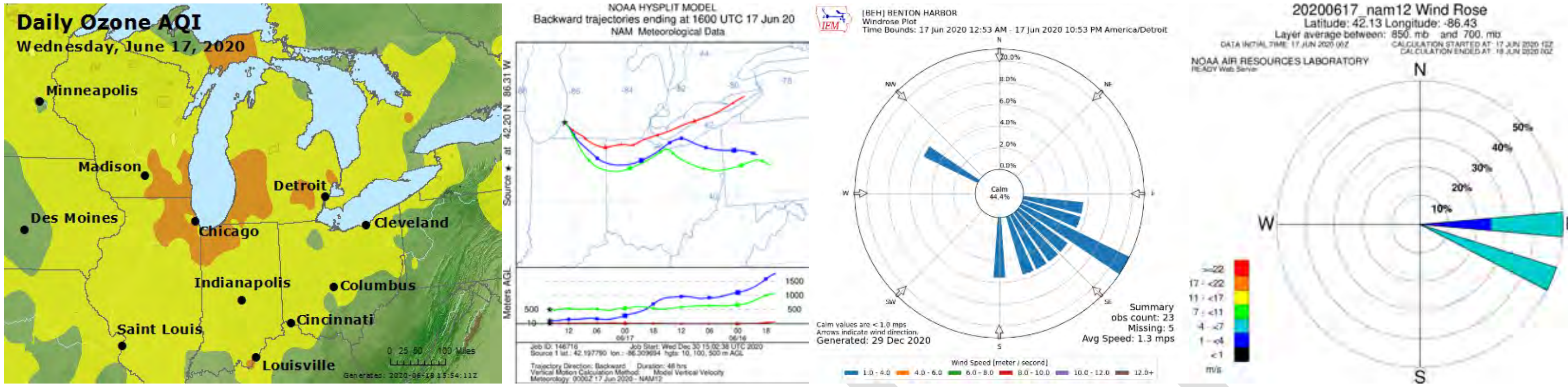

Figure 32. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Berrien NAA on June 17, 2020.

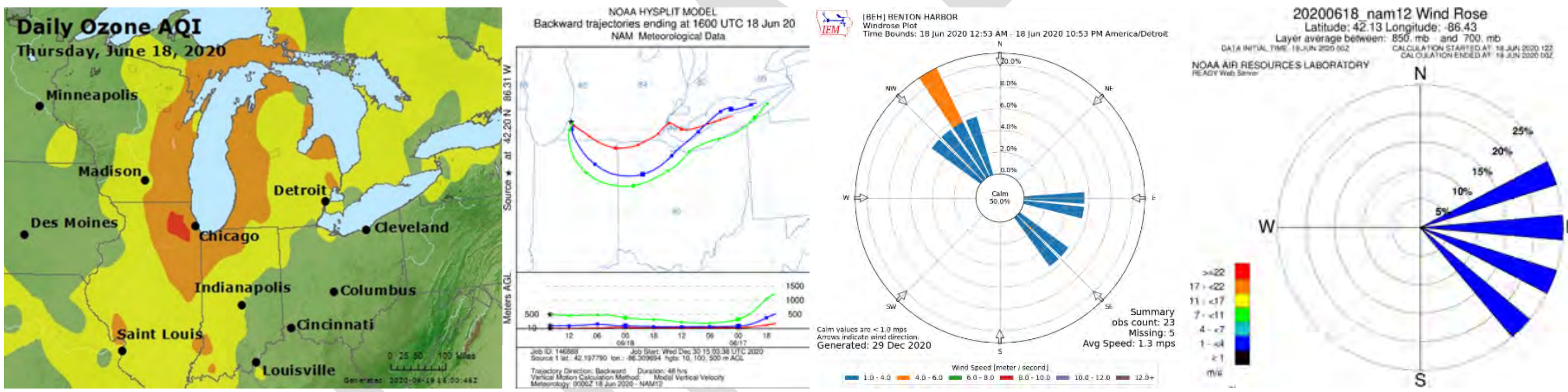

Figure 33. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Berrien NAA on June 18, 2020.

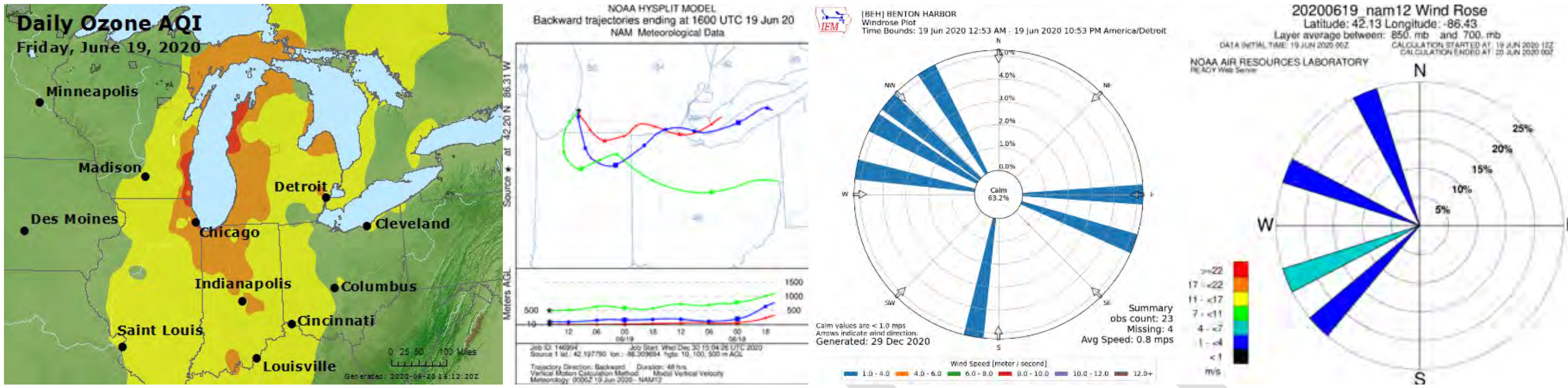

Figure 34. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Berrien NAA on June 19, 2020.

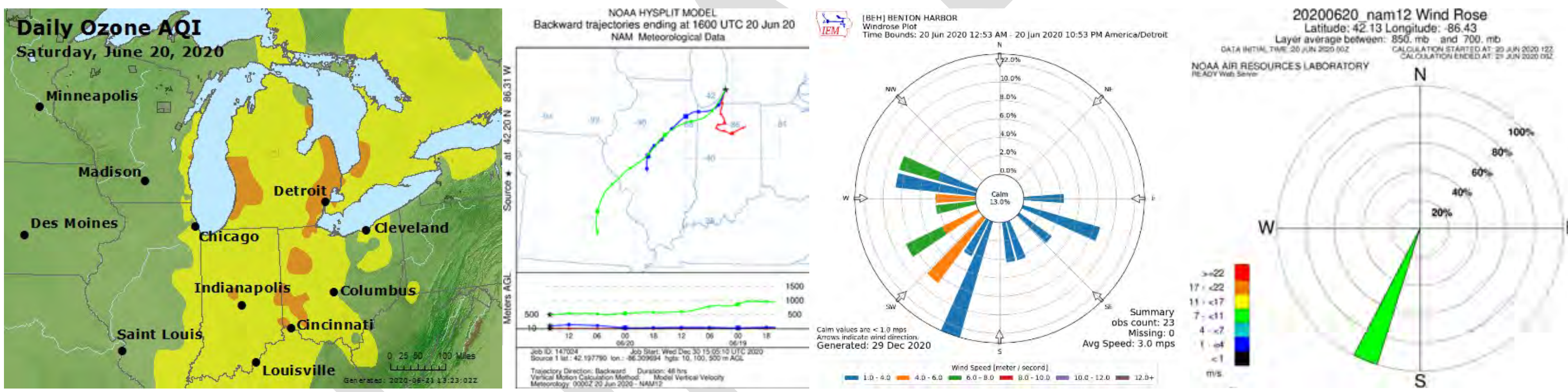

Figure 35. Ozone AQI, 48-hour HYSPLIT back trajectories, surface, and upper air wind roses for Berrien NAA on June 20, 2020.

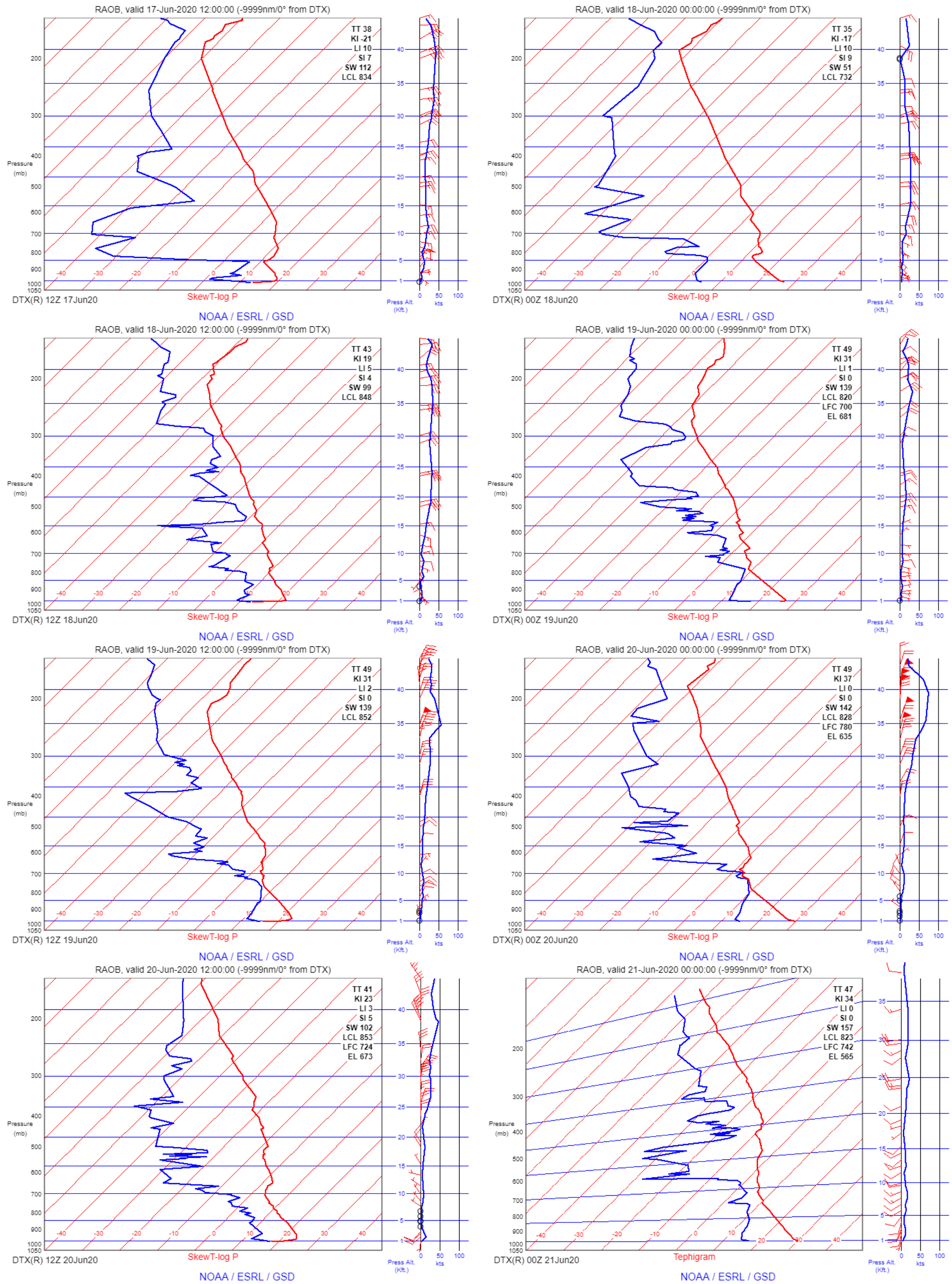

Figure 36. 8 am (left) and 8 pm (right) EDT sounding at DTX (left) on June 17 (top) through June 20 (bottom), 2020.

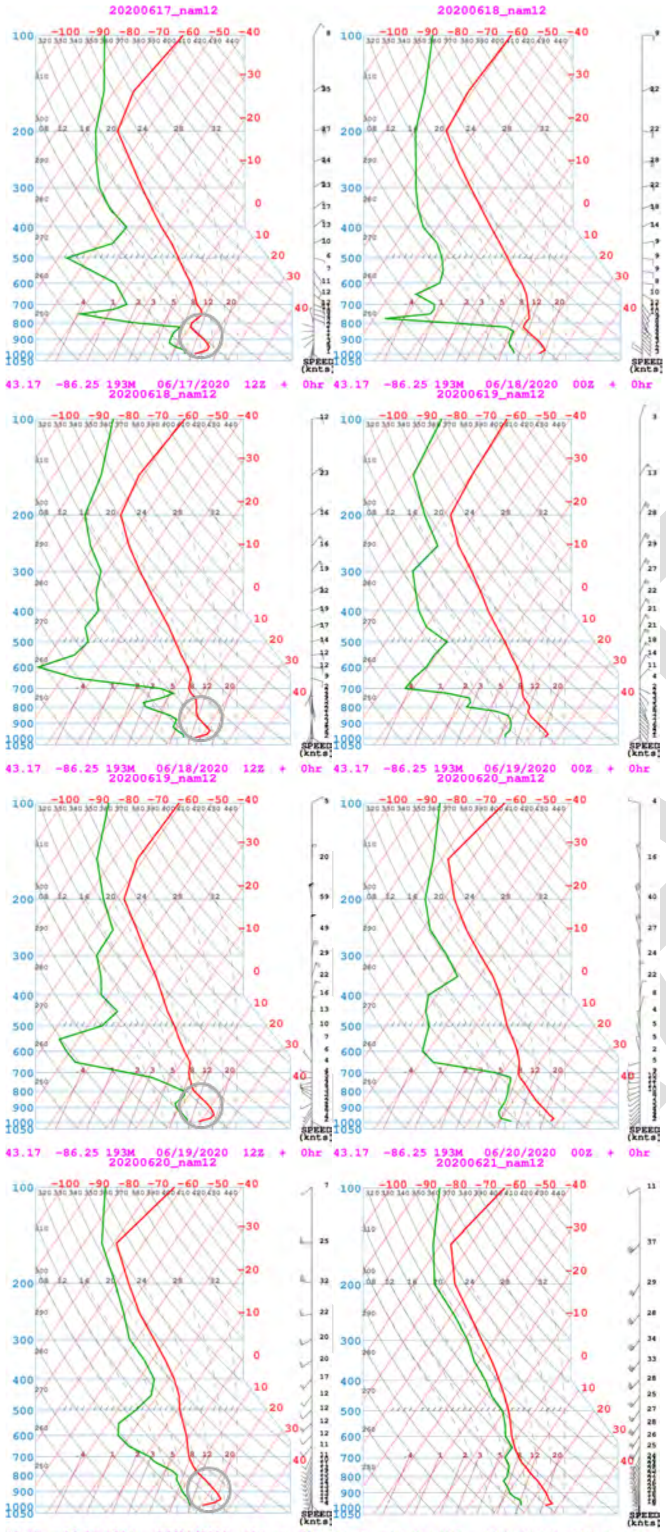

Figure 37. 8 am (left) and 8 pm (right) NAM12 modeled sounding at MKG on June 17 (top) through June 20 (bottom), 2020.

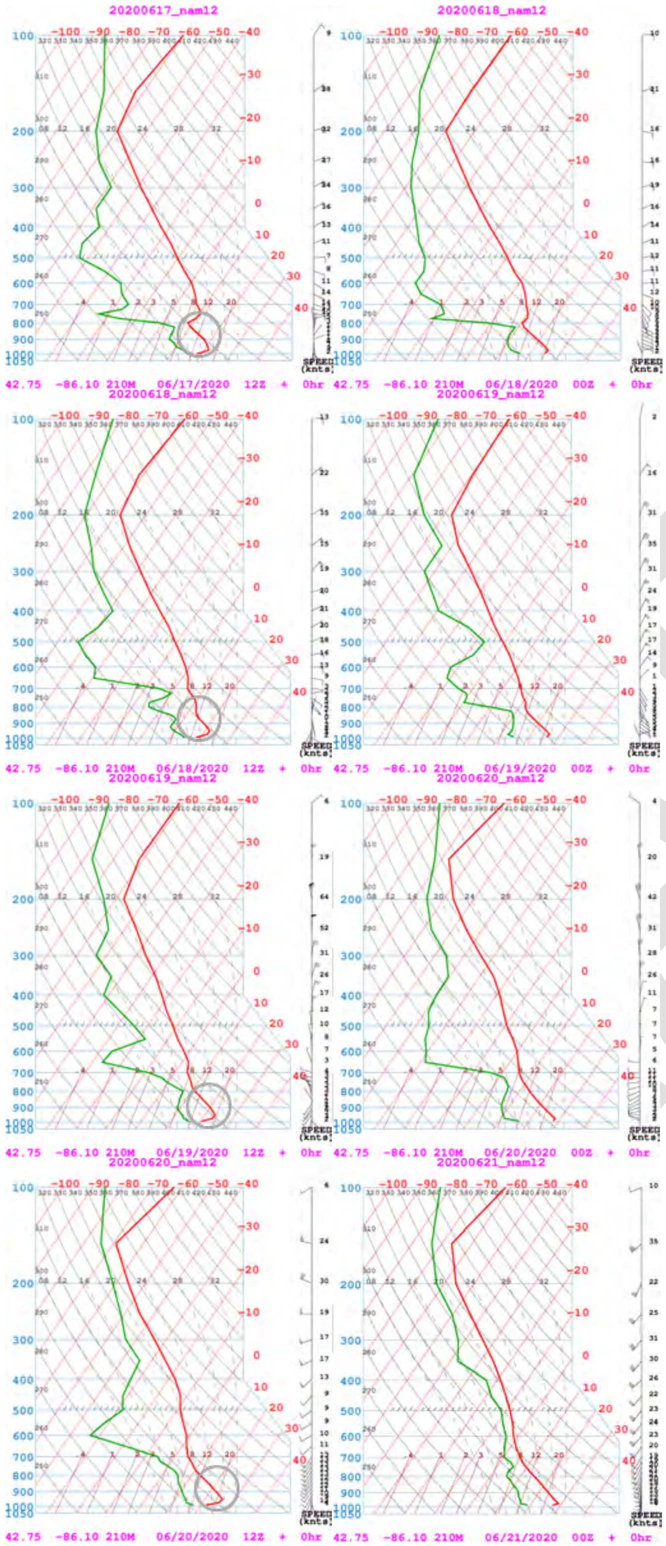

Figure 38. 8 am (left) and 8 pm (right) NAM12 modeled sounding at BIV on June 17 (top) through June 20 (bottom), 2020.

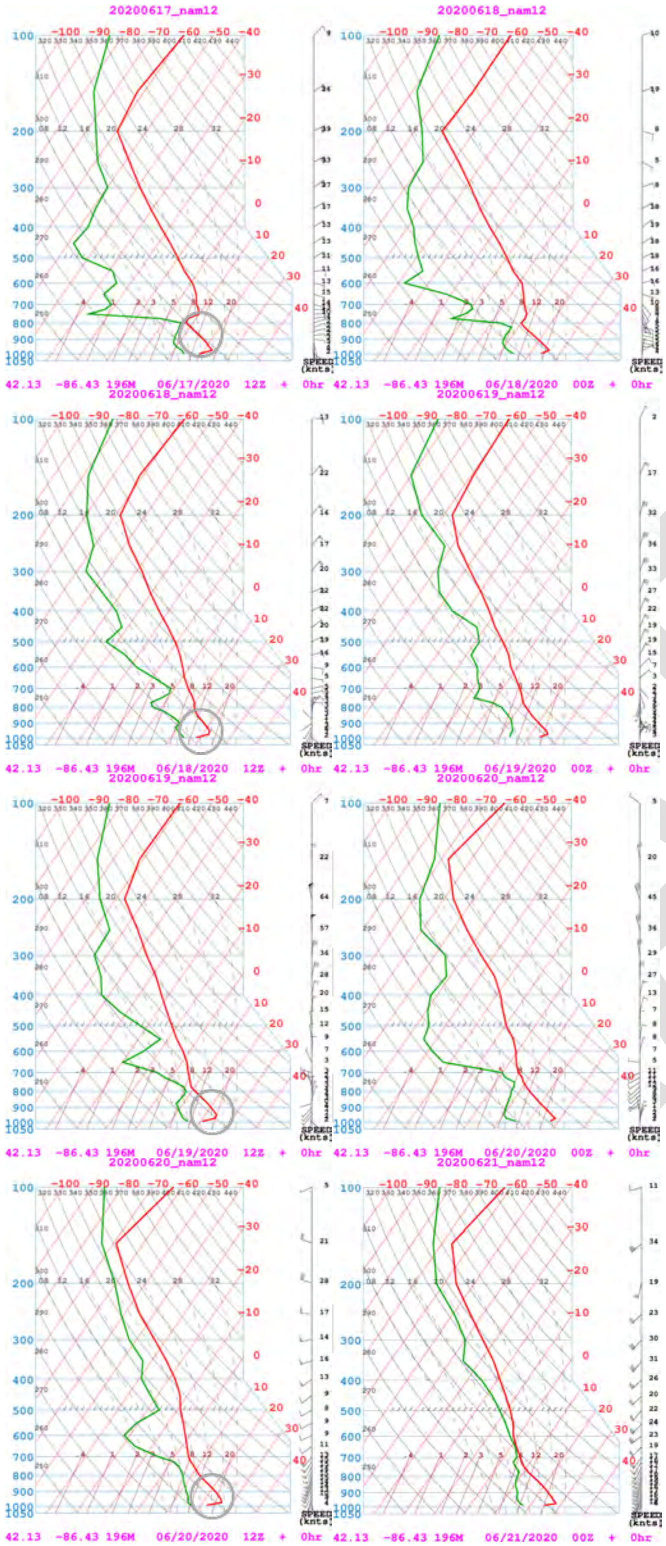

Figure 39. 8 am (left) and 8 pm (right) NAM12 modeled sounding at BEH on June 17 (top) through June 20 (bottom), 2020.

In the absence of any measured planetary boundary layer (PBL) height measurements at individual monitors, daily predicted values provided in Fire Weather Planning Forecasts[33] developed by the National Weather Service (NWS) can provide additional evidence related to the transport of smoke. These reports use meteorological parameters such as relative humidity, wind speed and direction, mixing heights, and soil moisture to determine whether conditions are favorable for fire growth and smoke dispersion. The National Weather Service issues several fire weather products on both a daily basis and when conditions warrant. Despite the limitations of frequency in the NWS day-night forecasts, these data provide crucial insight into the daily boundary layer dynamics at the nonattainment areas in Western Michigan.

Figure 40 provides an extract from a fire weather planning forecast for Southwest Lower Michigan as prepared by the NWS Grand Rapids, MI (KGRR) on the afternoon of June 1, 2020. From the aggregate of reports like these, NWS forecasts throughout the period predicted that mixing heights would fall to 0 – 200 m during the evening and overnight hours and then rise to altitudes of 2000 ft AGL or higher during the midday periods indicating a vigorous mixing from elevated layers. In combination with evidence presented later showing that smoke was observed in the free troposphere at these elevations (1000 - 2000 m AGL), these plots indicate a connection between the surface and the observed smoke aerosols in aloft layers. These were also accompanied with “poor to fair” smoke dispersal values (0 – 300), a numerical indicator of how well and how rapidly smoke will be dispersed (good to excellent) or if stagnant conditions exist (poor to fair), across all days when the smoke enhanced ozone exceedances were recorded (Figure 41).

We contrast these findings with a typical non-event ozone exceedance (June 2, 2020) in the region and immediately note the differences in the day-night mixing predicted and leading up to the event (Figure 42). Unlike the smoke enhanced events of June 17-20, 2020 which predicted a day-night variance in mixing heights, NWS predicted almost no change in mixing height leading up to the day of the June 2, 2020 ozone exceedance. This is an indicator of little vertical mixing and conditions where the ozone was likely maintained at low levels for extended periods as the buildup of ozone precursor emissions reacted favorably for ozone formation. This buildup analysis is also demonstrated in later sections of this document.

[33] https://www.weather.gov/fire/

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270
FNUS53 KGRR 181953
FWFGRR

Fire Weather Planning Forecast for Southwest Lower Michigan
National Weather Service Grand Rapids MI
353 PM EDT Thu Jun 18 2020

.DISCUSSION...
High pressure will continue the dry pattern for both tonight and
Friday. It is possible we even remain dry through Friday night.
Better chances for rain will come this weekend from Saturday through
Saturday night and into Sunday. The highest chances for showers and
thunderstorms will come from Saturday evening into Sunday. It will
remain warm into the weekend with lows tonight in many areas holding
into the 60s while rising to near 90 on Friday.

.TONIGHT...
Sky/weather.........Clear (0-10 percent).
Min temperature.....55-60.
   24 hr trend......9 degrees warmer.
Max humidity........88-93 percent.
   24 hr trend......Unchanged.
20-foot winds.......East winds up to 5 mph.
Haines Index........4 or low potential for large plume dominated
                    fire growth.
Mixing height.......Near Lake Michigan, 0-200 ft AGL.
Mixing height....... inland, 4300-4800 ft AGL decreasing to 0-100
                    ft AGL overnight.
Transport winds.....Southeast around 5 mph.
Smoke dispersal.....0-100 (poor).
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Figure 40. Extract of NWS fire weather planning forecast from Grand Rapids at 3:53 PM EDT on June 18, 2020.

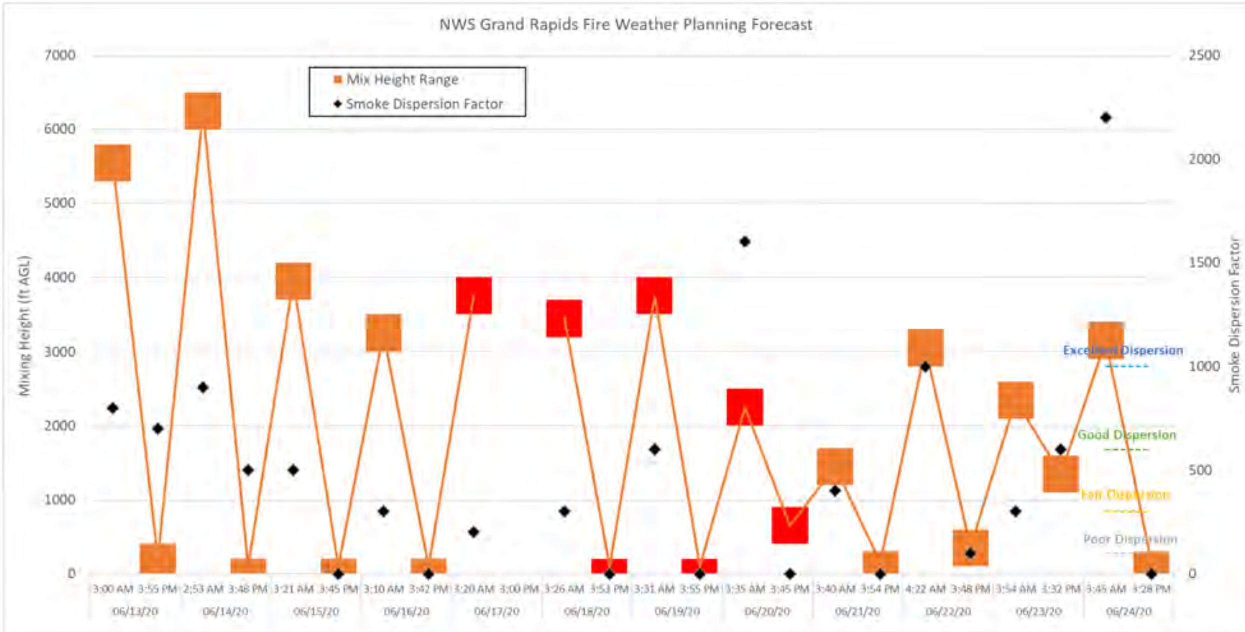

Figure 41. Fire weather planning forecast metrics at Grand Rapids (KGRR) between June 13 and June 24, 2020. Red highlight represents exceptional event ozone exceedance days of June 17-20, 2020. Note: No report was prepared for the afternoon of June 17, 2020.

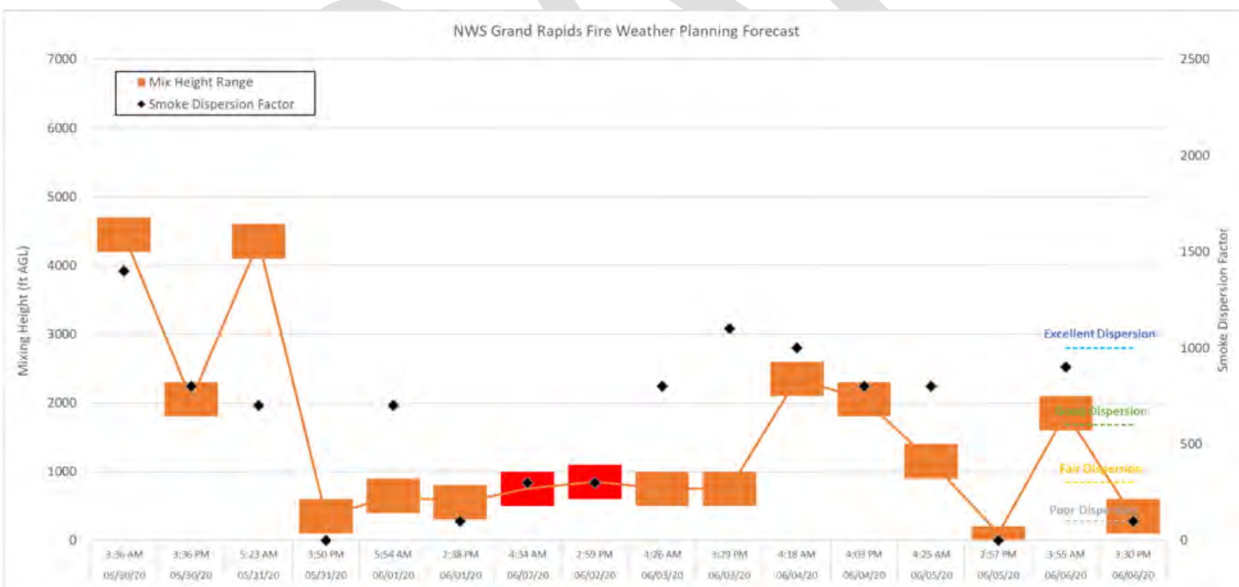

Figure 42. Fire weather planning forecast metrics at Grand Rapids (KGRR) between May 30 and June 5, 2020. Red highlight indicated typical, non-event ozone exceedance on June 2, 2020.

It is important to note that the episode of smoke transport between the Arizona wildfires and the western Michigan monitors occurred during relatively dry conditions. Figure 43 shows the 24-hour precipitation levels[34] (ending at 7:00 AM EST) for June 17 through 20, 2020. The path from Arizona northeast into the region during the key dates of June 17 through 19, 2020, has a clear channel of no precipitation. This provides evidence that ozone precursors and particulate matter (PM) species did not precipitate out during the transport from the wildfires to the western Michigan monitor locations during this period. On June 20, 2020, precipitation is seen on the western shore of Lake Michigan and in Wisconsin, leaving continued dry conditions across Michigan.

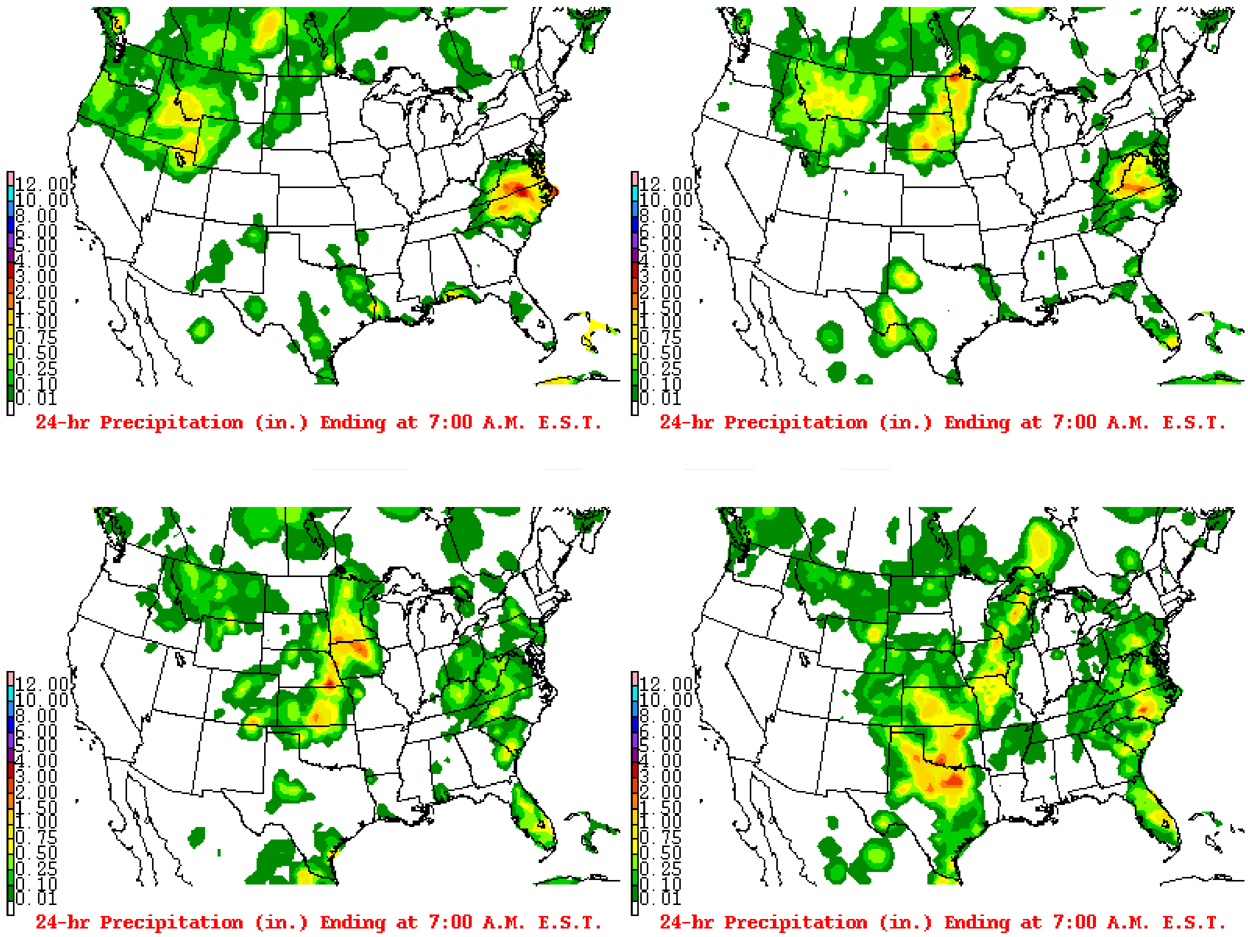

Figure 43. 24-hour Precipitation June 17 (top left), June 18 (top right), June 19 (bottom left), and June 20 (bottom right), 2020

[34] https://www.wpc.ncep.noaa.gov/dailywxmap/