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

Huy Tran[1], Erin Polka[2], Jonathan J. Buonocore[2], Ananya Roy[3], Beth Trask[3], Hillary Hull[3], Saravanan Arunachalam[1*]

[1] *Institute for the Environment, University of North Carolina at Chapel Hill*
[2] *Department of Environmental Health, Boston University School of Public Health*
[3] *Environmental Defense Fund*

* Corresponding Author: Saravanan Arunachalam (sarav@email.unc.edu)

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 upto 15 times for fine particulate matter ($PM_{2.5}$), two times for sulfur dioxides, and 22% higher for nitrogen oxides(NOx). Annual average contributions of FV to ozone (O_3), NO_2 and $PM_{2.5}$ in the conterminous U.S. (CONUS) are less than 0.15%, but significant contributions of upto 60% are found in O&G fields with FV. FV contributions are higher in winter than in summer months for O_3 and $PM_{2.5}$; an inverse behavior is found for NO_2. Nitrate aerosol contributions to $PM_{2.5}$ 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_3, and has negligible contributions to exceedance of NO_2 and $PM_{2.5}$ 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_2), O&G flaring release various pollutants including methane (CH_4), black carbon (soot), nitrogen oxides (NOx), sulfur dioxide (SO_2), 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_2e) 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 if CO_2e 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 $PM_{2.5}$ 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 reported through this mechanism is underreported (BBC, 2022; DOE, 2019). Methane emissions from O&G flaring in the U.S. has been found to be more than 5 times higher than what was expected (Plant et al., 2022). In New Mexico, North Dakota and Texas, the flared gas volume estimated from satellite observation is as much as double the volume reported to the states (DOE, 2019) during the years 2012 – 2017. In 2019, about 15.2 BCM of total vented and flared gas was reported over the U.S. (EIA, 2023) while flared gas alone is estimated at 17.3 BCM from satellite observations (EOG, 2023). Thus, the use of state-reported gas volumes exclusively could lead to underestimation of emissions from flaring and venting.

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

In this study, we address the potential underestimation of flaring and venting emissions in the current National Emission Inventory (NEI), by using flared gas volume estimates from the Visible Infrared Imaging Radiometer Suite (VIIRS) and other industry emission inventories. We further investigate impacts of flaring and venting from O&G production and processing in the U.S. on air quality and human health related to both gas-phase and aerosol pollutants using an integrated assessment framework previously used to evaluate health impacts of oil & gas 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_X 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_X 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_2 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; Wyat 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 aero6 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 $PM_{2.5}$ 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 $PM_{2.5}$, NO_2, 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;

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

3. Results and Discussion

Flare and Venting Emissions

The inclusion of $PM_{2.5}$ and SO_2 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 $PM_{2.5}$ emissions are 13 times and 15 times higher in *wFlare1* and *wFlare2*, respectively. FV SO_2 emissions are more than two times higher in *wFlare1* and *wFlare2* than in NEI 2017. As discussed in S.2, O&G SO_2 emissions are highly underestimated in the NEI 2017 and our FV SO_2 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_X emissions are lower in *wFlare1* than in NEI 2017. In *wFlare2* where excess emissions from Rystad and NEI 2017's estimates are considered, FV NOx emissions are 22% higher than NEI 2017 and VOC emissions match the NEI 2017's estimates. NH_3 emissions are identical among all three estimates as NH_3 emissions are only accounted as point flares in the NEI 2017 and no NH_3 emissions were accounted for VIIRS-detected flares.

Large gaps exist between VIIRS-detected and NEI-derived flares. Figure 1 shows NO_X and $PM_{2.5}$ 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, Ohia and West Virginia (Figure S6). In Pennsylvania (Appalachian basin), between *wFlare2* and wFlare1 there is a distinctly high FV NOx emissions hotspot which comes from a single flare-SCC (2310021500) for flaring from an onshore gas well completion. This distinctly high NO_X emission is attributed as an artifact in NEI 2017, and is treated "as is" in this study. As $PM_{2.5}$ emissions were only estimated for VIIRS-detected flares, there are no $PM_{2.5}$ emissions differences between *wFlare2* and *wFlare1*. $PM_{2.5}$ emissions from FV account for about 82% of total O&G $PM_{2.5}$ emissions (Table S6). In this study, FV SO_2 emissions account for 82% of total O&G SO_2 emissions, and this high percentage is attributed to underestimation of O&G SO_2 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 $PM_{2.5}$) 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 $PM_{2.5}$ 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)

	$PM_{2.5}$	**NH_3**	**VOC**	**CO**	**NO_X**	**SO_2**
NEI 2017 point flare	7.82E+01	1.84E-01	4.59E+03	8.44E+03	2.71E+03	1.53E+04
NEI 2017 nonpoint flare	2.22E+02	0	1.31E+06	3.94E+04	1.87E+04	3.37E+04
Total NEI 2017 flare	3.00E+02	1.84E-01	1.31E+06	4.78E+04	2.14E+04	4.90E+04
VIIRS-only flare	4.34E+03 ± 729	0	1.04E+06	6.79E+04 ± 3,066	1.42E+04 ± 987	8.98E+04 ±15,166
VIIRS + Rystad + NEI hybrid	4.82E+03	0	1.31E+06	7.75E+04	2.33E+04	1.00E+05
*Total wFlare1**	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 %)
*Total wFlare2**	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*

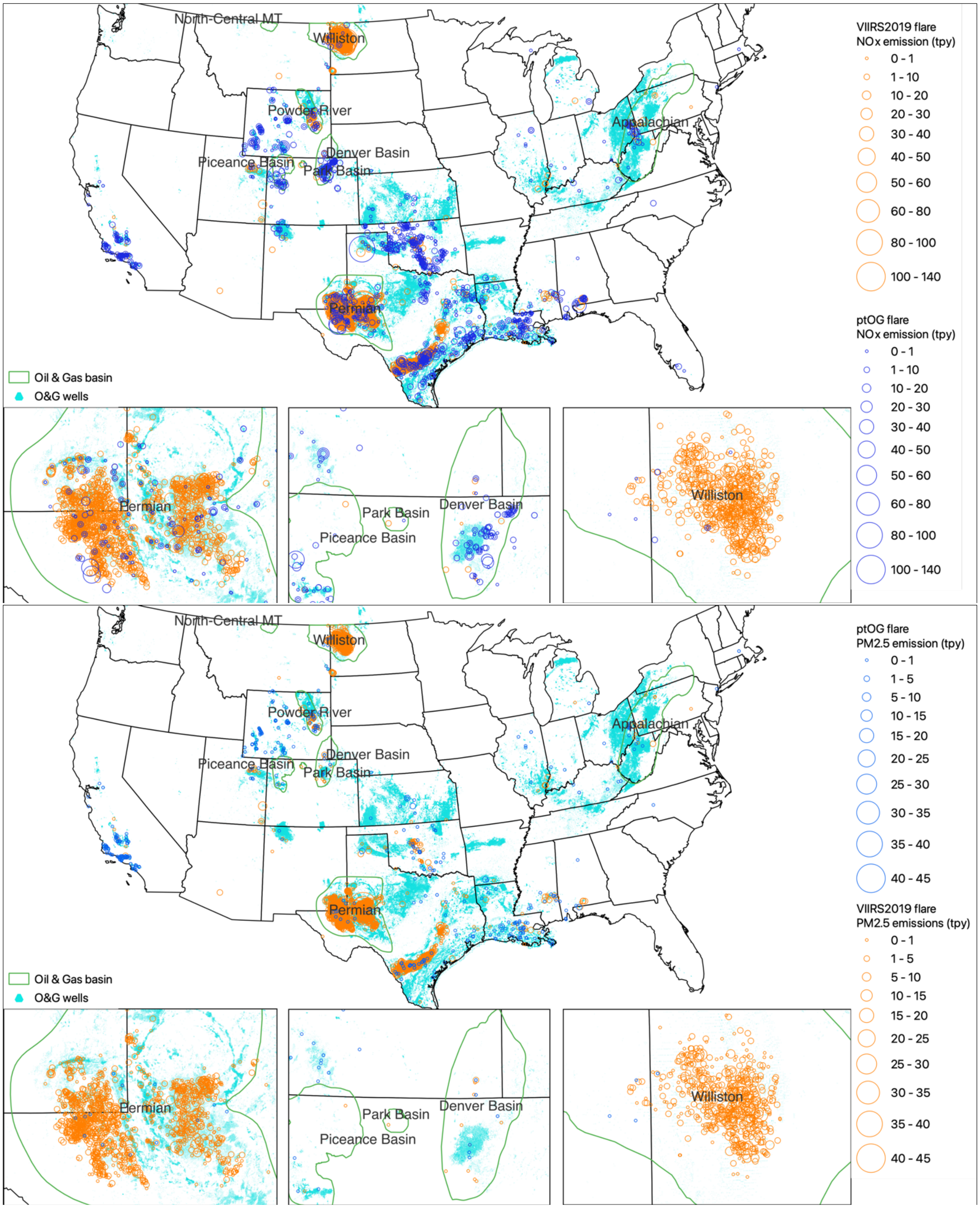

Figure 1. Combined (top) NO_X and (bottom) $PM_{2.5}$ emissions from FV as estimated from VIIRS and as derived from NEI 2017 point O&G (ptOG flare)

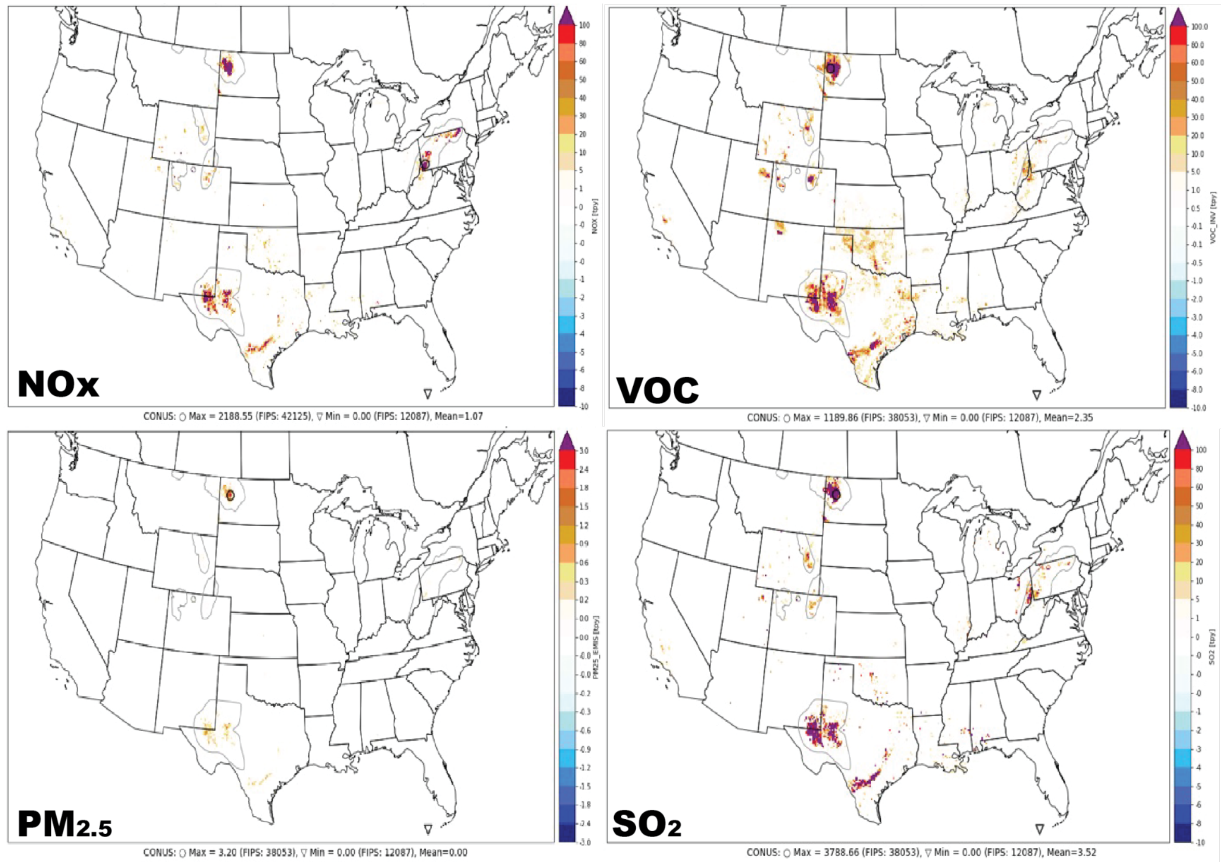

Figure 2. Annual emissions (tpy) NO_X, VOC, $PM_{2.5}$ and SO_2 from flaring and venting (FV)

Impact of Flaring and Venting on Air Quality

Domain-wide annual average impacts of FV on O_3, NO_2 and $PM_{2.5}$ 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 MDA8O3

FV's impact on 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_3 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_3 (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_3 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 NO_X emissions in the Appalachian basin enhanced O_3 formation for the area in July (up to 3 ppb) but suppressed O_3 formation by as much as -3.5 ppb in other months (Figure S9). Differences in FV's impact on O_3 between *wFlare2* and *wFlare1* scenarios (Figure S8) closely follow the differences in NO_X and VOC emissions between the two scenarios (Figure S6) and is less than 2 ppb for impacts on $MDA8O_3$.

Impacts on NO_2

In addition to heterogeneous distribution in space and time, FV's impacts on NO_2 have a distinguished distribution: increases of NO_2 are mostly localized to areas with FV emissions, and decreases of NO_2 are observed mostly elsewhere (Figure 4, Figure S10). The response of NO_2 concentration to FV's emissions highly depends on local background chemistry. During the daytime, NO_2 and NO interconvert through photolysis reactions and through reactions with O_3, organic (ROx) and hydrogen oxide radicals (HOx). NO_X is terminated by forming nitric acid (HNO_3) and dinitrogen pentoxide (N_2O_5, nighttime only), in which both are precursors of nitrate aerosols. In FV-major areas, NO-to-NO_2 conversation is enhanced by VOC emissions which results in net increase of NO_2, thus serving to quench the HOx-NOx cycle and shorten the OH chain (Womack et al., 2019). Such NO_2-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_2 through this pathway is higher in January than other months and is lowest in July (Figure S10). Monthly average hourly NO_2 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_2 in July. Appalachian basin observes the highest contribution of hourly (up to 8 ppb, or 46% of total NO_2) and daily-maximum (up to 25 ppb, 56%) NO_2, 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_2 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_2 are negligible (±0.02ppb, ± 0.4%) in both hourly and daily maximum (Figure 4).

Impacts on $PM_{2.5}$

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

FV's contribution to $PM_{2.5}$ varies with $PM_{2.5}$ compositions which differ significantly among FV-major as well as non-FV areas (Figure S12). In areas outside FV-major areas, sulfate aerosol (SO_4^-) is the largest component (48%), followed by elemental carbon (EC; 18%), NH_4^+ (15%), organic carbon (OC, 10%) and NO_3^- (4%). In the Bakken basin, the major $PM_{2.5}$ component is

EC (61%), followed by NO_3^- (22%), SO_4^- (14%) and less than 3% of other components. A similar distribution is observed in the Permian basin where 50% of $PM_{2.5}$ is EC followed by NO_3^- (29%), SO_4^- (12%), OC (7%) and NH_4^+ (2%). Differing significantly from other FV-major areas, $PM_{2.5}$ in the Denver basin has NO_3^- as a major component (42%), followed by EC (28%), NH_4^+ (13%), OC (12%) and SO_4^- (2%). In relevant to earlier discussions on NO_2 formation in FV-major areas, FV's contributions NO_3^- and NH_4^+ aerosols are largest in Denver basin. Meanwhile, FV's contributions to EC and SO_4^- are largest in Bakken basin. Reductions of NO_3^- and OC aerosol led to reductions of total $PM_{2.5}$ in Minnesota, Iowa, and other states in northeast of CONUS (Figure S12). Since there is negligible difference in primary $PM_{2.5}$ emissions from FV between *wFlare2* and *wFlare1*, the differences in $PM_{2.5}$ contributions between the two scenarios (< 0.164 $\mu g/m^3$ on annual average) are caused by secondary aerosols which are dominated by their inorganic components, i.e., NO_3^- and SO_4^- (Figure S8).

We found $PM_{2.5}$ contributions from FV are mainly driven by its contribution of SO_2 and primary $PM_{2.5}$ (mostly EC) emissions across CONUS especially in FV-major areas. Whereas in Denver basin and its downwind areas, we found increases in NO_3^- due to FV emissions enhancing the formation of nitric acid which favors the formation of NO_3^- .

Impacts on exceedance counts

Overall, FV emissions caused over 210 instances of MDA8O3 exceedances (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 $PM_{2.5}$ exceedances ($PM_{2.5}$ > 35 $\mu g/m^3$) is small. While FV added two additional $PM_{2.5}$ exceedances in Pennsylvania and New Jersey, it also reduced two exceedances in Minnesota and New York due to a reduction of $PM_{2.5}$ in these two states in the winter. For comparison, Buonocore et al. (2023) found 29 instances of $PM_{2.5}$ daily exceedances caused by O&G sector in 2016. If $PM_{2.5}$ daily NAAQS were lowered to 30 ug/m^3, FV emissions would contribute 10 instances of exceedances over 4 simulated months. No additional annual $PM_{2.5}$ exceedances (annual average $PM_{2.5}$ > 12 ug/m^3) were found. FV would have contributed 3 and 1 additional instances of annual exceedances in Pennsylvania and Illinois, respectively, if the annual $PM_{2.5}$ standard were lowered to 10 ug/m^3 (EPA, 2023).

FV causes no additional NO_2 exceedances (NO_2 > 100 ppb) which is expected given that Buonocore et al. (2023) found no NO_2 exceedances caused by total O&G in 2016. However, if the 1-hour 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 O_3, 0.17 ppb of NO_2 and 0.065 ug/m^3 of $PM_{2.5}$ on average over CONUS. This study found the corresponding contributions from FV are 0.026 ppb, 0.03 ppb and 0.008 ug/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_2 (25 ppb) and daily average $PM_{2.5}$ (5 ug/m^3) 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^3, 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.

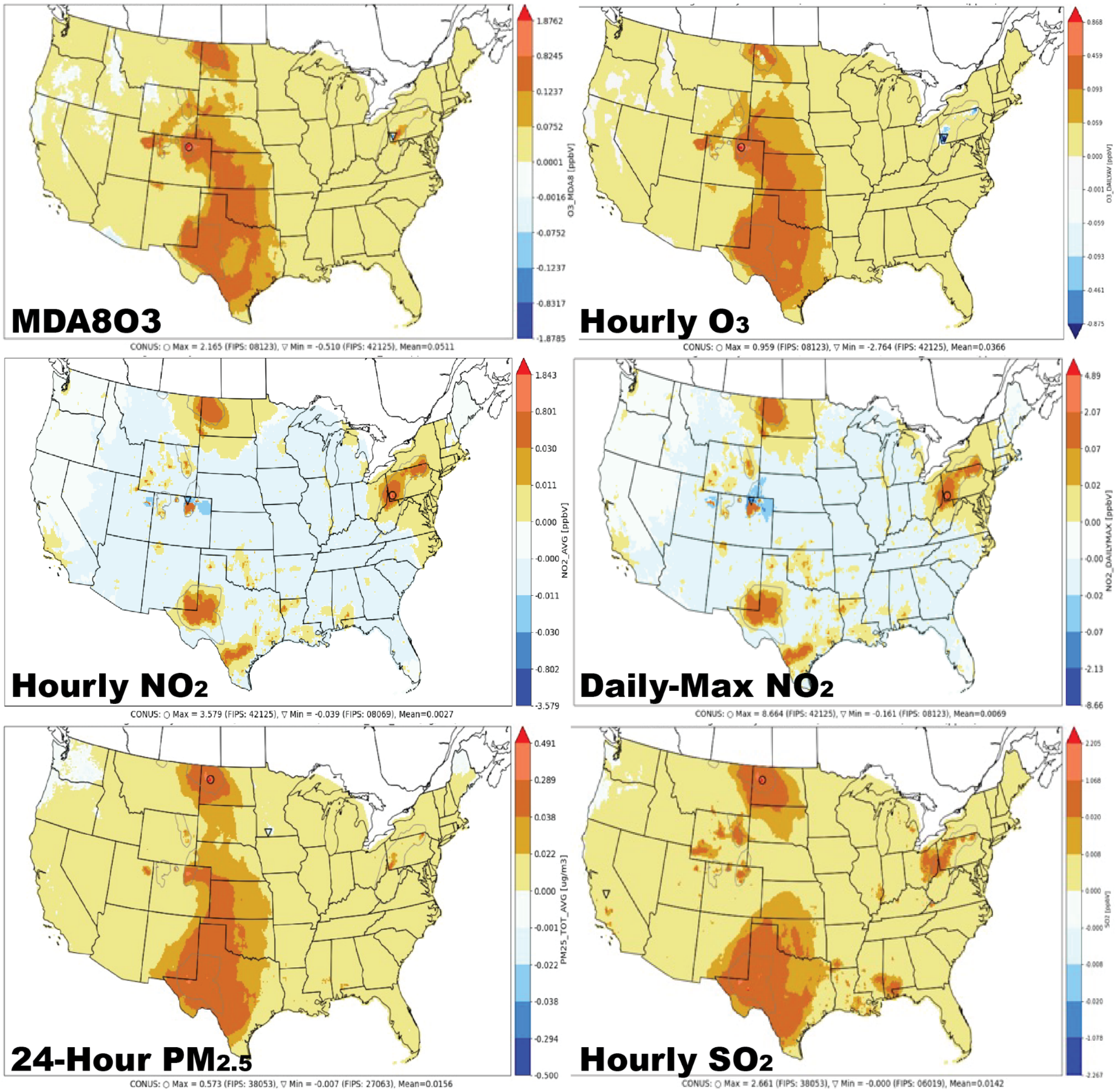

Figure 3. Annual-average of MDA8 Ozone (ppb), 24-hour average $PM_{2.5}$ (ug/m^3), daily-average and daily-maximum NO_2 (ppb), SO_2 (ppb) contributed by FV (i.e., differences between wFlare2 and woFlare)

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 $PM_{2.5}$, NO_2, 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 $PM_{2.5}$, NO_2, 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 $PM_{2.5}$ and NO_2, and $PM_{2.5}$ 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 NOx emissions, 81% of SO_2, 51% of VOCs, and 18% of $PM_{2.5}$ from oil & gas. The relative proportions that NO_2 from flaring contributes to total sector deaths and asthma exacerbations (Table S11) indicates the strong role of NO_2 in driving total health impacts.

As a whole, asthma outcomes calculated using all Althanti CRFs (for NO_2, $PM_{2.5}$ and ozone) are approximately three times larger than those calculated using all of the Orellano CRFs (NO_2 and $PM_{2.5}$). 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 $PM_{2.5}$ and NO_2 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 $PM_{2.5}$, O_3 and NO_2, 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 65th 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 65th 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_2 has such a strong role, this indicates that health benefits of emission control strategies can be increased by expanding coverage to more NO_X-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, $PM_{2.5}$, and NO_2, 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. (Smithosian, 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_2 exposure, and health outcomes not regularly included in air pollution health impact assessments, most notably birth and children's health outcomes.

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	$PM_{2.5}$	360 (300 - 420)	$3,700,000,000 ($1,900,000,000 - $6,000,000,000)
	O_3	230 (120 - 470)	$2,400,000,000 ($720,000,000 - $6,800,000,000)
	NO_2	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	$PM_{2.5}$	140 (47 - 230)	$8,200,000 ($1,100,000 - $22,000,000)
	NO_2	47 (19 - 65)	$2,800,000 ($460,000 - $6,100,000)
	$PM_{2.5}$ and NO_2	190 (66 - 300)	$11,000,000 ($1,600,000 - $28,000,000)
Asthma Hospitalizations (Alhanti)	$PM_{2.5}$	1.3 (0.63 - 2.5)	$23,000 ($12,000 - $46,000)
	O_3	5.7 (3.2 - 8.1)	$100,000 ($59,000 - $150,000)
	NO_2	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)	$PM_{2.5}$	9,700 (4,900 - 19,000)	$580,000 ($110,000 - $1,900,000)
	O_3	43,000 (24,000 - 61,000)	$2,500,000 ($530,000 - $6,000,000)
	NO_2	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)	$PM_{2.5}$	13 (6.3 - 25)	$5,700 ($2,700 - $12,000)
	O_3	54 (31 - 77)	$24,000 ($13,000 - $36,000)
	NO2	25 (20 - 36)	$11,000 ($8,800 - $17,000)
	All Three	92 (58 - 140)	$42,000 ($25,000 - $65,000)
Respiratory Hospitalizations	$PM_{2.5}$	19 (9.8 - 28)	$570,000 ($290,000 - $840,000)
	O_3	110 (40 - 180)	$3,400,000 ($1,200,000 - $5,500,000)
	$PM_{2.5}$ and Ozone	130 (50 - 210)	$3,900,000 ($1,500,000 - $6,400,000)
Heart Attacks	$PM_{2.5}$	16 (9.7 - 23)	$1,100,000 ($680,000 - $1,600,000)
	NO_2	6.9 (3.7 - 10)	$480,000 ($260,000 - $700,000)
	$PM_{2.5}$ and NO_2	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)

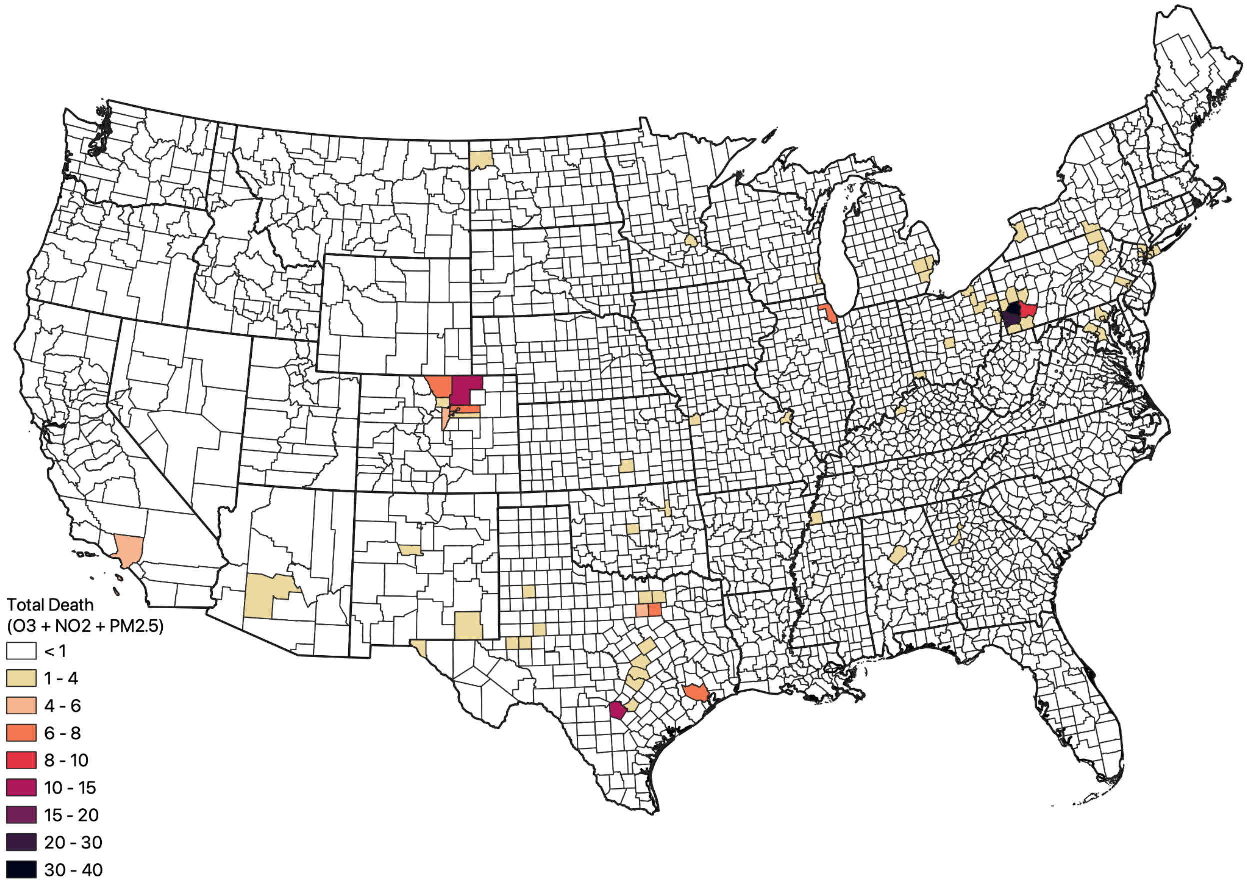

Figure 4. FV air pollution-attributable deaths in 2016

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 $PM_{2.5}$, 230 to O_3 and 120 to NO_2. This finding signifies that while most health impact studies so far have been focused on $PM_{2.5}$, health impact from O_3 & NO_2 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

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 ≥ 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/cmf-icd10.html; user's agreement to data use restrictions is required.

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

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