

# **A Refined Satellite-based Emissions Estimate from Onshore Oil and Gas Flaring and Venting Activities in the United States and their Impacts on Air Quality and Health**

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## **Key points**

A new satellite-based emissions inventory for oil and gas flaring and venting is developed to refine existing U.S. emissions inventories.

710 premature deaths dominate the \$7.3B monetized values of health impacts due to this sector.

Significant benefits in air quality and health could be gained by reducing emissions from flaring and venting activities.

## **Abstract**

Emissions from flaring and venting (FV) in oil and gas (O&G) production are difficult to quantify due to their intermittent activities and lack of adequate monitoring and reporting. Given their potentially significant contribution to total emissions from O&G sector in the United States, we estimate emissions from FV using Visible Infrared Imaging Radiometer Suite satellite observations and state/local reported data on flared gas volume. These refined estimates are higher than those reported in the National Emission Inventory: by up to 15 times for fine particulate matter (PM<sub>2.5</sub>), two times for sulfur dioxides, and 22% higher for nitrogen oxides (NO<sub>x</sub>). Annual average contributions of FV to ozone (O<sub>3</sub>), NO<sub>2</sub> and PM<sub>2.5</sub> in the conterminous U.S. (CONUS) are less than 0.15%, but significant contributions of up to 60% are found in O&G fields with FV. FV contributions are higher in winter than in summer months for O<sub>3</sub> and PM<sub>2.5</sub>; an inverse behavior is found for NO<sub>2</sub>. Nitrate aerosol contributions to PM<sub>2.5</sub> are highest in the Denver basin whereas in the Permian and Bakken basins, sulfate and elemental carbon aerosols are the major contributors. Over four simulated months in 2016 for the entire CONUS, FV contributes 210 additional instances of exceedances to the daily maximum 8-hour average O<sub>3</sub>, and has negligible contributions to exceedance of NO<sub>2</sub> and PM<sub>2.5</sub> given the current form of the national ambient air quality standards. FV emissions are found to cause over \$7.4 billion in health damages, 710 premature deaths and 73,000 asthma exacerbations among children annually.

## **Plain language summary**

Pollutant emissions from onshore flaring and venting activities in the oil and gas sector are often hard to capture creating inaccuracies in estimates of air pollution and health impacts from this sector. Here we use remote sensing and reported activity to create a refined estimate of emissions which reveal significant underestimates in official emissions estimates. These emissions contribute to air pollution, which results in \$7.4 billion in health damages annually due to hospitalizations, emergency room visits, worsening asthma and premature death among downwind populations.

## 1. Introduction

Flaring is an oil and gas (O&G) industry term to describe the practice of burning off excess natural gas that is produced along with crude oil, often called ‘associated gas’ or ‘associated petroleum gas’. This associated gas is a valuable commodity when it can be appropriately separated from oil and transported. In practice, however, a facility to support such processing is often absent, and thus flaring is used as a way to dispose of unwanted gas that would otherwise pose a safety hazard or interfere with oil production (DOE, 2019; GGFR, 2023). Multiple economic and technical reasons for why flaring of associated gas is needed are discussed by Soltanieh et al. (2016). According to the World Bank’s Global Gas Flaring Reduction Partnership (GGFR, 2023), global gas flaring stayed relatively constant throughout 2010 – 2020 and reached 150 billion cubic meters (BCM) in 2020, equivalent to the total annual gas consumption of sub-Saharan Africa, and the top five flaring countries are Russia (24 BCM), Iraq (17 BCM), Iran (13 BCM), U.S. (12 BCM), Algeria (9 BCM), and Venezuela (8 BCM). Venting of associated gas from O&G compression or processing equipment due to system upset conditions or pressure release during emergency is also common in O&G production and processing (DOE, 2019).

Besides emitting carbon dioxide (CO<sub>2</sub>), O&G flaring release various pollutants including methane (CH<sub>4</sub>), black carbon (soot), nitrogen oxides (NO<sub>x</sub>), sulfur dioxide (SO<sub>2</sub>), carbon monoxide (CO), various volatile organic compounds (VOCs) depending on flaring conditions and composition of the associated gas (Anejionu et al., 2015; Fawole et al., 2016; Umukoro & Ismail, 2017), all of which can cause various impacts on climate, air quality and human health. According to the World Bank’s Global Gas Flaring Tracker Report, flaring released over 400 million tons of carbon dioxide equivalent (CO<sub>2</sub>e) emissions into the atmosphere in 2020 (World Bank, 2022); such amount is roughly equivalent to the greenhouse gas emissions of around 77 million cars. Allen et al. (2016) estimated that O&G flaring contribute 20 – 21 million metric tons of CO<sub>2</sub>e of greenhouse gases per year in the U.S. Evaluating O&G flaring’s impact on nationally determined contributions (NDC) defined under the United Nations Framework Convention on Climate Change (UNFCCC) Paris Agreement, Elvidge et al. (2018) found global flaring represents less than 2% of the NDC reduction target, however, some countries (e.g., Yemen, Algeria, Iraq) may fully meet their NDC reduction target by just controlling for flaring.

Cushing et al. (2021) reported that flaring can cause negative effects on birth outcomes, such as preterm births, due to exposure to polycyclic aromatic hydrocarbons, and there are more than 500,000 Americans living within 3 miles of natural gas flares potentially exposed and at risk of adverse health effects. O&G production in the U.S. has adverse health impacts of 7,500 premature deaths and 410,000 asthma exacerbations annually (Buonocore et al, 2023). Emissions from flaring cause an increase in respiratory diseases, heart diseases and strokes due to black carbon particle exposure (Chen et al., 2022). Johnston et al. (2020) report flaring can impair lung function and cause asthma due to exposure to volatile organic compounds. Globally, flaring contributed about 0.12% of the health impacts related to PM<sub>2.5</sub> and 6.51% of the health impacts related to climate change (Motte et al., 2021). Motte et al. (2021) also found that health impacts related to gas-phase pollutants from flaring are negligible.

Operators of O&G production facilities which perform flaring and venting (FV) report the volume to local regulatory agencies. However, indicators show that the flared and vented volume

97 reported through this mechanism is underreported (BBC, 2022; DOE, 2019). Methane emissions  
98 from O&G flaring in the U.S. has been found to be more than 5 times higher than what was  
99 expected (Plant et al., 2022). In New Mexico, North Dakota and Texas, the flared gas volume  
100 estimated from satellite observation is as much as double the volume reported to the states (DOE,  
101 2019) during the years 2012 – 2017. In 2019, about 15.2 BCM of total vented and flared gas was  
102 reported over the U.S. (EIA, 2023) while flared gas alone is estimated at 17.3 BCM from satellite  
103 observations (EOG, 2023). Thus, the use of state-reported gas volumes exclusively could lead to  
104 underestimation of emissions from flaring and venting.

105  
106 Although the destruction efficiency (i.e., % of hydrocarbon compounds in flared gas that are  
107 converted to carbon dioxide) of flaring is often assumed to be greater than 95% (Caulton et al.,  
108 2014; Gvakharia et al., 2017; Pohl et al., 1986; Shaw et al., 2022), incomplete combustion and  
109 unlit flares are not uncommon and these issues can lead to lower destruction efficiency of flaring  
110 (Lyon et al., 2021; Plant et al., 2022; Tyner & Johnson, 2021).

111  
112 In this study, we address the potential underestimation of flaring and venting emissions in the  
113 current National Emission Inventory (NEI), by using flared gas volume estimates from the  
114 Visible Infrared Imaging Radiometer Suite (VIIRS) and other industry emission inventories. We  
115 further investigate impacts of flaring and venting from O&G production and processing in the  
116 U.S. on air quality and human health related to both gas-phase and aerosol pollutants using an  
117 integrated assessment framework previously used to evaluate health impacts of oil & gas  
118 production in the U.S. (Buonocore et al., 2023).

## 2. Materials and Methods

### Flare emissions from NEI

We identified 22 source classification codes (SCC) of O&G point sources from the NEI 2017 that have “flare” keyword in the descriptions of SCC (Table S1). Some of these 22 SCCs have zero emissions, however, and the majority of emissions are from the top four SCCs in Table S1. Searching for “flare” keywords in SCCs of O&G nonpoint sources in the NEI 2017 returned only 6 SCCs that represent flaring from well completions, and only two SCC have non-zero emissions. Furthermore, these nonpoint flare-SCCs were introduced in NEI 2017 but not in earlier NEI versions (NEI 2014 and NEI 2016). Through personal communications with technical staff in charge of O&G emissions inventory development in EPA OAQPS, TCEQ, Colorado DHCP, Wyoming DEQ and Utah DEQ, we learned that flare emissions may be associated with 19 nonpoint SCCs as shown in Table S2 and Table S3, though most of them do not have the keyword “flare” in the SCC descriptions.

It is worth highlighting that flare emissions are not separated out but rather lumped into total emissions from these 19 nonpoint-SCCs. Furthermore, not all O&G equipment (e.g., condensate tanks, crude oil tanks) represented by these SCCs are equipped with flares. Therefore, flare emissions from the 19 nonpoint-SCCs vary among counties in the U.S., the spatial resolution at which the nonpoint O&G emissions are allocated in the NEI. To estimate NEI 2017-derived flare emissions for each county in the CONUS, we first applied records with nonzero NO<sub>x</sub> emissions as an indicator for with-flare emissions from each of the 19 nonpoint flare-SCCs, then derived emissions of other criteria pollutants from the same SCC as considered from flare.

Special treatment is applied for VOC emissions from flares: due to the high destruction efficiency of flares (> 95%), VOC emissions coming out from flare stack mounts would be close to zero. (Thus, to our knowledge there is no VOC emissions factor developed for the flaring process). However, flares are not always operating properly which can lead to increased venting of natural gas through the flare stack flaring (DOE, 2019; Lyon et al., 2021; Plant et al., 2022; Reuters, 2022; Tyner & Johnson, 2021). For example, Lyon et al. (2021) reported that 11% of surveyed flares in the Permian Basin had combustion issues and 5% were unlit and emitted uncombusted gas directly into the atmosphere. Furthermore, if flaring was to be eliminated and the would-be gas captured instead through other means of emission controls, it is reasonable to assume the vented gas at the facility could be captured in the same way. Therefore, in this study, we considered VOC emissions from the 19 nonpoint flare-SCCs as from venting and combined this VOC emissions estimate with non-VOC emissions from flare into the same group of “flaring and venting” (FV). Note that “venting” in this way only represents VOC emissions from O&G sources equipped with a flare using the nonzero NO<sub>x</sub> emissions criteria as discussed above.

Figure S1 shows emissions of criteria pollutants from O&G categories including point- and nonpoint-flares as derived from NEI 2017. FV is also found to account for 10% of the total 7.3 million tons per year of methane from O&G although this is not a targeted pollutant for discussion on air quality and health impacts in this study.

### Flare emissions estimation from VIIRS

The Visible Infrared Imaging Radiometer Suite Nightfire dataset (referred hereafter as VIIRS) (EOG, 2023) was processed for annual natural gas flared volume in the U.S. from year 2017 to

2020 (Figure 1). In 2019, the year with highest flare gas volume during the period 2017 – 2020, there were 17.7 BCM of natural gas flared in both production (97%) and processing (3%) – the same amount of flare gas volume was reported in Chen et al. (2022). We performed the following processing on VIIRS data prior to emissions estimation: only onshore flares were considered; VIIRS-detected flares are excluded if they are found to be in close proximity of NEI 2017-point flare sources to avoid double counting. As a result, there were 17.3 BCM of VIIRS-detected flared gas remaining for emissions estimation.

We treat VIIRS-detected flares as point sources, include directly emitted primary particulate matter (in form of black carbon) and SO<sub>2</sub> emissions, utilize empirical algorithms for flare stack parameters, and employ state-level monthly flared and venting gas volume for temporal allocations. Additional methodologic details are described in supplemental information S2.

There is a high probability that not all flares were adequately detected by VIIRS. Certain criteria must be met for a flare to be detected by VIIRS such as flared temperature > 1,400 K, frequent combustion (consistency) and free of cloud cover and other contaminations (C. Elvidge et al., 2013, 2015). In our case study for the Uinta O&G basin in Utah, we compared the number of VIIRS-detected flares with the self-reported flare data from O&G operators (UDEQ, 2022) and found that only 11 (or 8%) of the 132 reported flares were detected by VIIRS in 2019.

Another uncertainty in VIIRS flare data is the flare gas volume (FGV). VIIRS estimates FGV by applying regression algorithms to the relationship between radiated heat energy from detected flare to the reported FGV at state and county levels (C. Elvidge et al., 2013, 2015). Large gaps exist when VIIRS-estimated FGV in 2019 are compared to FGV reported by Rystad Energy (2022) for the same year (Table S4). For example, Rystad Energy (2022) reports FGV in Colorado in 2019 to be 136 MCF/yr whereas only 18 MCF/yr is estimated by VIIRS. Non-detection of flares by VIIRS (more discussion in later) in Colorado are partially attributed to this gap.

To account for the potential under-estimation of FGV and, consequently, the under-estimation of emissions from FV, we developed two emissions scenarios for FV. The *wFlare1* scenario estimates emissions of criteria pollutants solely based on VIIRS estimated FGV. The *wFlare2* combines emissions in *wFlare1* with additional emissions estimated for FGV reported by Rystad Energy (2022) and FV emissions derived from NEI 2017. Specifically: in each county where either VIIRS's, Rystad's or NEI 2017's estimates exist, VIIRS's estimates are first compared against Rystad's. If Rystad's estimates are larger than VIIRS's, differences between the two are added to VIIRS's. If Rystad's estimates are not available or lower than VIIRS's, VIIRS's estimates are then compared against NEI 2017's. Differences between the two estimates are added to VIIRS's if NEI 2017's estimates are higher. In this sense, *wFlare2* represents a hybridized estimate of FV which partially compensates for missing FV data in any one source.

## **CMAQ model configurations and model performance evaluation**

The model configurations in this study refined the configurations applied in Buonocore et al. (2023) which evaluates impacts of O&G emissions to air quality and public health in 2016. As such, all anthropogenic and wildfire emissions other than O&G are based on the NEI 2016 v1. Emissions from all other O&G sources other than FV are taken from the NEI 2017. The use of

NEI 2017 instead of NEI 2016 v1 for O&G sources are due to two factors: introduction of new flare-SCCs as discussed above; and NEI 2017 is the latest national baseline estimate from the EPA and furthermore, it better represents year 2019 in which FGV is highest and for which FV emissions are estimated for.

The Models-3 / Community Multiscale Air Quality (CMAQ) modeling system (Byun & Schere, 2006; Wyatt Appel et al., 2018) version 5.2.1 was utilized to simulate atmospheric chemistry with Carbon-Bond version 6 revision 3 (CB6r3) gaseous chemistry and aerosol6 for aerosol treatment. Meteorological inputs are derived from the Weather Research and Forecasting model (Skamarock et al., 2008) version 4.7. WRF-CMAQ simulations were performed for January, April, July and October 2016 (to represent four seasons) for a modeling domain covering the conterminous U.S. (CONUS) in 12 km x 12 km horizontal grid resolution and 35 vertical layers (12US1 domain). Boundary and initial chemistry conditions were taken from the hemispheric CMAQ (HCMAQ) version 5.2.1 simulation for northern hemisphere. Evaluation of CMAQ model performance for key pollutants of interest are briefly discussed in Supporting Information S3.

Two model scenarios included FV emissions as estimated in *wFlare1* and *wFlare2* (discussed above) and all other non-FV emissions from all other anthropogenic and natural sources in the domain. FV emissions are excluded in a zero-out scenario (*woFlare*). Simulation results of *wFlare1* and *wFlare2* are compared against *woFlare*, alternatively, to quantify the impact of FV emissions on air quality and human health. For brevity, however, discussions on the impacts in the following sections are based on *wFlare2* scenario unless specified otherwise.

## Analyses

Modeled exceedance counts are determined for each of *woFlare* and *wFlare2* scenarios and then differences used to determine marginal impact of flaring and venting emissions on the National Ambient Air Quality Standard (NAAQS) threshold(s). A modeled exceedance event is identified when concentration in any grid-cell for any pollutant exceeded its corresponding NAAQS for the relevant timescale: e.g., Maximum Daily 8-hour average Ozone (MDA8O3) at any grid-cell for any day exceeded 70 ppbV. For this study, the model domain is 459 (columns) x 299 (rows) grid-cells and there are 123 simulation days in total. Thus there are up to 459 x 299 x 123 possibilities for MDA8O3 or Daily Average PM<sub>2.5</sub> exceedances to occur. Note that a high number of exceedances does not necessarily lead to violation of NAAQS.

## Method for health impact analyses

To calculate the health impacts of flaring, we subtracted the health impacts of air pollution under the Baseline/No-flaring scenario from those of the *wFlare2* scenario to isolate the health impacts from the flare portion of *wFlare2* scenario emissions. To estimate the health impacts of ambient ground level concentrations of PM<sub>2.5</sub>, NO<sub>2</sub>, and ozone under each scenario, we used BenMAPR which is a geospatial health impact assessment model in R and based on the Benefits Mapping and Analysis Program (BenMAP) from the U.S. EPA (Sacks et al., 2018), and used in two recent studies (Arter et al. 2022; Buonocore et al. 2023). BenMAPR accepts air pollution concentration outputs from CMAQ, and overlays it with population data from the U.S. American Community Survey from the U.S. Census Bureau; data on background rates of health outcomes from the U.S. Centers for Disease Control, Health Care Utilization Project, and BenMAP from the U.S. EPA;

259 and concentration response functions relating air pollution exposure and changes in risks for  
260 health outcomes from the epidemiological literature. These outcomes were then valued using  
261 valuation methods from the U.S. EPA and existing health literature. Details of the background  
262 health data sets, concentration response functions, and valuation functions are available in  
263 Supplemental Tables S9 through S11. Methods for additional health impact analyses, including  
264 environmental justice are presented in supplemental information S3.



### 3. Results and Discussion

#### Flare and Venting Emissions

The inclusion of PM<sub>2.5</sub> and SO<sub>2</sub> to FV emissions resulted in significantly higher emissions of the two pollutants in both *wFlare1* and *wFlare2* than in NEI 2017 over the entire CONUS (Table 1). Compared to NEI 2017, FV PM<sub>2.5</sub> emissions are 13 times and 15 times higher in *wFlare1* and *wFlare2*, respectively. FV SO<sub>2</sub> emissions are more than two times higher in *wFlare1* and *wFlare2* than in NEI 2017. As discussed in S.2, O&G SO<sub>2</sub> emissions are highly underestimated in the NEI 2017 and our FV SO<sub>2</sub> emissions are likely closer to actual emissions. Since VIIRS did not detect flares in some counties where FV emissions are reported in NEI 2017, VOC and NO<sub>x</sub> emissions are lower in *wFlare1* than in NEI 2017. In *wFlare2* where excess emissions from Rystad and NEI 2017's estimates are considered, FV NO<sub>x</sub> emissions are 22% higher than NEI 2017 and VOC emissions match the NEI 2017's estimates. NH<sub>3</sub> emissions are identical among all three estimates as NH<sub>3</sub> emissions are only accounted as point flares in the NEI 2017 and no NH<sub>3</sub> emissions were accounted for VIIRS-detected flares.

Large gaps exist between VIIRS-detected and NEI-derived flares. Figure 1 shows NO<sub>x</sub> and PM<sub>2.5</sub> emissions as estimated using VIIRS flared gas volume (FGV) and as derived from point O&G flare categories from NEI 2017. The majority of VIIRS-detected flares are over O&G production fields including Permian in New Mexico (NM) and Texas (TX), Eagle Ford in TX, Bakken/Williston in North Dakota (ND); whereas much less VIIRS-detected flares exist over other major O&G production fields such as the Barnett, Denver basin or Appalachian in Pennsylvania (PA) (Figure 1). On the one hand, the NEI-derived flares are reported in more O&G production fields. Over Denver basin in Colorado (CO), for example, most flares are from NEI-derived data and not detected by VIIRS. On the contrary, over the Bakken basin, the largest flare FVs are VIIRS-detected and very few are from NEI-derived data.

In accordance with the distribution of VIIRS-detected flares, emissions from FV are most noticeable in major O&G production fields in the U.S., especially those in NM, TX, ND, CO and WY (Figure 2). Emissions differences between *wFlare2* and *wFlare1* (i.e., FV emissions accounted for in the NEI 2017 but not in VIIRS) are shown in these states but also noticeable in Oklahoma, Kansas, Pennsylvania, Ohio and West Virginia (Figure S6). In Pennsylvania (Appalachian basin), between *wFlare2* and *wFlare1* there is a distinctly high FV NO<sub>x</sub> emissions hotspot which comes from a single flare-SCC (2310021500) for flaring from an onshore gas well completion. This distinctly high NO<sub>x</sub> emission is attributed as an artifact in NEI 2017, and is treated "as is" in this study. As PM<sub>2.5</sub> emissions were only estimated for VIIRS-detected flares, there are no PM<sub>2.5</sub> emissions differences between *wFlare2* and *wFlare1*. PM<sub>2.5</sub> emissions from FV account for about 82% of total O&G PM<sub>2.5</sub> emissions (Table S6). In this study, FV SO<sub>2</sub> emissions account for 82% of total O&G SO<sub>2</sub> emissions, and this high percentage is attributed to underestimation of O&G SO<sub>2</sub> emissions in the NEI 2017 (see SI.2). VOC emissions from FV account for about 50% of total O&G VOC emissions over CONUS mainly due to the inclusion of storage tank's venting. The benefit of treating FV VOC emissions in this study, however, provides an opportunity to quantify air quality and health benefits from potentially controlling both flaring and venting together.

This study estimates BC (analogously to primary PM<sub>2.5</sub>) emissions in CONUS from FV to be 4,340 ± 729 tpy with Texas (2,244 ± 250 tpy) and North Dakota (1,713 ± 216 tpy) as the top two emitting states. Chen et al. (2022) estimated black carbon (BC) emission from upstream flaring in the CONUS to be 15,986 tpy, with North Dakota (10,036 tpy) and Texas (4,317) as the two leading states. A different set of emission factors and heating values applied in Chen et al. (2022) gives reason for their higher estimations of PM<sub>2.5</sub> emissions than in this study. Although the VIIRS-detected flare gas volume is higher in Texas (8.73 BCM) than in North Dakota (6.09 BCM), Chen et al. (2022) estimated higher black carbon in North Dakota because of the higher heating value applied to this region. Based on in-situ measurements in 2013 – 2014, Schwarz et al. (2015) estimate BC emissions in Bakken basin to be 1,400 ± 360 tpy, which is relatively closer to our estimates. Only about 0.36 Gg per year (or 397 tpy) of BC emissions from flaring in Bakken basin were estimated by Weyant et al. (2016).

*Table 1. Annual emissions (tpy) of criteria pollutants from flaring and venting (FV)*

	PM <sub>2.5</sub>	NH <sub>3</sub>	VOC	CO	NO <sub>x</sub>	SO <sub>2</sub>
<i>NEI 2017 point flare</i>	7.82E+01	1.84E-01	4.59E+03	8.44E+03	2.71E+03	1.53E+04
<i>NEI 2017 nonpoint flare</i>	2.22E+02	0	1.31E+06	3.94E+04	1.87E+04	3.37E+04
<i>Total NEI 2017 flare</i>	3.00E+02	1.84E-01	1.31E+06	4.78E+04	2.14E+04	4.90E+04
<i>VIIRS-only flare</i>	4.34E+03 ± 729	0	1.04E+06	6.79E+04 ± 3,066	1.42E+04 ± 987	8.98E+04 ±15,166
<i>VIIRS + Rystad + NEI hybrid</i>	4.82E+03	0	1.31E+06	7.75E+04	2.33E+04	1.00E+05
<i>Total wFlare1*</i>	4.42E+03 (1,373 %)	1.84E-01 (0 %)	1.04E+06 (-20 %)	7.63E+04 (60 %)	1.69E+04 (-21 %)	1.05E+05 (114 %)
<i>Total wFlare2*</i>	4.90E+03 (1,533 %)	1.84E-01 (0 %)	1.31E+06 (00 %)	8.59E+04 (80 %)	2.60E+04 (22 %)	1.15E+05 (135 %)

\* Numbers in parentheses indicate changes in emissions from NEI 2017: e.g., 100\*(wFlare2 – NEI 2017)/NEI 2017

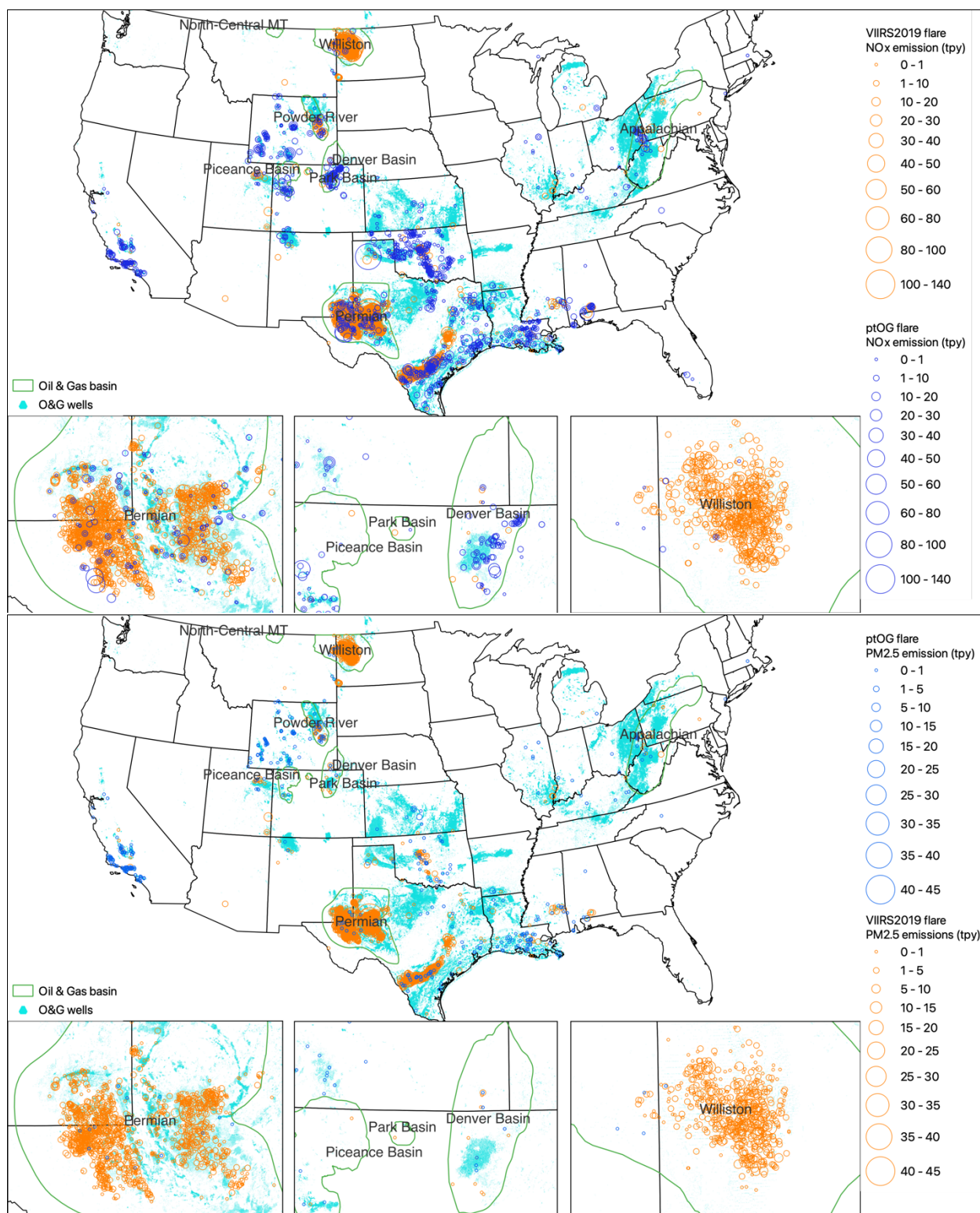


Figure 1. Combined (top) NO<sub>x</sub> and (bottom) PM<sub>2.5</sub> emissions from FV as estimated from VIIRS and as derived from NEI 2017 point O&G (ptOG flare)

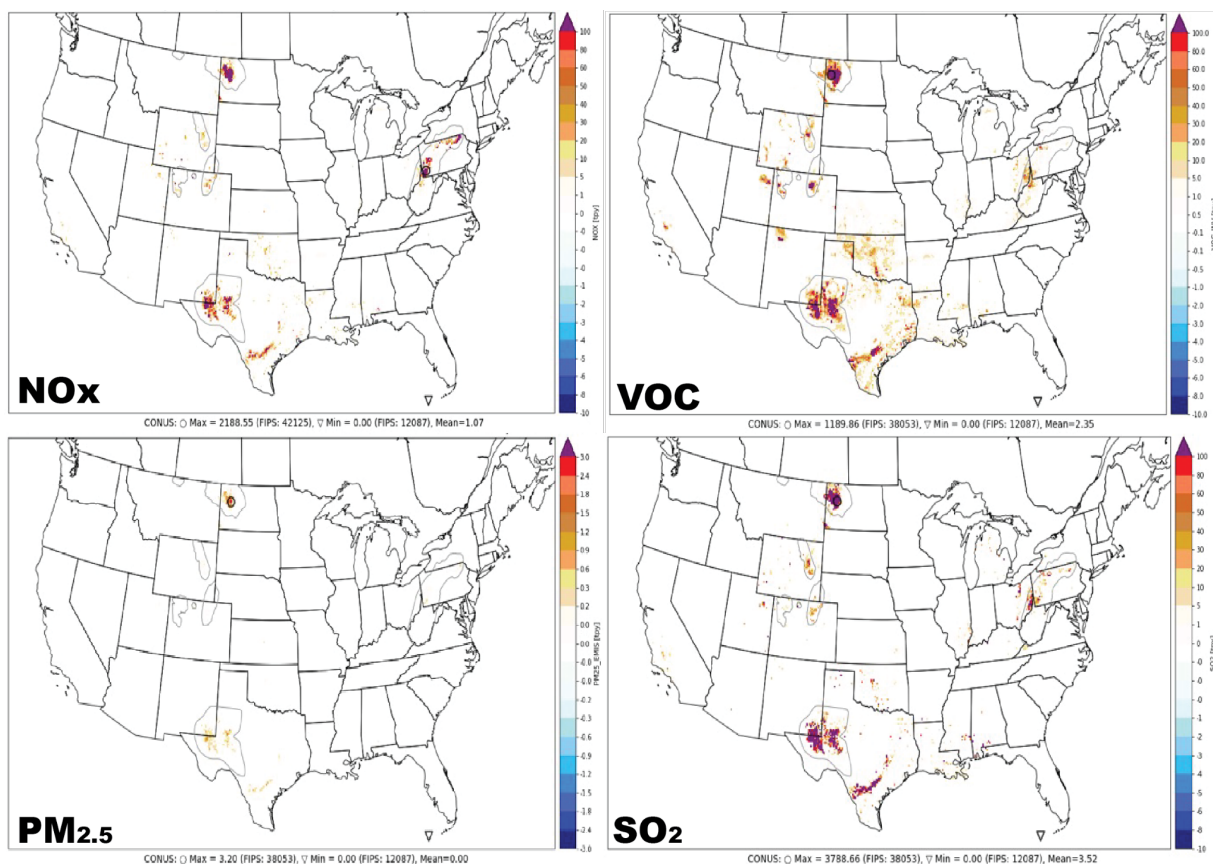


Figure 2. Annual emissions (tpy)  $\text{NO}_x$ , VOC,  $\text{PM}_{2.5}$  and  $\text{SO}_2$  from flaring and venting (FV)

### Impact of Flaring and Venting on Air Quality

Domain-wide annual average impacts of FV on  $\text{O}_3$ ,  $\text{NO}_2$  and  $\text{PM}_{2.5}$  concentrations in the CONUS are relatively small (<0.15%). However, the impacts greatly vary with locations and seasons (Figure S9 through S12) with the strongest impacts typically occurring in areas with intense FV emissions (Figure 3), emphasizing the advantage of the detailed treatment of the emissions and processes at the modeled grid resolution of 12 km x 12 km over the entire country.

### Impacts on MDA8O<sub>3</sub>

FV's impact on  $\text{O}_3$  is stronger in January than in other months (Figure S9). FV is found to contribute 4 – 47% (4 – 16 ppb) and 2 – 15% (2 – 10 ppb) of MDA8O<sub>3</sub> in January and July, respectively, over FV-major areas (e.g., Permian, Denver, Bakken). Among O&G production fields with emissions from FV (referred hereafter as FV-major areas), the Denver basin observed the highest FV's impact on MDA8O<sub>3</sub> (up to 16 ppb or 47%, occurred in January) followed by Bakken (10 ppb, 19%, July) and Permian (7 ppb, 11%, October) as the second and third highest impacted basins. The impact on MDA8O<sub>3</sub> is also noticeable in areas within 100 km of FV-major areas (4 – 6 ppb or 9 – 30%) but reduces to less than 1 ppb (5%) elsewhere. We found high  $\text{NO}_x$  emissions in the Appalachian basin enhanced  $\text{O}_3$  formation for the area in July (up to 3 ppb) but suppressed  $\text{O}_3$  formation by as much as -3.5 ppb in other months (Figure S9). Differences in FV's impact on  $\text{O}_3$  between *wFlare2* and *wFlare1* scenarios (Figure S8) closely follow the



differences in NO<sub>x</sub> and VOC emissions between the two scenarios (Figure S6) and is less than 2 ppb for impacts on MDA8O<sub>3</sub>.

### Impacts on NO<sub>2</sub>

In addition to heterogeneous distribution in space and time, FV's impacts on NO<sub>2</sub> have a distinguished distribution: increases of NO<sub>2</sub> are mostly localized to areas with FV emissions, and decreases of NO<sub>2</sub> are observed mostly elsewhere (Figure 4, Figure S10). The response of NO<sub>2</sub> concentration to FV's emissions highly depends on local background chemistry. During the daytime, NO<sub>2</sub> and NO interconvert through photolysis reactions and through reactions with O<sub>3</sub>, organic (RO<sub>x</sub>) and hydrogen oxide radicals (HO<sub>x</sub>). NO<sub>x</sub> is terminated by forming nitric acid (HNO<sub>3</sub>) and dinitrogen pentoxide (N<sub>2</sub>O<sub>5</sub>, nighttime only), in which both are precursors of nitrate aerosols. In FV-major areas, NO-to-NO<sub>2</sub> conversation is enhanced by VOC emissions which results in net increase of NO<sub>2</sub>, thus serving to quench the HO<sub>x</sub>-NO<sub>x</sub> cycle and shorten the OH chain (Womack et al., 2019). Such NO<sub>2</sub>-reduction effects are most noticeable downwind of the Denver basin.

Since the partition of nitrate to aerosol is favorable under low temperature conditions (Ansari & Pandis, 1998; Park, 2004), loss of NO<sub>2</sub> through this pathway is higher in January than other months and is lowest in July (Figure S10). Monthly average hourly NO<sub>2</sub> decreases by 0.1 ppb in January in the downwind areas of Denver and Uinta basins, and parts of these areas observe an increase by 0.5 ppb in monthly average hourly NO<sub>2</sub> in July. Appalachian basin observes the highest contribution of hourly (up to 8 ppb, or 46% of total NO<sub>2</sub>) and daily-maximum (up to 25 ppb, 56%) NO<sub>2</sub>, both in July, from FV, followed by Denver, Permian and Bakken basins where observed contributions in 2 – 6 ppb (8 -30%) and 2 – 20 ppb (18 – 36%) to hourly and daily-maximum NO<sub>2</sub> are seen, respectively, with highest contributions often seen in January or July. Elsewhere in the CONUS, other than FV-majors and their downwind areas, changes in monthly average of NO<sub>2</sub> are negligible ( $\pm 0.02$ ppb,  $\pm 0.4\%$ ) in both hourly and daily maximum (Figure 4).

### Impacts on PM<sub>2.5</sub>

FV's contribution to annual PM<sub>2.5</sub> is less than 0.01  $\mu\text{g}/\text{m}^3$  (or 0.2% of total PM<sub>2.5</sub>) on average over CONUS), 0.1  $\mu\text{g}/\text{m}^3$  (3%) over Permian, 0.1  $\mu\text{g}/\text{m}^3$  (4%) over Bakken, 0.01  $\mu\text{g}/\text{m}^3$  (0.2%) over Appalachian, 0.02  $\mu\text{g}/\text{m}^3$  (0.9%) over Powder River, and 0.1  $\mu\text{g}/\text{m}^3$  (1.2%) over Denver basins (Figure 4). The Denver basin observed the highest FV's contribution by as much as 5  $\mu\text{g}/\text{m}^3$  to daily average PM<sub>2.5</sub>, whereas highest contributions in other FV-major areas are upto 1.5  $\mu\text{g}/\text{m}^3$ . Negative PM<sub>2.5</sub> contributions (reductions due to FV emissions) occurred in Minnesota, Iowa, and other states in the northeast of CONUS in January, but at a relatively small margin (up to 0.06  $\mu\text{g}/\text{m}^3$  on monthly average). Positive monthly average PM<sub>2.5</sub> contribution from FV is greatest in January (up to 1.4  $\mu\text{g}/\text{m}^3$ , 10%) and lowest in October (0.5  $\mu\text{g}/\text{m}^3$ , 16%) (Figure S11). On average over CONUS, however, FV's contribution to monthly-average of PM<sub>2.5</sub> is larger in July (0.018  $\mu\text{g}/\text{m}^3$ ; 0.5%) than in January (0.012  $\mu\text{g}/\text{m}^3$ ; 0.3%) due to negative PM<sub>2.5</sub> contributions (or PM<sub>2.5</sub> reductions) mostly occurring in January (Figure S12).

FV's contribution to PM<sub>2.5</sub> varies with PM<sub>2.5</sub> compositions which differ significantly among FV-major as well as non-FV areas (Figure S12). In areas outside FV-major areas, sulfate aerosol (SO<sub>4</sub><sup>-</sup>) is the largest component (48%), followed by elemental carbon (EC; 18%), NH<sub>4</sub><sup>+</sup> (15%), organic carbon (OC, 10%) and NO<sub>3</sub><sup>-</sup> (4%). In the Bakken basin, the major PM<sub>2.5</sub> component is

EC (61%), followed by  $\text{NO}_3^-$  (22%),  $\text{SO}_4^-$  (14%) and less than 3% of other components. A similar distribution is observed in the Permian basin where 50% of  $\text{PM}_{2.5}$  is EC followed by  $\text{NO}_3^-$  (29%),  $\text{SO}_4^-$  (12%), OC (7%) and  $\text{NH}_4^+$  (2%). Differing significantly from other FV-major areas,  $\text{PM}_{2.5}$  in the Denver basin has  $\text{NO}_3^-$  as a major component (42%), followed by EC (28%),  $\text{NH}_4^+$  (13%), OC (12%) and  $\text{SO}_4^-$  (2%). In relevant to earlier discussions on  $\text{NO}_2$  formation in FV-major areas, FV's contributions  $\text{NO}_3^-$  and  $\text{NH}_4^+$  aerosols are largest in Denver basin. Meanwhile, FV's contributions to EC and  $\text{SO}_4^-$  are largest in Bakken basin. Reductions of  $\text{NO}_3^-$  and OC aerosol led to reductions of total  $\text{PM}_{2.5}$  in Minnesota, Iowa, and other states in northeast of CONUS (Figure S12). Since there is negligible difference in primary  $\text{PM}_{2.5}$  emissions from FV between *wFlare2* and *wFlare1*, the differences in  $\text{PM}_{2.5}$  contributions between the two scenarios ( $< 0.164 \mu\text{g}/\text{m}^3$  on annual average) are caused by secondary aerosols which are dominated by their inorganic components, i.e.,  $\text{NO}_3^-$  and  $\text{SO}_4^-$  (Figure S8).

We found  $\text{PM}_{2.5}$  contributions from FV are mainly driven by its contribution of  $\text{SO}_2$  and primary  $\text{PM}_{2.5}$  (mostly EC) emissions across CONUS especially in FV-major areas. Whereas in Denver basin and its downwind areas, we found increases in  $\text{NO}_3^-$  due to FV emissions enhancing the formation of nitric acid which favors the formation of  $\text{NO}_3^-$ .

#### Impacts on exceedance counts

Overall, FV emissions caused over 210 instances of MDA8O3 exceedances ( $\text{MDA8O3} > 70$  ppb) over 4 simulated months in 2016. This is about one-third of MDA8O3 exceedances caused by the O&G sector in 2016 reported by Buonocore et al. (2023). In this study, MDA8O3 exceedances are largest in counties of the Denver basin and its downwind area, followed by counties in Permian basin and those in Pennsylvania and Michigan (Table S7). FV contributes to no MDA8O3 exceedances in January and most of its exceedance contribution occurred only in the summer month of July.

Contributions of FV to daily  $\text{PM}_{2.5}$  exceedances ( $\text{PM}_{2.5} > 35 \mu\text{g}/\text{m}^3$ ) is small. While FV added two additional  $\text{PM}_{2.5}$  exceedances in Pennsylvania and New Jersey, it also reduced two exceedances in Minnesota and New York due to a reduction of  $\text{PM}_{2.5}$  in these two states in the winter. For comparison, Buonocore et al. (2023) found 29 instances of  $\text{PM}_{2.5}$  daily exceedances caused by O&G sector in 2016. If  $\text{PM}_{2.5}$  daily NAAQS were lowered to  $30 \mu\text{g}/\text{m}^3$ , FV emissions would contribute 10 instances of exceedances over 4 simulated months. No additional annual  $\text{PM}_{2.5}$  exceedances (annual average  $\text{PM}_{2.5} > 12 \mu\text{g}/\text{m}^3$ ) were found. FV would have contributed 3 and 1 additional instances of annual exceedances in Pennsylvania and Illinois, respectively, if the annual  $\text{PM}_{2.5}$  standard were lowered to  $10 \mu\text{g}/\text{m}^3$  (EPA, 2023).

FV causes no additional  $\text{NO}_2$  exceedances ( $\text{NO}_2 > 100$  ppb) which is expected given that Buonocore et al. (2023) found no  $\text{NO}_2$  exceedances caused by total O&G in 2016. However, if the 1-hour  $\text{NO}_2$  standard were lowered to 60 ppb then it would add 9 instances of exceedances in Colorado and 1 instance in Florida.

Buonocore et al. (2023) estimated that in 2016 the O&G sector contributed 0.6 ppb of  $\text{O}_3$ , 0.17 ppb of  $\text{NO}_2$  and  $0.065 \mu\text{g}/\text{m}^3$  of  $\text{PM}_{2.5}$  on average over CONUS. This study found the corresponding contributions from FV are 0.026 ppb, 0.03 ppb and  $0.008 \mu\text{g}/\text{m}^3$ , respectively. Since FV emissions do not exist in all areas with O&G activities, such relatively small FV's

contribution in comparison to total O&G when taking average over CONUS is anticipated. However, the highest FV's contributions to MDA8O3 (15 ppb), daily maximum NO<sub>2</sub> (25 ppb) and daily average PM<sub>2.5</sub> (5 ug/m<sup>3</sup>) are much higher than the values found for total O&G by Buonocore et al. (2023) (3 ppb, 17 ppb, and 1.7 ug/m<sup>3</sup>, respectively), emphasizing the near-field impacts of FV that one should focus on. Note that this study combined O&G emissions from NEI 2017 and FV emissions calculated based on VIIRS-Rystad-NEI hybrid dataset, whereas Buonocore et al. (2023) utilized O&G emissions from NEI 2016 as is. Regardless, this finding illustrated that estimation of impacts on O&G sectors on air quality, and consequently human health, could greatly vary with input emissions, and improvements in the emissions estimates as we have done provide increased confidence in the modeled estimates.

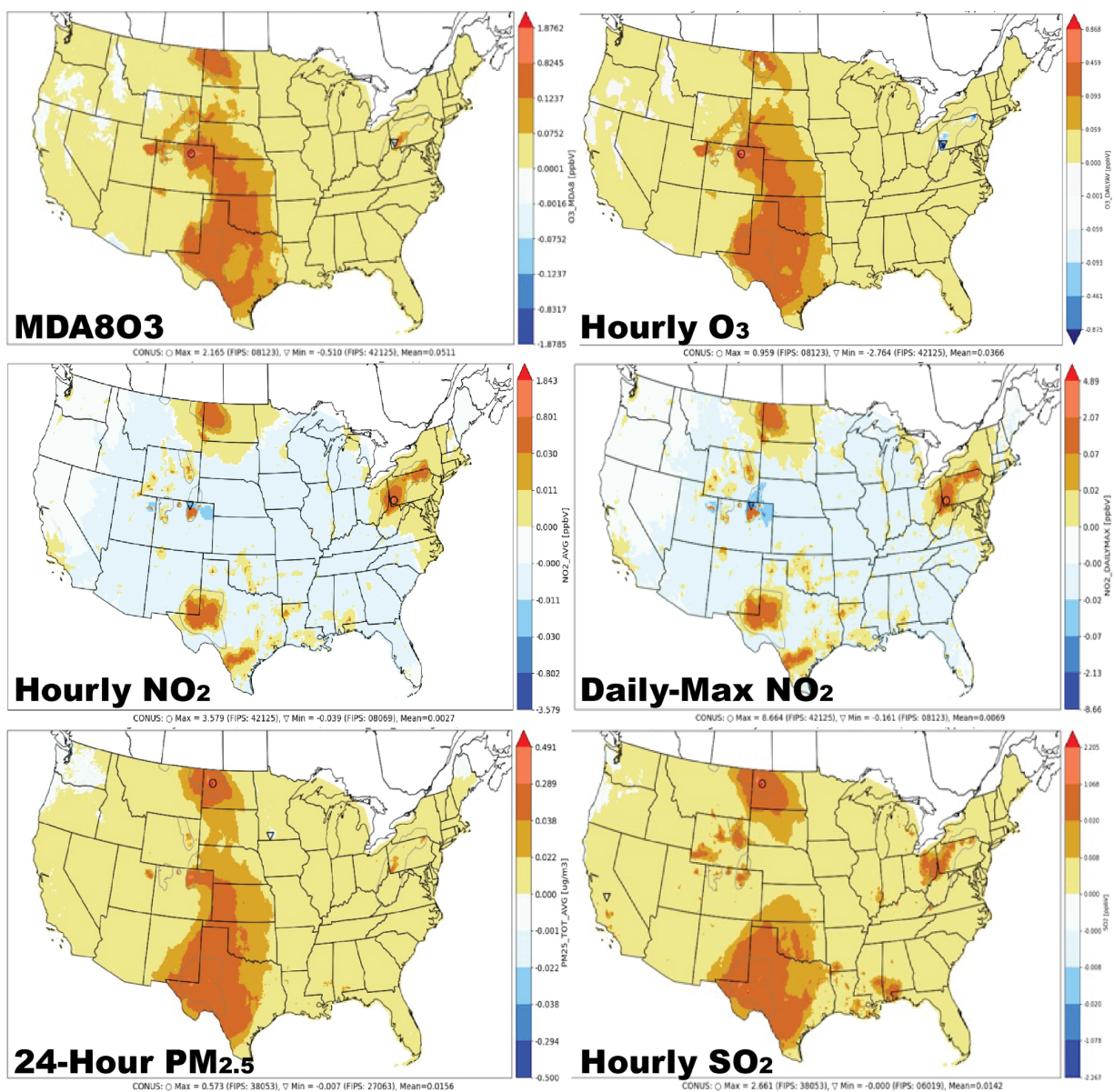


Figure 3. Annual-average of MDA8 Ozone (ppb), 24-hour average PM<sub>2.5</sub> (ug/m<sup>3</sup>), daily-average and daily-maximum NO<sub>2</sub> (ppb), SO<sub>2</sub> (ppb) contributed by FV (i.e., differences between wFlare2 and woFlare)



## Health Impacts of Flaring and Venting

Our results show in 2016, emissions due to flaring in Flare Scenario 2 have a mortality burden of 710 (95% CI: 480 – 1,100) excess deaths attributable to PM<sub>2.5</sub>, NO<sub>2</sub>, and ozone compared to baseline scenario emissions. Additionally, our results show an estimated annual excess of 73,000 (95% CI: 46,000 – 110,000) childhood asthma exacerbations, 92 (95% CI: 58 – 140) childhood asthma emergency department visits, and 10 (95% CI: 6.4 – 15) asthma hospitalizations attributable to PM<sub>2.5</sub>, NO<sub>2</sub>, and ozone. An excess of 190 (95% CI: 66 – 300) childhood asthma incidence and 130 (95% CI: 50 – 120) respiratory hospitalizations were also found, for combinations of PM<sub>2.5</sub> and NO<sub>2</sub>, and PM<sub>2.5</sub> and ozone, respectively.

A recent paper (Buonocore et al., 2023) using a similar framework and data inputs showed that the health burden of oil & gas as a whole is 7,500 (95% CI: 4,500 – 12,000) deaths, 410,000 (95% CI: 9,200 – 810,000) childhood asthma exacerbations, and 2,200 (95% CI: 830 – 3,200) childhood asthma incidence. Comparing these two studies indicates that flaring and venting contributes just under 10% of the mortality cases and incident asthma cases from O&G production as a whole, and around 5.4% of the asthma exacerbations from O&G production. FV contributes 2% of NO<sub>x</sub> emissions, 81% of SO<sub>2</sub>, 51% of VOCs, and 18% of PM<sub>2.5</sub> from oil & gas. The relative proportions that NO<sub>2</sub> from flaring contributes to total sector deaths and asthma exacerbations (Table S11) indicates the strong role of NO<sub>2</sub> in driving total health impacts.

As a whole, asthma outcomes calculated using all Alhanti CRFs (for NO<sub>2</sub>, PM<sub>2.5</sub> and ozone) are approximately three times larger than those calculated using all of the Orellano CRFs (NO<sub>2</sub> and PM<sub>2.5</sub>). The increase in cases is predominantly due to the inclusion of ozone, which accounted for nearly 60% of all Alhanti asthma outcomes. Estimates due to PM<sub>2.5</sub> and NO<sub>2</sub> were consistently higher for Alhanti than for Orellano. All three asthma related outcomes estimated by Alhanti were three times larger than those estimated by Orellano (Table S11).

Figure 2 presents spatial distribution of FV air pollution-attributable deaths in 2016. Texas observed the highest FV-attributed premature deaths at 133 incidences, of which 76, 51 and 6 incidences are caused by PM<sub>2.5</sub>, O<sub>3</sub> and NO<sub>2</sub>, respectively. The second (115) and third (76) highest premature deaths are observed in Pennsylvania and Colorado, respectively. The top three numbers of FV-attributed asthma incidences by state are also observed in Texas (14,935), Colorado (13,748) and Pennsylvania (11,184).

Although being an FV-major area, only 6 premature deaths and 464 asthma incidences are observed in North Dakota, which is explained by the transport of these emissions to downwind locations. Between 20 – 30 FV-attributed premature deaths and 2,000 – 3,000 asthma incidences are observed in Ohio, New York, Oklahoma and Illinois.

Of the 710 deaths attributable to *wFlare2*, 1 in 3 occurred in low-income census tracts, 1 in 10 occurred in tracts identified as 65<sup>th</sup> percentile or higher for Native persons (i.e., greater than or equal to 2% Native populations), and 1 in 3 occurred in tracts identified as 65<sup>th</sup> percentile or higher for Hispanic/Latino-identified persons (i.e., greater than or equal to 14% Hispanic/Latino populations) (Table S12). Similar proportions of impact were seen for childhood asthma exacerbations among low-income and Native-identified tracts, and a slightly larger proportion of impact was seen among Hispanic/Latino-identified tracts (40%). Results from the impact risk

ratio found that Native-identified tracts had 1.2 times the risk of premature mortality and 1.1 times the risk of childhood asthma exacerbations from *wFlare2* exposure than non-Native-identified tracts (Table S13). Hispanic/Latino-identified tracts had 1.1 times the risk of childhood asthma exacerbations than non-Hispanic/Latino-identified tracts. Pollutant exposure from *wFlare2* was not found to disproportionately increase the risk of premature death among census tracts identified as low income or Hispanic/Latino, or to disproportionately increase the risk of childhood asthma among low-income tracts.

Table 2 shows that the monetized values of the health impacts due to FV total \$7.4B, while Buonocore et al (2023) reported \$77B from the entire OG sector, indicating a rather significant (~10%) contribution of FV to the overall monetized health risk from the OG sector. Industry analysis (Rystad 2022) indicates that solutions for operators to address flaring and capture this otherwise wasted gas are readily available and cost-effective, and even potentially profitable. The results of this study reveal the near-term air quality health benefits from addressing flaring and venting emissions. Since ~90% of the health impacts of O&G production originate from outside flaring, and since NO<sub>2</sub> has such a strong role, this indicates that health benefits of emission control strategies can be increased by expanding coverage to more NO<sub>x</sub>-rich subsectors of the O&G production sector such as compressors and pumpjack engines, well drilling and completions, in addition to the solutions for reducing FV emissions.

## Limitations

The analysis year for this study is 2016/2017 based on availability of various input datasets. The flaring volume applied in this study was estimated for 2019 which is the highest estimate from VIIRS throughout 2017 - 2022 period (EOG, 2023). So, our estimates for emission from flaring in this study could be lower in most recent years. On the other hand, the emissions inventories used here may still be underestimated due to missing flares invisible due to cloud cover and flares too small to detect via satellite. There is known to be missing compressor engines in the NEI, and it is possible that other sources are similarly underreported. While the 12km x 12km grid used here is the finest grid readily available, it still may not be fine enough to detect high air pollution hotspots near sources, especially those that may coincide with areas that have high rates of background disease, and thus have potentially higher impacts at community scale.

While the health impact modeling uses the best background health data available, some outcomes only have data at state or national resolution – missing potential hotspots for diseases, most notably asthma. Additionally, since the health modeling framework exclusively captures health impacts due to exposure to the three pollutants ozone, PM<sub>2.5</sub>, and NO<sub>2</sub>, this model is unlikely to fully capture many of the complex, multifactorial impacts occurring in communities hosting O&G production. These health outcomes include but are not limited to adverse birth outcomes (Willis et al., 2022), asthma (Willis et al., 2018), and childhood leukemia (Clark et al., 2022).

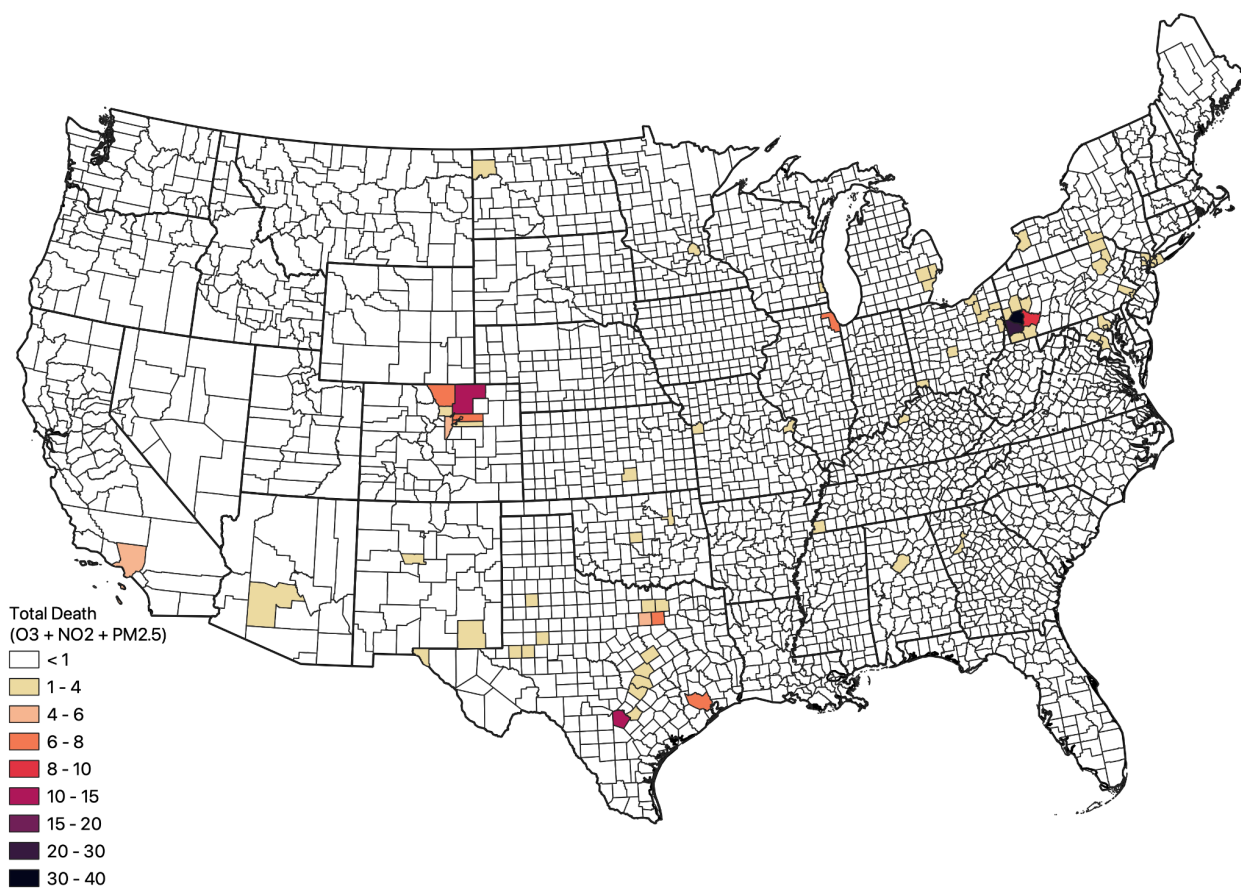
The environmental justice analysis that we conducted was not comprehensive of all Justice40 indicators that were available. Future research could consider including other populations at risk of environmental exposures as defined by CJEST (Council on Environmental Quality, 2022). Furthermore, the U.S. Census Bureau American Community Survey (ACA) data underreports

the true number of Native people living in the U.S. (Smithsonian, 2023), and the undercounting and misclassification of Native peoples in health databases (Jim et al., 2014; Stehr-Green et al., 2002) along with existing cultural, structural, and social barriers to health care (USCCR, 2004) contribute to the true representation of Native persons and their health status (Adakai et al., 2018). Having access to the true number of Native peoples and fairly represented mortality and asthma outcomes could impact our IRR calculations.

Despite these limitations, this study still produces a robust estimate of health impacts of flaring by using the best data available for emissions from flaring. This study also uses a novel health impact assessment framework that includes health impacts of direct NO<sub>2</sub> exposure, and health outcomes not regularly included in air pollution health impact assessments, most notably birth and children's health outcomes.

562 Table 2. Monetary Valuation of Health Impacts

Health Impact	Flaring Health Impact Outcomes, based on Pollutant Type		Monetary Valuation of Health Impacts
	Pollutant Type	Flare 2 Cases (95% CI)	\$ Flare 2 (95% CI)
Premature Deaths	PM <sub>2.5</sub>	360 (300 - 420)	\$3,700,000,000 (\$1,900,000,000 - \$6,000,000,000)
	O <sub>3</sub>	230 (120 - 470)	\$2,400,000,000 (\$720,000,000 - \$6,800,000,000)
	NO <sub>2</sub>	120 (61 - 180)	\$1,300,000,000 (\$380,000,000 - \$2,700,000,000)
	All Three	710 (480 - 1,100)	\$7,300,000,000 (\$3,000,000,000 - \$16,000,000,000)
Asthma Incidence	PM <sub>2.5</sub>	140 (47 - 230)	\$8,200,000 (\$1,100,000 - \$22,000,000)
	NO <sub>2</sub>	47 (19 - 65)	\$2,800,000 (\$460,000 - \$6,100,000)
	PM <sub>2.5</sub> and NO <sub>2</sub>	190 (66 - 300)	\$11,000,000 (\$1,600,000 - \$28,000,000)
Asthma Hospitalizations (Alhanti)	PM <sub>2.5</sub>	1.3 (0.63 - 2.5)	\$23,000 (\$12,000 - \$46,000)
	O <sub>3</sub>	5.7 (3.2 - 8.1)	\$100,000 (\$59,000 - \$150,000)
	NO <sub>2</sub>	3.2 (2.5 - 4.5)	\$58,000 (\$46,000 - \$81,000)
	All Three	10 (6.4 - 15)	\$180,000 (\$120,000 - \$280,000)
Asthma Exacerbations (Alhanti)	PM <sub>2.5</sub>	9,700 (4,900 - 19,000)	\$580,000 (\$110,000 - \$1,900,000)
	O <sub>3</sub>	43,000 (24,000 - 61,000)	\$2,500,000 (\$530,000 - \$6,000,000)
	NO <sub>2</sub>	21,000 (16,000 - 29,000)	\$1,200,000 (\$360,000 - \$2,800,000)
	All Three	73,000 (46,000 - 110,000)	\$4,300,000 (\$990,000 - \$11,000,000)
Asthma ED Visits (Alhanti)	PM <sub>2.5</sub>	13 (6.3 - 25)	\$5,700 (\$2,700 - \$12,000)
	O <sub>3</sub>	54 (31 - 77)	\$24,000 (\$13,000 - \$36,000)
	NO <sub>2</sub>	25 (20 - 36)	\$11,000 (\$8,800 - \$17,000)
	All Three	92 (58 - 140)	\$42,000 (\$25,000 - \$65,000)
Respiratory Hospitalizations	PM <sub>2.5</sub>	19 (9.8 - 28)	\$570,000 (\$290,000 - \$840,000)
	O <sub>3</sub>	110 (40 - 180)	\$3,400,000 (\$1,200,000 - \$5,500,000)
	PM <sub>2.5</sub> and Ozone	130 (50 - 210)	\$3,900,000 (\$1,500,000 - \$6,400,000)
Heart Attacks	PM <sub>2.5</sub>	16 (9.7 - 23)	\$1,100,000 (\$680,000 - \$1,600,000)
	NO <sub>2</sub>	6.9 (3.7 - 10)	\$480,000 (\$260,000 - \$700,000)
	PM <sub>2.5</sub> and NO <sub>2</sub>	23 (13 - 33)	\$1,600,000 (\$940,000 - \$2,300,000)
Sum Total			\$7,400,000,000 (\$3,000,000,000 - \$16,000,000,000)



564

565 *Figure 4. FV air pollution-attributable deaths in 2016*

## Conclusions

Combining satellite-based observation of flare activities in O&G activities and new set of emission factors resulted in higher emissions from FV in the U.S. than what were reported in the EPA's NEI. Impacts of FV on air quality are most noticeable and significant within FV-major areas and its immediate downwind regions, even though domain-wide averages are relatively small compared to the overall OG impacts. However, FV sector still contributes up to about \$7.4B (~10%) of the total burden of health risk from O&G, and thus highlighting the potential need to focus on them for protecting public health. Of the total 710 premature deaths estimated from F&V, 360 are attributed to PM<sub>2.5</sub>, 230 to O<sub>3</sub> and 120 to NO<sub>2</sub>. This finding signifies that while most health impact studies so far have been focused on PM<sub>2.5</sub>, health impact from O<sub>3</sub> & NO<sub>2</sub> should not be overlooked. Findings from this study suggest controlling emissions from flaring and venting from O&G production, besides being cost effective and profitable to the operators, additionally provides an opportunity for yielding significant public health benefits.

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## Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

## Open research (availability statement)

Estimates of flared gas volume from the Visible Infrared Imaging Radiometer Suite - VIIRS for 2019 and other years are publicly available at

[https://eogdata.mines.edu/products/vnf/global\\_gas\\_flare.html#data\\_download](https://eogdata.mines.edu/products/vnf/global_gas_flare.html#data_download)

National emissions inventories (NEI) data for 2016 and 2017 were downloaded from U.S. EPA Emission Modeling Platforms, publicly available at <https://www.epa.gov/air-emissions-modeling/emissions-modeling-platforms>, and processed using the Sparse Matrix Operator Kernel Emissions – SMOKE version 4.8, publicly available at <https://doi.org/10.5281/zenodo.4088945>

Flare stack parameters were estimated with reanalysis II wind data from NOAA Physical Sciences Laboratory, publicly available at

<https://psl.noaa.gov/data/gridded/data.ncep.reanalysis2.html>

Air quality simulations were performed using the Community Multiscale Air Quality – CMAQ model version 5.2.1, publicly available at <https://doi.org/10.5281/zenodo.1212601>

Source code of the BenMAPR is publicly available at <https://doi.org/10.5281/zenodo.8306380>

Baseline health and economic data were extracted from U.S. EPA BenMAP model which is publicly available at <https://www.epa.gov/benmap/benmap-downloads>

Mortality and population counts for individual counties for the entire U.S. for adults  $\geq 25$  years and infants  $<1$  year old were obtained from Centers for Disease Control and Prevention Wide-ranging ONline Data for Epidemiologic Research – CDC WONDER, available through request at <https://wonder.cdc.gov/cmfi-icd10.html>; user's agreement to data use restrictions is required.

Environmental justice analysis was performed with data from the Climate and Economic Justice Screening Tool – CJEST, publicly available at <https://screeningtool.geoplatform.gov/en/#3/33.47/-97.5>

## References

- Adakai, M., Sandoval-Rosario, M., Xu, F., Aseret-Manygoats, T., Allison, M., Greenlund, K. J., & Barbour, K. E. (2018). Health Disparities Among American Indians/Alaska Natives - Arizona, 2017. *MMWR. Morbidity and Mortality Weekly Report*, 67(47), 1314–1318. <https://doi.org/10.15585/mmwr.mm6747a4>
- Allen, D. T., Smith, D., Torres, V. M., & Saldaña, F. C. (2016). Carbon dioxide, methane and black carbon emissions from upstream oil and gas flaring in the United States. *Current Opinion in Chemical Engineering*, 13, 119–123. <https://doi.org/10.1016/j.coche.2016.08.014>
- Anejionu, O. C. D., Whyatt, J. D., Blackburn, G. A., & Price, C. S. (2015). Contributions of gas flaring to a global air pollution hotspot: Spatial and temporal variations, impacts and alleviation. *Atmospheric Environment*, 118, 184–193. <https://doi.org/10.1016/j.atmosenv.2015.08.006>
- Alhanti, B. A., Chang, H. H., Winkvist, A., Mulholland, J. A., Darrow, L. A., & Sarnat, S. E. (2016). Ambient air pollution and emergency department visits for asthma: a multi-city assessment of effect modification by age. *Journal of Exposure Science & Environmental Epidemiology*, 26(2), 180–188. <https://doi.org/10.1038/jes.2015.57>
- Ansari, A. S., & Pandis, S. N. (1998). Response of inorganic PM to precursor concentrations. *Environmental Science & Technology*, 32(18), 2706–2714. <https://doi.org/10.1021/es971130j>
- BBC. (2022, September 29). Revealed: Huge gas flaring emissions never reported. *BBC News*.



638 Becerra, T. A., Wilhelm, M., Olsen, J., Cockburn, M., & Ritz, B. (2013). Ambient air pollution  
639 and autism in Los Angeles county, California. *Environmental Health Perspectives*, 121(3),  
640 380–386. <https://doi.org/10.1289/ehp.1205827>

641 Buonocore, J. J., Reka, S., Yang, D., Chang, C., Roy, A., Thompson, T., Lyon, D., McVay, R.,  
642 Michanowicz, D., & Arunachalam, S. (2023). Air pollution and health impacts of oil & gas  
643 production in the United States. *Environmental Research: Health*, 1(2), 021006.  
644 <https://doi.org/10.1088/2752-5309/acc886>

645 Byun, D., & Schere, K. L. (2006). Review of the Governing Equations, Computational  
646 Algorithms, and Other Components of the Models-3 Community Multiscale Air Quality  
647 (CMAQ) Modeling System. *Applied Mechanics Reviews*, 59(2), 51.  
648 <https://doi.org/10.1115/1.2128636>

649 Caulton, D. R., Shepson, P. B., Cambaliza, M. O. L., McCabe, D., Baum, E., & Stirm, B. H.  
650 (2014). Methane destruction efficiency of natural gas flares associated with shale formation  
651 wells. *Environmental Science & Technology*, 48(16), 9548–9554.  
652 <https://doi.org/10.1021/es500511w>

653 Chen, C., McCabe, D. C., Fleischman, L. E., & Cohan, D. S. (2022). Black carbon emissions and  
654 associated health impacts of gas flaring in the united states. *Atmosphere*, 13(3), 385.  
655 <https://doi.org/10.3390/atmos13030385>

656 Clark, C. J., Johnson, N. P., Soriano, M., Warren, J. L., Sorrentino, K. M., Kadan-Lottick, N. S.,  
657 Saiers, J. E., Ma, X., & Deziel, N. C. (2022). Unconventional Oil and Gas Development  
658 Exposure and Risk of Childhood Acute Lymphoblastic Leukemia: A Case-Control Study in

659 Pennsylvania, 2009-2017. *Environmental Health Perspectives*, 130(8), 87001.  
660 <https://doi.org/10.1289/EHP11092>

661 Council on Environmental Quality. (2022, November 22). *Climate and Economic Justice Screen*  
662 *Tool (CJEST)*. <https://screeningtool.geoplatform.gov/en/downloads>

663 Cushing, L. J., Chau, K., Franklin, M., & Johnston, J. E. (2021). Up in smoke: characterizing the  
664 population exposed to flaring from unconventional oil and gas development in the contiguous  
665 US. *Environmental Research Letters*, 16(3), 034032. [https://doi.org/10.1088/1748-](https://doi.org/10.1088/1748-9326/abd3d4)  
666 [9326/abd3d4](https://doi.org/10.1088/1748-9326/abd3d4)

667 DOE. (2019). *Natural Gas Flaring and Venting: State and Federal Regulatory Overview, Trends,*  
668 *and Impacts*.

669 EIA. (2023, April 28). *Natural Gas Vented and Flared Summary*. U.S. Energy Information  
670 Administration (EIA).  
671 [https://www.eia.gov/dnav/ng/NG\\_SUM\\_LSUM\\_A\\_EPG0\\_VGV\\_MMCF\\_A.htm](https://www.eia.gov/dnav/ng/NG_SUM_LSUM_A_EPG0_VGV_MMCF_A.htm)

672 Elvidge, C., Zhizhin, M., Baugh, K., Hsu, F.-C., & Ghosh, T. (2015). Methods for Global Survey  
673 of Natural Gas Flaring from Visible Infrared Imaging Radiometer Suite Data. *Energies*, 9(1),  
674 14. <https://doi.org/10.3390/en9010014>

675 Elvidge, C., Zhizhin, M., Hsu, F.-C., & Baugh, K. (2013). VIIRS nightfire: satellite pyrometry at  
676 night. *Remote Sensing*, 5(9), 4423–4449. <https://doi.org/10.3390/rs5094423>

677 Elvidge, C. D., Bazilian, M. D., Zhizhin, M., Ghosh, T., Baugh, K., & Hsu, F.-C. (2018). The  
678 potential role of natural gas flaring in meeting greenhouse gas mitigation targets. *Energy*  
679 *Strategy Reviews*, 20, 156–162. <https://doi.org/10.1016/j.esr.2017.12.012>

680 EOG. (2023). *Global Gas Flaring Observation from Space*. Earth Observation Group (EOG).  
681 [https://eogdata.mines.edu/products/vnf/global\\_gas\\_flare.html](https://eogdata.mines.edu/products/vnf/global_gas_flare.html)

682 EPA. (2023). *EPA Proposes to Strengthen Air Quality Standards to Protect the Public from*  
683 *Harmful Effects of Soot | US EPA*. [https://www.epa.gov/newsreleases/epa-proposes-](https://www.epa.gov/newsreleases/epa-proposes-strengthen-air-quality-standards-protect-public-harmful-effects-soot)  
684 [strengthen-air-quality-standards-protect-public-harmful-effects-soot](https://www.epa.gov/newsreleases/epa-proposes-strengthen-air-quality-standards-protect-public-harmful-effects-soot)

685 Faustini, A., Rapp, R., & Forastiere, F. (2014). Nitrogen dioxide and mortality: review and meta-  
686 analysis of long-term studies. *The European Respiratory Journal*, 44(3), 744–753.  
687 <https://doi.org/10.1183/09031936.00114713>

688 Fawole, O. G., Cai, X. M., & MacKenzie, A. R. (2016). Gas flaring and resultant air pollution: A  
689 review focusing on black carbon. *Environmental Pollution*, 216, 182–197.  
690 <https://doi.org/10.1016/j.envpol.2016.05.075>

691 GGFR. (2023, February). *Global Gas Flaring Reduction Partnership (GGFR)*. World Bank.  
692 <https://www.worldbank.org/en/programs/gasflaringreduction/gas-flaring-explained>

693 Gvakharia, A., Kort, E. A., Brandt, A., Peischl, J., Ryerson, T. B., Schwarz, J. P., Smith, M. L.,  
694 & Sweeney, C. (2017). Methane, Black Carbon, and Ethane Emissions from Natural Gas  
695 Flares in the Bakken Shale, North Dakota. *Environmental Science & Technology*, 51(9),  
696 5317–5325. <https://doi.org/10.1021/acs.est.6b05183>

697 Institute of Medicine. (2007). *Preterm Birth: Causes, Consequences, and Prevention* (R. E.  
698 Behrman & A. S. Butler, Eds.). National Academies Press (US).  
699 <https://doi.org/10.17226/11622>

700 Ji, M., Cohan, D. S., & Bell, M. L. (2011). Meta-analysis of the Association between Short-Term  
701 Exposure to Ambient Ozone and Respiratory Hospital Admissions. *Environmental Research*  
702 *Letters : ERL [Web Site]*, 6(2). <https://doi.org/10.1088/1748-9326/6/2/024006>

703 Jim, M. A., Arias, E., Seneca, D. S., Hoopes, M. J., Jim, C. C., Johnson, N. J., & Wiggins, C. L.  
704 (2014). Racial misclassification of American Indians and Alaska Natives by Indian Health  
705 Service Contract Health Service Delivery Area. *American Journal of Public Health*, 104 Suppl  
706 3, S295-302. <https://doi.org/10.2105/AJPH.2014.301933>

707 Johnston, J. E., Chau, K., Franklin, M., & Cushing, L. (2020). Environmental Justice Dimensions  
708 of Oil and Gas Flaring in South Texas: Disproportionate Exposure among Hispanic  
709 communities. *Environmental Science & Technology*, 54(10), 6289–6298.  
710 <https://doi.org/10.1021/acs.est.0c00410>

711 Kerr-Wilson, C. O., Mackay, D. F., Smith, G. C. S., & Pell, J. P. (2012). Meta-analysis of the  
712 association between preterm delivery and intelligence. *Journal of Public Health*, 34(2), 209–  
713 216. <https://doi.org/10.1093/pubmed/fdr024>

714 Khreis, H., Kelly, C., Tate, J., Parslow, R., Lucas, K., & Nieuwenhuijsen, M. (2017). Exposure  
715 to traffic-related air pollution and risk of development of childhood asthma: A systematic  
716 review and meta-analysis. *Environment International*, 100, 1–31.  
717 <https://doi.org/10.1016/j.envint.2016.11.012>

718 Levy, J. I., Diez, D., Dou, Y., Barr, C. D., & Dominici, F. (2012). A meta-analysis and multisite  
 719 time-series analysis of the differential toxicity of major fine particulate matter constituents.  
 720 *American Journal of Epidemiology*, 175(11), 1091–1099. <https://doi.org/10.1093/aje/kwr457>

721 Lyon, D. R., Hmiel, B., Gautam, R., Omara, M., Roberts, K. A., Barkley, Z. R., Davis, K. J.,  
 722 Miles, N. L., Monteiro, V. C., Richardson, S. J., Conley, S., Smith, M. L., Jacob, D. J., Shen,  
 723 L., Varon, D. J., Deng, A., Rudelis, X., Sharma, N., Story, K. T., ... Hamburg, S. P. (2021).  
 724 Concurrent variation in oil and gas methane emissions and oil price during the COVID-19  
 725 pandemic. *Atmospheric Chemistry and Physics*, 21(9), 6605–6626.  
 726 <https://doi.org/10.5194/acp-21-6605-2021>

727 Motte, J., Alvarenga, R. A. F., Thybaut, J. W., & Dewulf, J. (2021). Quantification of the global  
 728 and regional impacts of gas flaring on human health via spatial differentiation. *Environmental*  
 729 *Pollution*, 291, 118213. <https://doi.org/10.1016/j.envpol.2021.118213>

730 Mustafic, H., Jabre, P., Caussin, C., Murad, M. H., Escolano, S., Tafflet, M., Périer, M.-C.,  
 731 Marijon, E., Vernerey, D., Empana, J.-P., & Jouven, X. (2012). Main air pollutants and  
 732 myocardial infarction: a systematic review and meta-analysis. *The Journal of the American*  
 733 *Medical Association*, 307(7), 713–721. <https://doi.org/10.1001/jama.2012.126>

734 Nurmagambetov, T., Kuwahara, R., & Garbe, P. (2018). The Economic Burden of Asthma in the  
 735 United States, 2008-2013. *Annals of the American Thoracic Society*, 15(3), 348–356.  
 736 <https://doi.org/10.1513/AnnalsATS.201703-259OC>

737 Orellano, P., Quaranta, N., Reynoso, J., Balbi, B., & Vasquez, J. (2017). Effect of outdoor air  
738 pollution on asthma exacerbations in children and adults: Systematic review and multilevel  
739 meta-analysis. *Plos One*, 12(3), e0174050. <https://doi.org/10.1371/journal.pone.0174050>

740 Park, R. J. (2004). Natural and transboundary pollution influences on sulfate-nitrate-ammonium  
741 aerosols in the United States: Implications for policy. *Journal of Geophysical Research*,  
742 109(D15). <https://doi.org/10.1029/2003JD004473>

743 Plant, G., Kort, E. A., Brandt, A. R., Chen, Y., Fordice, G., Gorchov Negron, A. M., Schwietzke,  
744 S., Smith, M., & Zavala-Araiza, D. (2022). Inefficient and unlit natural gas flares both emit  
745 large quantities of methane. *Science*, 377(6614), 1566–1571.  
746 <https://doi.org/10.1126/science.abq0385>

747 Pohl, J. H., Tichenor, B. A., Lee, J., & Payne, R. (1986). Combustion efficiency of flares.  
748 *Combustion Science and Technology*, 50(4–6), 217–231.  
749 <https://doi.org/10.1080/00102208608923934>

750 Reuters. (2022, February). *Never dark for families under the red glow of Mexico's gas flares*.  
751 Reuters. <https://www.reuters.com/investigates/special-report/mexico-pemex-flaring/>

752 Russell, R. B., Green, N. S., Steiner, C. A., Meikle, S., Howse, J. L., Poschman, K., Dias, T.,  
753 Potetz, L., Davidoff, M. J., Damus, K., & Petrini, J. R. (2007). Cost of hospitalization for  
754 preterm and low birth weight infants in the United States. *Pediatrics*, 120(1), e1-9.  
755 <https://doi.org/10.1542/peds.2006-2386>

756 Rystad. (2022). *Cost of Flaring Abatement - Final Report*.

757 Sacks, J. D., Lloyd, J. M., Zhu, Y., Anderton, J., Jang, C. J., Hubbell, B., & Fann, N. (2018). The  
 758 Environmental Benefits Mapping and Analysis Program – Community Edition (BenMAP–  
 759 CE): A tool to estimate the health and economic benefits of reducing air pollution.  
 760 *Environmental Modelling & Software*, 104, 118–129.  
 761 <https://doi.org/10.1016/j.envsoft.2018.02.009>

762 Schwarz, J. P., Holloway, J. S., Katich, J. M., McKeen, S., Kort, E. A., Smith, M. L., Ryerson, T.  
 763 B., Sweeney, C., & Peischl, J. (2015). Black Carbon Emissions from the Bakken Oil and Gas  
 764 Development Region. *Environmental Science & Technology Letters*, 2(10), 281–285.  
 765 <https://doi.org/10.1021/acs.estlett.5b00225>

766 Shaw, J. T., Foulds, A., Wilde, S., Barker, P., Squires, F., Lee, J., Purvis, R., Burton, R.,  
 767 Colfescu, I., Mobbs, S., Bauguitte, S. J.-B., Young, S., Schwietzke, S., & Allen, G. (2022).  
 768 *Flaring efficiencies and NO<sub>x</sub> emission ratios measured for offshore oil and gas facilities in the*  
 769 *North Sea*. <https://doi.org/10.5194/acp-2022-679>

770 Skamarock, W., Klemp, J., Dudhia, J., Gill, D., Barker, D., Wang, W., Huang, X.-Y., & Duda,  
 771 M. (2008). A Description of the Advanced Research WRF Version 3. *UCAR/NCAR*.  
 772 <https://doi.org/10.5065/d68s4mvh>

773 Smithsonian. (2023, July). *Frequently Asked Questions | Native Knowledge 360° - Interactive*  
 774 *Teaching Resources*. <https://americanindian.si.edu/nk360/faq/did-you-know#>

775 Soltanieh, M., Zohrabian, A., Gholipour, M. J., & Kalnay, E. (2016). A review of global gas  
 776 flaring and venting and impact on the environment: Case study of Iran. *International Journal*  
 777 *of Greenhouse Gas Control*, 49, 488–509. <https://doi.org/10.1016/j.ijggc.2016.02.010>

778 Stehr-Green, P., Bettles, J., & Robertson, L. D. (2002). Effect of racial/ethnic misclassification  
779 of American Indians and Alaskan Natives on Washington State death certificates, 1989-1997.  
780 *American Journal of Public Health*, 92(3), 443–444. <https://doi.org/10.2105/ajph.92.3.443>

781 Sun, X., Luo, X., Zhao, C., Chung Ng, R. W., Lim, C. E. D., Zhang, B., & Liu, T. (2015). The  
782 association between fine particulate matter exposure during pregnancy and preterm birth: a  
783 meta-analysis. *BMC Pregnancy and Childbirth*, 15, 300. [https://doi.org/10.1186/s12884-015-](https://doi.org/10.1186/s12884-015-0738-2)  
784 0738-2

785 Sun, X., Luo, X., Zhao, C., Zhang, B., Tao, J., Yang, Z., Ma, W., & Liu, T. (2016). The  
786 associations between birth weight and exposure to fine particulate matter (PM<sub>2.5</sub>) and its  
787 chemical constituents during pregnancy: A meta-analysis. *Environmental Pollution*, 211, 38–  
788 47. <https://doi.org/10.1016/j.envpol.2015.12.022>

789 Turner, M. C., Jerrett, M., Pope, C. A., Krewski, D., Gapstur, S. M., Diver, W. R., Beckerman,  
790 B. S., Marshall, J. D., Su, J., Crouse, D. L., & Burnett, R. T. (2016). Long-Term Ozone  
791 Exposure and Mortality in a Large Prospective Study. *American Journal of Respiratory and*  
792 *Critical Care Medicine*, 193(10), 1134–1142. <https://doi.org/10.1164/rccm.201508-1633OC>

793 Tyner, D. R., & Johnson, M. R. (2021). Where the Methane Is-Insights from Novel Airborne  
794 LiDAR Measurements Combined with Ground Survey Data. *Environmental Science &*  
795 *Technology*, 55(14), 9773–9783. <https://doi.org/10.1021/acs.est.1c01572>

796 UDEQ. (2022, August). *2020 Statewide Oil & Gas Emissions Inventory*. Utah Department of  
797 Environmental Quality (UDEQ). [https://deq.utah.gov/air-quality/statewide-oil-gas-emissions-](https://deq.utah.gov/air-quality/statewide-oil-gas-emissions-inventory)  
798 inventory



799 Umukoro, G. E., & Ismail, O. S. (2017). Modelling emissions from natural gas flaring. *Journal*  
800 *of King Saud University - Engineering Sciences*, 29(2), 178–182.  
801 <https://doi.org/10.1016/j.jksues.2015.08.001>

802 USCCR. (2004). *Broken Promises: Evaluating the Native American Health Care System*. U.S.  
803 Commission on Civil Rights (USCCR). <https://www.usccr.gov/files/pubs/docs/nabroken.pdf>

804 Vodonos, A., Awad, Y. A., & Schwartz, J. (2018). The concentration-response between long-  
805 term PM2.5 exposure and mortality; A meta-regression approach. *Environmental Research*,  
806 166, 677–689. <https://doi.org/10.1016/j.envres.2018.06.021>

807 Weyant, C. L., Shepson, P. B., Subramanian, R., Cambaliza, M. O. L., Heimbürger, A., McCabe,  
808 D., Baum, E., Stirr, B. H., & Bond, T. C. (2016). Black Carbon Emissions from Associated  
809 Natural Gas Flaring. *Environmental Science & Technology*, 50(4), 2075–2081.  
810 <https://doi.org/10.1021/acs.est.5b04712>

811 Willis, M. D., Hill, E. L., Kile, M. L., Carozza, S., & Hystad, P. (2022). Associations between  
812 residential proximity to oil and gas extraction and hypertensive conditions during pregnancy: a  
813 difference-in-differences analysis in Texas, 1996-2009. *International Journal of*  
814 *Epidemiology*, 51(2), 525–536. <https://doi.org/10.1093/ije/dyab246>

815 Willis, M. D., Jusko, T. A., Halterman, J. S., & Hill, E. L. (2018). Unconventional natural gas  
816 development and pediatric asthma hospitalizations in Pennsylvania. *Environmental Research*,  
817 166, 402–408. <https://doi.org/10.1016/j.envres.2018.06.022>

818 Winer, R. A., Qin, X., Harrington, T., Moorman, J., & Zahran, H. (2012). Asthma incidence  
819 among children and adults: findings from the Behavioral Risk Factor Surveillance system  
820 asthma call-back survey--United States, 2006-2008. *The Journal of Asthma*, 49(1), 16–22.  
821 <https://doi.org/10.3109/02770903.2011.637594>

822 Womack, C. C., McDuffie, E. E., Edwards, P. M., Bares, R., Gouw, J. A., Docherty, K. S., Dubé,  
823 W. P., Fibiger, D. L., Franchin, A., Gilman, J. B., Goldberger, L., Lee, B. H., Lin, J. C., Long,  
824 R., Middlebrook, A. M., Millet, D. B., Moravek, A., Murphy, J. G., Quinn, P. K., ... Brown,  
825 S. S. (2019). An odd oxygen framework for wintertime ammonium nitrate aerosol pollution in  
826 urban areas: no<sub>x</sub> and VOC control as mitigation strategies. *Geophysical Research Letters*.  
827 <https://doi.org/10.1029/2019GL082028>

828 World Bank. (2022, May 5). *2022 Global Gas Flaring Tracker Report*. World Bank.  
829 [https://www.worldbank.org/en/topic/extractiveindustries/publication/2022-global-gas-flaring-](https://www.worldbank.org/en/topic/extractiveindustries/publication/2022-global-gas-flaring-tracker-report)  
830 [tracker-report](https://www.worldbank.org/en/topic/extractiveindustries/publication/2022-global-gas-flaring-tracker-report)

831 Wyat Appel, K., Napelenok, S., Hogrefe, C., Pouliot, G., Foley, K. M., Roselle, S. J., Pleim, J.  
832 E., Bash, J., Pye, H. O. T., Heath, N., Murphy, B., & Mathur, R. (2018). Overview and  
833 evaluation of the community multiscale air quality (CMAQ) modeling system version 5.2. In  
834 C. Mensink & G. Kallos (Eds.), *Air pollution modeling and its application XXV* (pp. 69–73).  
835 Springer International Publishing. [https://doi.org/10.1007/978-3-319-57645-9\\_11](https://doi.org/10.1007/978-3-319-57645-9_11)

836 Zanobetti, A., Franklin, M., Koutrakis, P., & Schwartz, J. (2009). Fine particulate air pollution  
837 and its components in association with cause-specific emergency admissions. *Environmental*  
838 *Health: A Global Access Science Source*, 8, 58. <https://doi.org/10.1186/1476-069X-8-58>