

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

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14 **Key points**

15 A new satellite-based emissions inventory for oil and gas flaring and venting is developed to  
16 refine existing U.S. emissions inventories.  
17

18 710 premature deaths dominate the \$7.3B monetized values of health impacts due to this sector.  
19

20 Significant benefits in air quality and health could be gained by reducing emissions from flaring  
21 and venting activities.  
22

23 **Abstract**

24 Emissions from flaring and venting (FV) in oil and gas (O&G) production are difficult to  
25 quantify due to their intermittent activities and lack of adequate monitoring and reporting. Given  
26 their potentially significant contribution to total emissions from O&G sector in the United States,  
27 we estimate emissions from FV using Visible Infrared Imaging Radiometer Suite satellite  
28 observations and state/local reported data on flared gas volume. These refined estimates are  
29 higher than those reported in the National Emission Inventory: by upto 15 times for fine  
30 particulate matter (PM<sub>2.5</sub>), two times for sulfur dioxides, and 22% higher for nitrogen  
31 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  
32 conterminous U.S. (CONUS) are less than 0.15%, but significant contributions of upto 60% are  
33 found in O&G fields with FV. FV contributions are higher in winter than in summer months for  
34 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  
35 highest in the Denver basin whereas in the Permian and Bakken basins, sulfate and elemental  
36 carbon aerosols are the major contributors. Over four simulated months in 2016 for the entire  
37 CONUS, FV contributes 210 additional instances of exceedances to the daily maximum 8-hour  
38 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  
39 form of the national ambient air quality standards. FV emissions are found to cause over \$7.4  
40 billion in health damages, 710 premature deaths and 73,000 asthma exacerbations among  
41 children annually.

42

43 **Plain language summary**

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

## 51 **1. Introduction**

52 Flaring is an oil and gas (O&G) industry term to describe the practice of burning off excess  
53 natural gas that is produced along with crude oil, often called ‘associated gas’ or ‘associated  
54 petroleum gas’. This associated gas is a valuable commodity when it can be appropriately  
55 separated from oil and transported. In practice, however, a facility to support such processing is  
56 often absent, and thus flaring is used as a way to dispose of unwanted gas that would otherwise  
57 pose a safety hazard or interfere with oil production (DOE, 2019; GGFR, 2023). Multiple  
58 economic and technical reasons for why flaring of associated gas is needed are discussed by  
59 Soltanieh et al. (2016). According to the World Bank’s Global Gas Flaring Reduction  
60 Partnership (GGFR, 2023), global gas flaring stayed relatively constant throughout 2010 – 2020  
61 and reached 150 billion cubic meters (BCM) in 2020, equivalent to the total annual gas  
62 consumption of sub-Saharan Africa, and the top five flaring countries are Russia (24 BCM), Iraq  
63 (17 BCM), Iran (13 BCM), U.S. (12 BCM), Algeria (9 BCM), and Venezuela (8 BCM). Venting  
64 of associated gas from O&G compression or processing equipment due to system upset  
65 conditions or pressure release during emergency is also common in O&G production and  
66 processing (DOE, 2019).

67  
68 Besides emitting carbon dioxide (CO<sub>2</sub>), O&G flaring release various pollutants including  
69 methane (CH<sub>4</sub>), black carbon (soot), nitrogen oxides (NO<sub>x</sub>), sulfur dioxide (SO<sub>2</sub>), carbon  
70 monoxide (CO), various volatile organic compounds (VOCs) depending on flaring conditions  
71 and composition of the associated gas (Anejionu et al., 2015; Fawole et al., 2016; Umukoro &  
72 Ismail, 2017), all of which can cause various impacts on climate, air quality and human health.  
73 According to the World Bank’s Global Gas Flaring Tracker Report, flaring released over 400  
74 million tons of carbon dioxide equivalent (CO<sub>2</sub>e) emissions into the atmosphere in 2020 (World  
75 Bank, 2022); such amount is roughly equivalent to the greenhouse gas emissions of around 77  
76 million cars. Allen et al. (2016) estimated that O&G flaring contribute 20 – 21 million metric  
77 tons of CO<sub>2</sub>e of greenhouse gases per year in the U.S. Evaluating O&G flaring’s impact on  
78 nationally determined contributions (NDC) defined under the United Nations Framework  
79 Convention on Climate Change (UNFCCC) Paris Agreement, Elvidge et al. (2018) found global  
80 flaring represents less than 2% of the NDC reduction target, however, some countries (e.g.,  
81 Yemen, Algeria, Iraq) may fully meet their NDC reduction target by just controlling for flaring.

82  
83 Cushing et al. (2021) reported that flaring can cause negative effects on birth outcomes, such as  
84 preterm births, due to exposure to polycyclic aromatic hydrocarbons, and there are more than  
85 500,000 Americans living within 3 miles of natural gas flares potentially exposed and at risk of  
86 adverse health effects. O&G production in the U.S. has adverse health impacts of 7,500  
87 premature deaths and 410,000 asthma exacerbations annually (Buonocore et al, 2023). Emissions  
88 from flaring cause an increase in respiratory diseases, heart diseases and strokes due to black  
89 carbon particle exposure (Chen et al., 2022). Johnston et al. (2020) report flaring can impair lung  
90 function and cause asthma due to exposure to volatile organic compounds. Globally, flaring  
91 contributed about 0.12% of the health impacts related to PM<sub>2.5</sub> and 6.51% of the health impacts  
92 related to climate change (Motte et al., 2021). Motte et al. (2021) also found that health impacts  
93 related to gas-phase pollutants from flaring are negligible.

94  
95 Operators of O&G production facilities which perform flaring and venting (FV) report the  
96 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).

119  
120

## 121 **2. Materials and Methods**

### 122 **Flare emissions from NEI**

123 We identified 22 source classification codes (SCC) of O&G point sources from the NEI 2017  
124 that have “flare” keyword in the descriptions of SCC (Table S1). Some of these 22 SCCs have  
125 zero emissions, however, and the majority of emissions are from the top four SCCs in Table S1.  
126 Searching for “flare” keywords in SCCs of O&G nonpoint sources in the NEI 2017 returned only  
127 6 SCCs that represent flaring from well completions, and only two SCC have non-zero  
128 emissions. Furthermore, these nonpoint flare-SCCs were introduced in NEI 2017 but not in  
129 earlier NEI versions (NEI 2014 and NEI 2016). Through personal communications with  
130 technical staff in charge of O&G emissions inventory development in EPA OAQPS, TCEQ,  
131 Colorado DHCP, Wyoming DEQ and Utah DEQ, we learned that flare emissions may be  
132 associated with 19 nonpoint SCCs as shown in Table S2 and Table S3, though most of them do  
133 not have the keyword “flare” in the SCC descriptions.

134  
135 It is worth highlighting that flare emissions are not separated out but rather lumped into total  
136 emissions from these 19 nonpoint-SCCs. Furthermore, not all O&G equipment (e.g., condensate  
137 tanks, crude oil tanks) represented by these SCCs are equipped with flares. Therefore, flare  
138 emissions from the 19 nonpoint-SCCs vary among counties in the U.S., the spatial resolution at  
139 which the nonpoint O&G emissions are allocated in the NEI. To estimate NEI 2017-derived flare  
140 emissions for each county in the CONUS, we first applied records with nonzero NO<sub>x</sub> emissions  
141 as an indicator for with-flare emissions from each of the 19 nonpoint flare-SCCs, then derived  
142 emissions of other criteria pollutants from the same SCC as considered from flare.

143  
144 Special treatment is applied for VOC emissions from flares: due to the high destruction  
145 efficiency of flares (> 95%), VOC emissions coming out from flare stack mounts would be close  
146 to zero. (Thus, to our knowledge there is no VOC emissions factor developed for the flaring  
147 process). However, flares are not always operating properly which can lead to increased venting  
148 of natural gas through the flare stack flaring (DOE, 2019; Lyon et al., 2021; Plant et al., 2022;  
149 Reuters, 2022; Tyner & Johnson, 2021). For example, Lyon et al. (2021) reported that 11% of  
150 surveyed flares in the Permian Basin had combustion issues and 5% were unlit and emitted  
151 uncombusted gas directly into the atmosphere. Furthermore, if flaring was to be eliminated and  
152 the would-be gas captured instead through other means of emission controls, it is reasonable to  
153 assume the vented gas at the facility could be captured in the same way. Therefore, in this study,  
154 we considered VOC emissions from the 19 nonpoint flare-SCCs as from venting and combined  
155 this VOC emissions estimate with non-VOC emissions from flare into the same group of “flaring  
156 and venting” (FV). Note that “venting” in this way only represents VOC emissions from O&G  
157 sources equipped with a flare using the nonzero NO<sub>x</sub> emissions criteria as discussed above.

158  
159 Figure S1 shows emissions of criteria pollutants from O&G categories including point- and  
160 nonpoint-flares as derived from NEI 2017. FV is also found to account for 10% of the total 7.3  
161 million tons per year of methane from O&G although this is not a targeted pollutant for  
162 discussion on air quality and health impacts in this study.

163

### 164 **Flare emissions estimation from VIIRS**

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

167 2020 (Figure 1). In 2019, the year with highest flare gas volume during the period 2017 – 2020,  
168 there were 17.7 BCM of natural gas flared in both production (97%) and processing (3%) – the  
169 same amount of flare gas volume was reported in Chen et al. (2022). We performed the  
170 following processing on VIIRS data prior to emissions estimation: only onshore flares were  
171 considered; VIIRS-detected flares are excluded if they are found to be in close proximity of NEI  
172 2017-point flare sources to avoid double counting. As a result, there were 17.3 BCM of VIIRS-  
173 detected flared gas remaining for emissions estimation.

174

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

179

180 There is a high probability that not all flares were adequately detected by VIIRS. Certain criteria  
181 must be met for a flare to be detected by VIIRS such as flared temperature > 1,400 K, frequent  
182 combustion (consistency) and free of cloud cover and other contaminations (C. Elvidge et al.,  
183 2013, 2015). In our case study for the Uinta O&G basin in Utah, we compared the number of  
184 VIIRS-detected flares with the self-reported flare data from O&G operators (UDEQ, 2022) and  
185 found that only 11 (or 8%) of the 132 reported flares were detected by VIIRS in 2019.

186

187 Another uncertainty in VIIRS flare data is the flare gas volume (FGV). VIIRS estimates FGV by  
188 applying regression algorithms to the relationship between radiated heat energy from detected  
189 flare to the reported FGV at state and county levels (C. Elvidge et al., 2013, 2015). Large gaps  
190 exist when VIIRS-estimated FGV in 2019 are compared to FGV reported by Rystad Energy  
191 (2022) for the same year (Table S4). For example, Rystad Energy (2022) reports FGV in  
192 Colorado in 2019 to be 136 MCF/yr whereas only 18 MCF/yr is estimated by VIIRS. Non-  
193 detection of flares by VIIRS (more discussion in later) in Colorado are partially attributed to this  
194 gap.

195

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

207

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

209 The model configurations in this study refined the configurations applied in Buonocore et al.  
210 (2023) which evaluates impacts of O&G emissions to air quality and public health in 2016. As  
211 such, all anthropogenic and wildfire emissions other than O&G are based on the NEI 2016 v1.  
212 Emissions from all other O&G sources other than FV are taken from the NEI 2017. The use of

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

217  
218 The Models-3 / Community Multiscale Air Quality (CMAQ) modeling system (Byun & Schere,  
219 2006; Wyatt Appel et al., 2018) version 5.2.1 was utilized to simulate atmospheric chemistry with  
220 Carbon-Bond version 6 revision 3 (CB6r3) gaseous chemistry and aero6 for aerosol treatment.  
221 Meteorological inputs are derived from the Weather Research and Forecasting model  
222 (Skamarock et al., 2008) version 4.7. WRF-CMAQ simulations were performed for January,  
223 April, July and October 2016 (to represent four seasons) for a modeling domain covering the  
224 conterminous U.S. (CONUS) in 12 km x 12 km horizontal grid resolution and 35 vertical layers  
225 (12US1 domain). Boundary and initial chemistry conditions were taken from the hemispheric  
226 CMAQ (HCMAQ) version 5.2.1 simulation for northern hemisphere. Evaluation of CMAQ  
227 model performance for key pollutants of interest are briefly discussed in Supporting Information  
228 S3.

229  
230 Two model scenarios included FV emissions as estimated in *wFlare1* and *wFlare2* (discussed  
231 above) and all other non-FV emissions from all other anthropogenic and natural sources in the  
232 domain. FV emissions are excluded in a zero-out scenario (*woFlare*). Simulation results of  
233 *wFlare1* and *wFlare2* are compared against *woFlare*, alternatively, to quantify the impact of FV  
234 emissions on air quality and human health. For brevity, however, discussions on the impacts in  
235 the following sections are based on *wFlare2* scenario unless specified otherwise.

236

### 237 **Analyses**

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

247

### 248 **Method for health impact analyses**

249 To calculate the health impacts of flaring, we subtracted the health impacts of air pollution under  
250 the Baseline/No-flaring scenario from those of the *wFlare2* scenario to isolate the health impacts  
251 from the flare portion of *wFlare2* scenario emissions. To estimate the health impacts of ambient  
252 ground level concentrations of PM<sub>2.5</sub>, NO<sub>2</sub>, and ozone under each scenario, we used BenMAPR  
253 which is a geospatial health impact assessment model in R and based on the Benefits Mapping  
254 and Analysis Program (BenMAP) from the U.S. EPA (Sacks et al., 2018), and used in two recent  
255 studies (Arter et al. 2022; Buonocore et al. 2023). BenMAPR accepts air pollution concentration  
256 outputs from CMAQ, and overlays it with population data from the U.S. American Community  
257 Survey from the U.S. Census Bureau; data on background rates of health outcomes from the U.S.  
258 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.

### 265 3. Results and Discussion

266

#### 267 Flare and Venting Emissions

268 The inclusion of PM<sub>2.5</sub> and SO<sub>2</sub> to FV emissions resulted in significantly higher emissions of the  
269 two pollutants in both *wFlare1* and *wFlare2* than in NEI 2017 over the entire CONUS (Table 1).  
270 Compared to NEI 2017, FV PM<sub>2.5</sub> emissions are 13 times and 15 times higher in *wFlare1* and  
271 *wFlare2*, respectively. FV SO<sub>2</sub> emissions are more than two times higher in *wFlare1* and  
272 *wFlare2* than in NEI 2017. As discussed in S.2, O&G SO<sub>2</sub> emissions are highly underestimated  
273 in the NEI 2017 and our FV SO<sub>2</sub> emissions are likely closer to actual emissions. Since VIIRS did  
274 not detect flares in some counties where FV emissions are reported in NEI 2017, VOC and NO<sub>x</sub>  
275 emissions are lower in *wFlare1* than in NEI 2017. In *wFlare2* where excess emissions from  
276 Rystad and NEI 2017's estimates are considered, FV NO<sub>x</sub> emissions are 22% higher than NEI  
277 2017 and VOC emissions match the NEI 2017's estimates. NH<sub>3</sub> emissions are identical among  
278 all three estimates as NH<sub>3</sub> emissions are only accounted as point flares in the NEI 2017 and no  
279 NH<sub>3</sub> emissions were accounted for VIIRS-detected flares.

280

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

291

292 In accordance with the distribution of VIIRS-detected flares, emissions from FV are most  
293 noticeable in major O&G production fields in the U.S., especially those in NM, TX, ND, CO and  
294 WY (Figure 2). Emissions differences between *wFlare2* and *wFlare1* (i.e., FV emissions  
295 accounted for in the NEI 2017 but not in VIIRS) are shown in these states but also noticeable in  
296 Oklahoma, Kansas, Pennsylvania, Ohio and West Virginia (Figure S6). In Pennsylvania  
297 (Appalachian basin), between *wFlare2* and *wFlare1* there is a distinctly high FV NO<sub>x</sub> emissions  
298 hotspot which comes from a single flare-SCC (2310021500) for flaring from an onshore gas well  
299 completion. This distinctly high NO<sub>x</sub> emission is attributed as an artifact in NEI 2017, and is  
300 treated "as is" in this study. As PM<sub>2.5</sub> emissions were only estimated for VIIRS-detected flares,  
301 there are no PM<sub>2.5</sub> emissions differences between *wFlare2* and *wFlare1*. PM<sub>2.5</sub> emissions from  
302 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>  
303 emissions account for 82% of total O&G SO<sub>2</sub> emissions, and this high percentage is attributed to  
304 underestimation of O&G SO<sub>2</sub> emissions in the NEI 2017 (see SI.2). VOC emissions from FV  
305 account for about 50% of total O&G VOC emissions over CONUS mainly due to the inclusion  
306 of storage tank's venting. The benefit of treating FV VOC emissions in this study, however,  
307 provides an opportunity to quantify air quality and health benefits from potentially controlling  
308 both flaring and venting together.

309

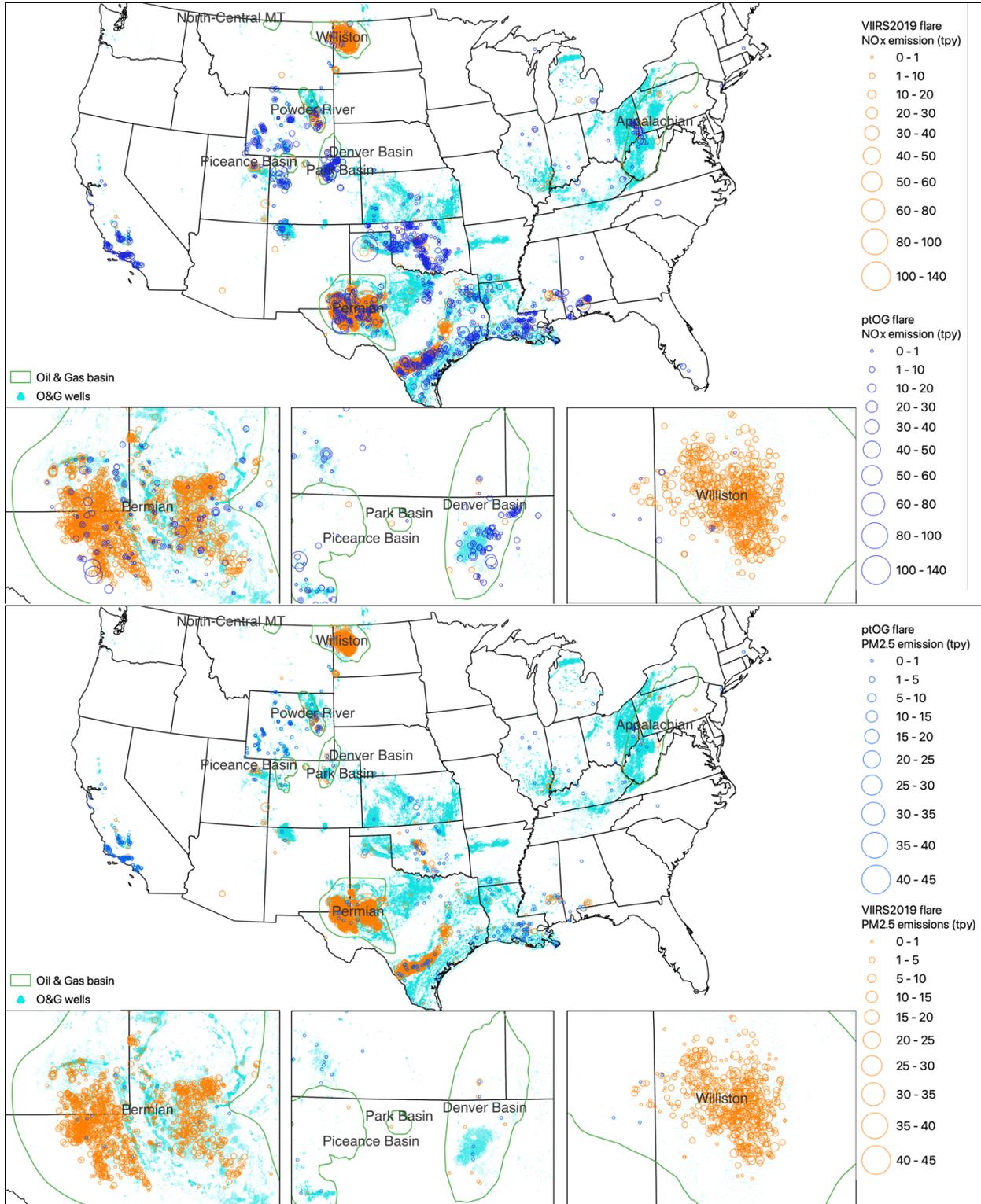
310 This study estimates BC (analogously to primary PM<sub>2.5</sub>) emissions in CONUS from FV to be  
 311 4,340 ± 729 tpy with Texas (2,244 ± 250 tpy) and North Dakota (1,713 ± 216 tpy) as the top two  
 312 emitting states. Chen et al. (2022) estimated black carbon (BC) emission from upstream flaring  
 313 in the CONUS to be 15,986 tpy, with North Dakota (10,036 tpy) and Texas (4,317) as the two  
 314 leading states. A different set of emission factors and heating values applied in Chen et al. (2022)  
 315 gives reason for their higher estimations of PM<sub>2.5</sub> emissions than in this study. Although the  
 316 VIIRS-detected flare gas volume is higher in Texas (8.73 BCM) than in North Dakota (6.09  
 317 BCM), Chen et al. (2022) estimated higher black carbon in North Dakota because of the higher  
 318 heating value applied to this region. Based on in-situ measurements in 2013 – 2014, Schwarz et  
 319 al. (2015) estimate BC emissions in Bakken basin to be 1,400 ± 360 tpy, which is relatively  
 320 closer to our estimates. Only about 0.36 Gg per year (or 397 tpy) of BC emissions from flaring in  
 321 Bakken basin were estimated by Weyant et al. (2016).

322  
 323

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

	<b>PM<sub>2.5</sub></b>	<b>NH<sub>3</sub></b>	<b>VOC</b>	<b>CO</b>	<b>NO<sub>x</sub></b>	<b>SO<sub>2</sub></b>
<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 %)

324 \* Numbers in parentheses indicate changes in emissions from NEI 2017: e.g., 100\*(wFlare2 –  
 325 NEI 2017)/NEI 2017  
 326



327

328

329

330

331

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)

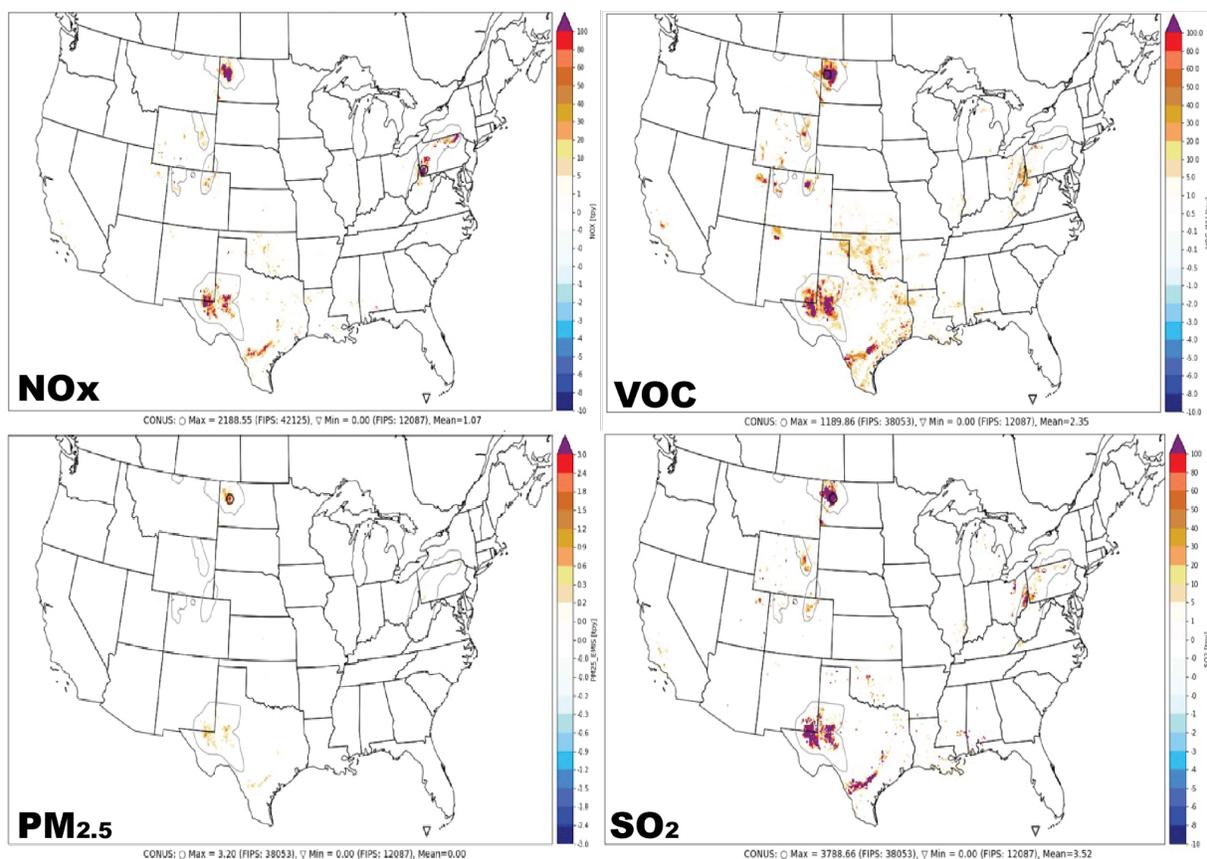


Figure 2. Annual emissions (tpy) NO<sub>x</sub>, VOC, PM<sub>2.5</sub> and SO<sub>2</sub> from flaring and venting (FV)

332

### 333 Impact of Flaring and Venting on Air Quality

334 Domain-wide annual average impacts of FV on O<sub>3</sub>, NO<sub>2</sub> and PM<sub>2.5</sub> concentrations in the CONUS  
 335 are relatively small (<0.15%). However, the impacts greatly vary with locations and seasons  
 336 (Figure S9 through S12) with the strongest impacts typically occurring in areas with intense FV  
 337 emissions (Figure 3), emphasizing the advantage of the detailed treatment of the emissions and  
 338 processes at the modeled grid resolution of 12 km x 12 km over the entire country.

339

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

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

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

354

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

356 In addition to heterogeneous distribution in space and time, FV's impacts on NO<sub>2</sub> have a  
357 distinguished distribution: increases of NO<sub>2</sub> are mostly localized to areas with FV emissions, and  
358 decreases of NO<sub>2</sub> are observed mostly elsewhere (Figure 4, Figure S10). The response of NO<sub>2</sub>  
359 concentration to FV's emissions highly depends on local background chemistry. During the  
360 daytime, NO<sub>2</sub> and NO interconvert through photolysis reactions and through reactions with O<sub>3</sub>,  
361 organic (RO<sub>x</sub>) and hydrogen oxide radicals (HO<sub>x</sub>). NO<sub>x</sub> is terminated by forming nitric acid  
362 (HNO<sub>3</sub>) and dinitrogen pentoxide (N<sub>2</sub>O<sub>5</sub>, nighttime only), in which both are precursors of nitrate  
363 aerosols. In FV-major areas, NO-to-NO<sub>2</sub> conversation is enhanced by VOC emissions which  
364 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  
365 chain (Womack et al., 2019). Such NO<sub>2</sub>-reduction effects are most noticeable downwind of the  
366 Denver basin.

367

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

379

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

381 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  
382 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%)  
383 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  
384 basins (Figure 4). The Denver basin observed the highest FV's contribution by as much as 5  
385  $\mu\text{g}/\text{m}^3$  to daily average PM<sub>2.5</sub>, whereas highest contributions in other FV-major areas are upto 1.5  
386  $\mu\text{g}/\text{m}^3$ . Negative PM<sub>2.5</sub> contributions (reductions due to FV emissions) occurred in Minnesota,  
387 Iowa, and other states in the northeast of CONUS in January, but at a relatively small margin (up  
388 to 0.06  $\mu\text{g}/\text{m}^3$  on monthly average). Positive monthly average PM<sub>2.5</sub> contribution from FV is  
389 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).  
390 On average over CONUS, however, FV's contribution to monthly-average of PM<sub>2.5</sub> is larger in  
391 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>  
392 contributions (or PM<sub>2.5</sub> reductions) mostly occurring in January (Figure S12).

393

394 FV's contribution to PM<sub>2.5</sub> varies with PM<sub>2.5</sub> compositions which differ significantly among FV-  
395 major as well as non-FV areas (Figure S12). In areas outside FV-major areas, sulfate aerosol  
396 (SO<sub>4</sub><sup>-</sup>) is the largest component (48%), followed by elemental carbon (EC; 18%), NH<sub>4</sub><sup>+</sup> (15%),  
397 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

398 EC (61%), followed by  $\text{NO}_3^-$  (22%),  $\text{SO}_4^-$  (14%) and less than 3% of other components. A  
399 similar distribution is observed in the Permian basin where 50% of  $\text{PM}_{2.5}$  is EC followed by  $\text{NO}_3^-$   
400 (29%),  $\text{SO}_4^-$  (12%), OC (7%) and  $\text{NH}_4^+$  (2%). Differing significantly from other FV-major areas,  
401  $\text{PM}_{2.5}$  in the Denver basin has  $\text{NO}_3^-$  as a major component (42%), followed by EC (28%),  $\text{NH}_4^+$   
402 (13%), OC (12%) and  $\text{SO}_4^-$  (2%). In relevant to earlier discussions on  $\text{NO}_2$  formation in FV-  
403 major areas, FV's contributions  $\text{NO}_3^-$  and  $\text{NH}_4^+$  aerosols are largest in Denver basin. Meanwhile,  
404 FV's contributions to EC and  $\text{SO}_4^-$  are largest in Bakken basin. Reductions of  $\text{NO}_3^-$  and OC  
405 aerosol led to reductions of total  $\text{PM}_{2.5}$  in Minnesota, Iowa, and other states in northeast of  
406 CONUS (Figure S12). Since there is negligible difference in primary  $\text{PM}_{2.5}$  emissions from FV  
407 between *wFlare2* and *wFlare1*, the differences in  $\text{PM}_{2.5}$  contributions between the two scenarios  
408 ( $< 0.164 \mu\text{g}/\text{m}^3$  on annual average) are caused by secondary aerosols which are dominated by  
409 their inorganic components, i.e.,  $\text{NO}_3^-$  and  $\text{SO}_4^-$  (Figure S8).

410  
411 We found  $\text{PM}_{2.5}$  contributions from FV are mainly driven by its contribution of  $\text{SO}_2$  and primary  
412  $\text{PM}_{2.5}$  (mostly EC) emissions across CONUS especially in FV-major areas. Whereas in Denver  
413 basin and its downwind areas, we found increases in  $\text{NO}_3^-$  due to FV emissions enhancing the  
414 formation of nitric acid which favors the formation of  $\text{NO}_3^-$ .

#### 415 416 Impacts on exceedance counts

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

424  
425 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  
426 two additional  $\text{PM}_{2.5}$  exceedances in Pennsylvania and New Jersey, it also reduced two  
427 exceedances in Minnesota and New York due to a reduction of  $\text{PM}_{2.5}$  in these two states in the  
428 winter. For comparison, Buonocore et al. (2023) found 29 instances of  $\text{PM}_{2.5}$  daily exceedances  
429 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  
430 would contribute 10 instances of exceedances over 4 simulated months. No additional annual  
431  $\text{PM}_{2.5}$  exceedances (annual average  $\text{PM}_{2.5} > 12 \mu\text{g}/\text{m}^3$ ) were found. FV would have contributed 3  
432 and 1 additional instances of annual exceedances in Pennsylvania and Illinois, respectively, if the  
433 annual  $\text{PM}_{2.5}$  standard were lowered to  $10 \mu\text{g}/\text{m}^3$  (EPA, 2023).

434  
435 FV causes no additional  $\text{NO}_2$  exceedances ( $\text{NO}_2 > 100$  ppb) which is expected given that  
436 Buonocore et al. (2023) found no  $\text{NO}_2$  exceedances caused by total O&G in 2016. However, if  
437 the 1-hour  $\text{NO}_2$  standard were lowered to 60 ppb then it would add 9 instances of exceedances in  
438 Colorado and 1 instance in Florida.

439  
440 Buonocore et al. (2023) estimated that in 2016 the O&G sector contributed 0.6 ppb of  $\text{O}_3$ , 0.17  
441 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  
442 corresponding contributions from FV are 0.026 ppb, 0.03 ppb and  $0.008 \mu\text{g}/\text{m}^3$ , respectively.  
443 Since FV emissions do not exist in all areas with O&G activities, such relatively small FV's

444 contribution in comparison to total O&G when taking average over CONUS is anticipated.  
445 However, the highest FV's contributions to MDA8O3 (15 ppb), daily maximum NO<sub>2</sub> (25 ppb)  
446 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  
447 Buonocore et al. (2023) (3 ppb, 17 ppb, and 1.7 ug/m<sup>3</sup>, respectively), emphasizing the near-field  
448 impacts of FV that one should focus on. Note that this study combined O&G emissions from  
449 NEI 2017 and FV emissions calculated based on VIIRS-Rystad-NEI hybrid dataset, whereas  
450 Buonocore et al. (2023) utilized O&G emissions from NEI 2016 as is. Regardless, this finding  
451 illustrated that estimation of impacts on O&G sectors on air quality, and consequently human  
452 health, could greatly vary with input emissions, and improvements in the emissions estimates as  
453 we have done provide increased confidence in the modeled estimates.  
454

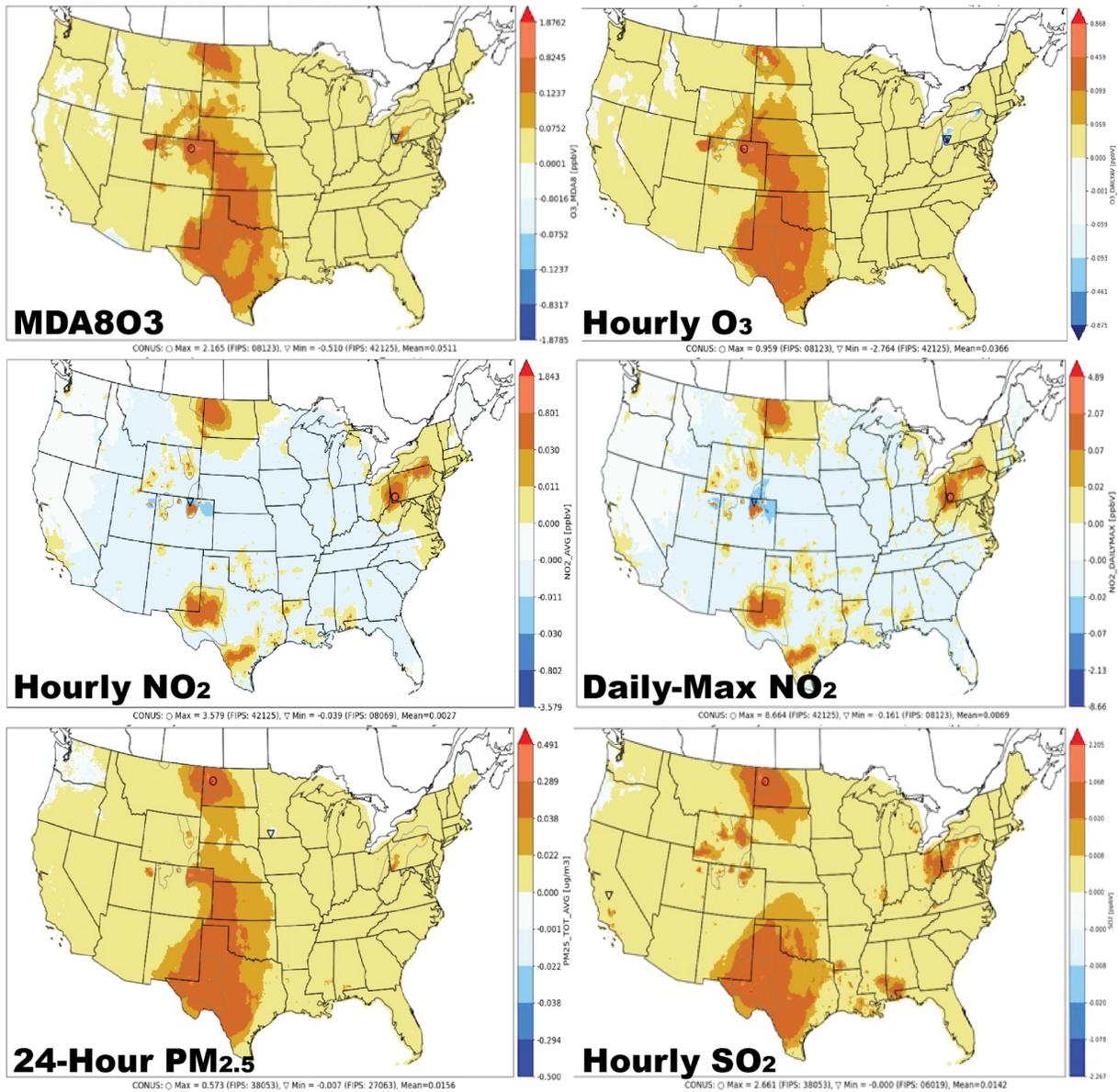


Figure 3. Annual-average of MDA8 Ozone (ppb), 24-hour average PM<sub>2.5</sub> (ug/m<sup>3</sup>), daily-average and daily-maximum NO<sub>2</sub> (ppb), SO<sub>2</sub> (ppb) contributed by FV (i.e., differences between wFlare2 and woFlare)

## 457 Health Impacts of Flaring and Venting

458

459 Our results show in 2016, emissions due to flaring in Flare Scenario 2 have a mortality burden of  
460 710 (95% CI: 480 – 1,100) excess deaths attributable to PM<sub>2.5</sub>, NO<sub>2</sub>, and ozone compared to  
461 baseline scenario emissions. Additionally, our results show an estimated annual excess of 73,000  
462 (95% CI: 46,000 – 110,000) childhood asthma exacerbations, 92 (95% CI: 58 – 140) childhood  
463 asthma emergency department visits, and 10 (95% CI: 6.4 – 15) asthma hospitalizations  
464 attributable to PM<sub>2.5</sub>, NO<sub>2</sub>, and ozone. An excess of 190 (95% CI: 66 - 300) childhood asthma  
465 incidence and 130 (95% CI: 50 – 120) respiratory hospitalizations were also found, for  
466 combinations of PM<sub>2.5</sub> and NO<sub>2</sub>, and PM<sub>2.5</sub> and ozone, respectively.

467

468 A recent paper (Buonocore et al., 2023) using a similar framework and data inputs showed that  
469 the health burden of oil & gas as a whole is 7,500 (95% CI: 4,500 – 12,000) deaths, 410,000  
470 (95% CI: 9,200 – 810,000) childhood asthma exacerbations, and 2,200 (95% CI: 830 - 3,200)  
471 childhood asthma incidence. Comparing these two studies indicates that flaring and venting  
472 contributes just under 10% of the mortality cases and incident asthma cases from O&G  
473 production as a whole, and around 5.4% of the asthma exacerbations from O&G production. FV  
474 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.  
475 The relative proportions that NO<sub>2</sub> from flaring contributes to total sector deaths and asthma  
476 exacerbations (Table S11) indicates the strong role of NO<sub>2</sub> in driving total health impacts.

477

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

484

485 Figure 2 presents spatial distribution of FV air pollution-attributable deaths in 2016. Texas  
486 observed the highest FV-attributed premature deaths at 133 incidences, of which 76, 51 and 6  
487 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)  
488 highest premature deaths are observed in Pennsylvania and Colorado, respectively. The top three  
489 numbers of FV-attributed asthma incidences by state are also observed in Texas (14,935),  
490 Colorado (13,748) and Pennsylvania (11,184).

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

495

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

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

510  
511 Table 2 shows that the monetized values of the health impacts due to FV total \$7.4B, while  
512 Buonocore et al (2023) reported \$77B from the entire OG sector, indicating a rather significant  
513 (~10%) contribution of FV to the overall monetized health risk from the OG sector. Industry  
514 analysis (Rystad 2022) indicates that solutions for operators to address flaring and capture this  
515 otherwise wasted gas are readily available and cost-effective, and even potentially profitable.  
516 The results of this study reveal the near-term air quality health benefits from addressing flaring  
517 and venting emissions. Since ~90% of the health impacts of O&G production originate from  
518 outside flaring, and since NO<sub>2</sub> has such a strong role, this indicates that health benefits of  
519 emission control strategies can be increased by expanding coverage to more NO<sub>x</sub>-rich subsectors  
520 of the O&G production sector such as compressors and pumpjack engines, well drilling and  
521 completions, in addition to the solutions for reducing FV emissions.

522  
523

#### 524 **Limitations**

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

535  
536 While the health impact modeling uses the best background health data available, some  
537 outcomes only have data at state or national resolution – missing potential hotspots for diseases,  
538 most notably asthma. Additionally, since the health modeling framework exclusively captures  
539 health impacts due to exposure to the three pollutants ozone, PM<sub>2.5</sub>, and NO<sub>2</sub>, this model is  
540 unlikely to fully capture many of the complex, multifactorial impacts occurring in communities  
541 hosting O&G production. These health outcomes include but are not limited to adverse birth  
542 outcomes (Willis et al., 2022), asthma (Willis et al., 2018), and childhood leukemia (Clark et al.,  
543 2022).

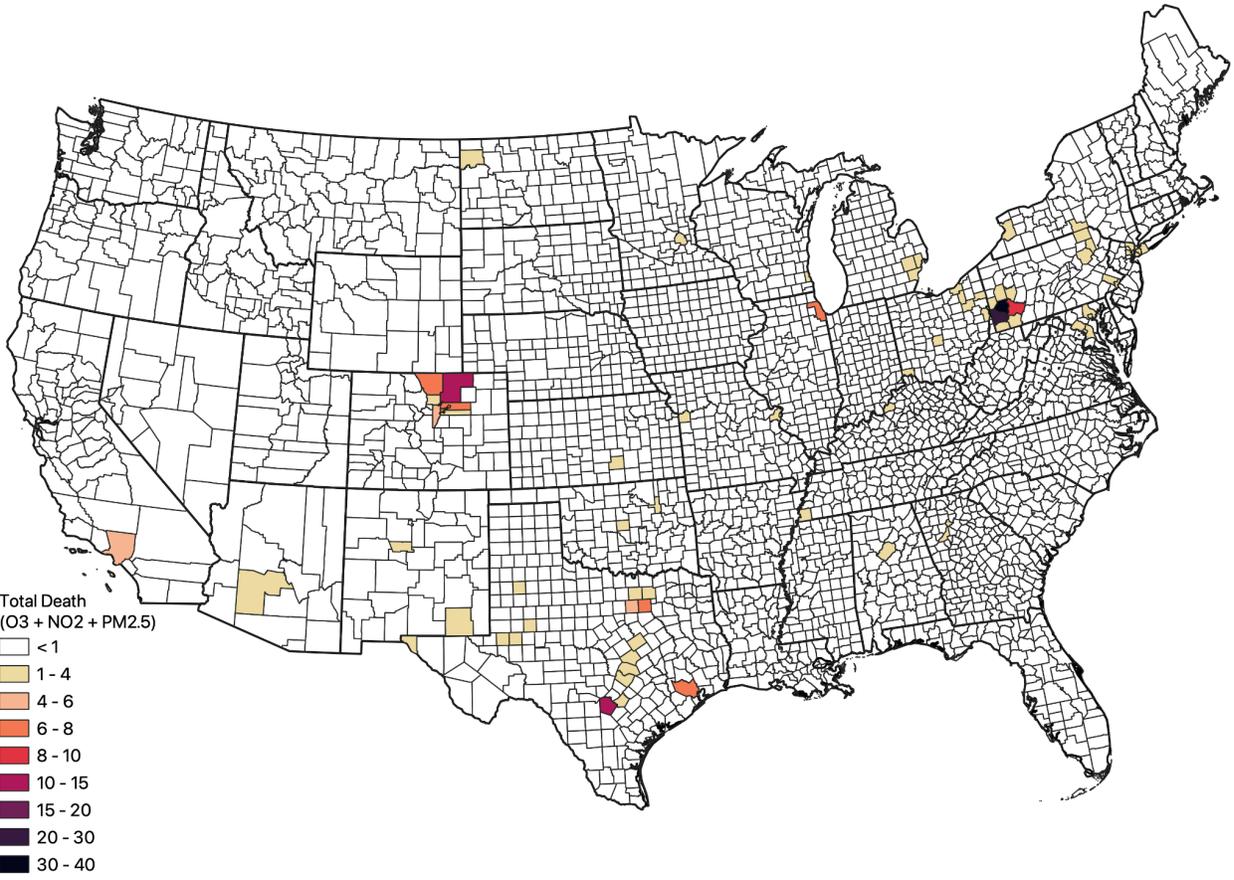
544  
545 The environmental justice analysis that we conducted was not comprehensive of all Justice40  
546 indicators that were available. Future research could consider including other populations at risk  
547 of environmental exposures as defined by CJEST (Council on Environmental Quality, 2022).  
548 Furthermore, the U.S. Census Bureau American Community Survey (ACA) data underreports

549 the true number of Native people living in the U.S. (Smithsonian, 2023), and the undercounting  
550 and misclassification of Native peoples in health databases (Jim et al., 2014; Stehr-Green et al.,  
551 2002) along with existing cultural, structural, and social barriers to health care (USCCR, 2004)  
552 contribute to the true representation of Native persons and their health status (Adakai et al.,  
553 2018). Having access to the true number of Native peoples and fairly represented mortality and  
554 asthma outcomes could impact our IRR calculations.

555  
556 Despite these limitations, this study still produces a robust estimate of health impacts of flaring  
557 by using the best data available for emissions from flaring. This study also uses a novel health  
558 impact assessment framework that includes health impacts of direct NO<sub>2</sub> exposure, and health  
559 outcomes not regularly included in air pollution health impact assessments, most notably birth  
560 and children's health outcomes.  
561

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*

566 **Conclusions**

567 Combining satellite-based observation of flare activities in O&G activities and new set of  
568 emission factors resulted in higher emissions from FV in the U.S. than what were reported in the  
569 EPA's NEI. Impacts of FV on air quality are most noticeable and significant within FV-major  
570 areas and its immediate downwind regions, even though domain-wide averages are relatively  
571 small compared to the overall OG impacts. However, FV sector still contributes up to about  
572 \$7.4B (~10%) of the total burden of health risk from O&G, and thus highlighting the potential  
573 need to focus on them for protecting public health. Of the total 710 premature deaths estimated  
574 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  
575 while most health impact studies so far have been focused on PM<sub>2.5</sub>, health impact from O<sub>3</sub> &  
576 NO<sub>2</sub> should not be overlooked. Findings from this study suggest controlling emissions from  
577 flaring and venting from O&G production, besides being cost effective and profitable to the  
578 operators, additionally provides an opportunity for yielding significant public health benefits.  
579  
580

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585 quality scientists across multiple state, local, tribal, and regional agencies, and EPA and Federal  
586 Land Management agencies who were involved in the development of the national inventories.  
587 We also acknowledge the U.S. EPA for providing meteorological inputs from WRF and outputs  
588 from the CMAQ hemispherical scale simulations for extracting initial and boundary conditions  
589 used in this study.

591 **Conflict of Interest**

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

594 **Open research (availability statement)**

595 Estimates of flared gas volume from the Visible Infrared Imaging Radiometer Suite - VIIRS for  
596 2019 and other years are publicly available at  
597 [https://eogdata.mines.edu/products/vnf/global\\_gas\\_flare.html#data\\_download](https://eogdata.mines.edu/products/vnf/global_gas_flare.html#data_download)

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

602 Flare stack parameters were estimated with reanalysis II wind data from NOAA Physical  
603 Sciences Laboratory, publicly available at  
604 <https://psl.noaa.gov/data/gridded/data.ncep.reanalysis2.html>

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

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

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

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

614 Environmental justice analysis was performed with data from the Climate and Economic Justice  
615 Screen Tool – CJEST, publicly available at [https://screeningtool.geoplatform.gov/en/#3/33.47/-](https://screeningtool.geoplatform.gov/en/#3/33.47/-97.5)  
616 [97.5](https://screeningtool.geoplatform.gov/en/#3/33.47/-97.5)

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