

30 and hazard index (HI) decreased from 4.89×10^{-7} and 5.90×10^{-3} and 4.51×10^{-7} and 5.40×10^{-3} ,
31 respectively. China and India showed the higher health benefits due to benzene pollution mitigation
32 compared with other countries, highlighting the importance of benzene emission reduction.

33 **1. Introduction**

34 Volatile organic compounds (VOCs) are an important class of organic pollutants in the urban
35 air and have aroused great attention (Kamal et al., 2016; Koppmann, 2008; Mozaffar and Zhang,
36 2020). As one of the typical toxic VOC species, benzene poses a variety of negative impacts on
37 human health including respiratory irritation, asthma, and allergies (Cui et al., 2019; Kim et al.,
38 2013; Tang et al., 2007). Moreover, benzene has high chemical reactivity, and could participate in
39 photochemical reactions in the atmosphere, thereby leading to the formation of secondary organic
40 aerosols (SOA) and ozone (Dumanoglu et al., 2014; Hsu et al., 2018; Li et al., 2019). Given the high
41 toxicity to human health and tremendous harm to air quality (Dumanoglu et al., 2014; Lu et al.,
42 2020), it is highly imperative to decrease the ambient benzene concentration. It was well
43 documented that ambient benzene mainly originated from anthropogenic emission (Mozaffar and
44 Zhang, 2020; Pakkattil et al., 2021). Therefore, understanding the response of ambient benzene to
45 anthropogenic emission was favorable to evaluate the effectiveness of abatement strategies and
46 inform policy decisions.

47 Recently, the ongoing global outbreak COVID-19 has resulted in paroxysmal public health
48 responses including travel restrictions, lockdown, curfews, and quarantines around the world. These
49 drastic lockdown measures inevitably triggered sweeping disruptions of social and economic
50 activities, and further affected the emissions and concentrations of some air pollutants (Bauwens et
51 al., 2020; Berg et al., 2021; Doumbia et al., 2021; Zheng et al., 2021b). The unexpected public health
52 emergency provided us an unprecedented chance to assess the response of air pollutants to emission
53 reduction. Bauwens et al. (2020) has observed that the average NO_2 column in China during
54 January-April 2020 decreased by about 40% relative to the same period in 2019 due to the dramatic
55 decreases of NO_x emissions. Later on, Keller et al. (2021) has analyzed the impact of COVID-19
56 lockdown on global NO_2 concentrations and found that the surface NO_2 concentrations were 18%
57 lower than business as usual from February 2020 onward. In addition, Hammer et al. (2021)
58 estimated that population-weighted mean $\text{PM}_{2.5}$ concentrations in China, Europe, and North

59 America experienced changes of -11 to -15, -2 to 1, and -2 to 1 $\mu\text{g}/\text{m}^3$ during COVID-19 lockdown
60 period, respectively. Compared with NO_2 and $\text{PM}_{2.5}$, ambient SO_2 levels in China (-4.6%) and India
61 (-14%) did not experience marked variations after lockdown (Zhang et al., 2021; Zhao et al., 2020).
62 It should be noted that the global O_3 concentration even increased up to 50% during this period
63 (Keller et al., 2021). To date, most of the current studies focused on regional or global criteria
64 pollutant (e.g., $\text{PM}_{2.5}$, NO_2 , and O_3) concentration changes after the COVID-19 outbreak, while few
65 studies assessed the impact of COVID-19 lockdown on ambient benzene levels.

66 Currently, only several studies assessed the impact of COVID-19 lockdown on atmospheric
67 benzene level. Mor et al. (2021) observed that the atmospheric benzene level in Chandigarh, India
68 decreased by 27% during COVID-19 period. Afterwards, Pakkattil et al. (2021) demonstrated that
69 the ambient benzene levels in Delhi (-93%) and Mumbai (-72%) have suffered from drastic
70 decreases after COVID-19 lockdown. In China, Pei et al. (2022) revealed the VOC concentration in
71 Pearl River Delta (PRD) decreased by 19% and the decrease rate of ambient benzene reached ~40%
72 after lockdown. In Europe, Cai et al. (2022) revealed that the ambient benzene level in Orléans even
73 slightly increased after lockdown, which might be associated with the unfavorable meteorological
74 conditions. Although the ground-level measurement could reflect the regional ambient benzene
75 changes during COVID-19 lockdown period to some extents, few regions, especially in developing
76 countries, have collected sufficient observations for ambient benzene exposure assessment (Geddes
77 et al., 2016; Van Donkelaar et al., 2015). Moreover, the limited monitoring sites around the world
78 cannot accurately reflect the global benzene pollution because of large spatial gaps and restricted
79 spatial representativeness of these ground-based sites (Shi et al., 2018). The health effect assessment
80 based on these scarce sites alone inevitably increased the probability of exposure misclassification
81 (Ling and Li, 2021). Fortunately, chemical transport models (CTMs) gave us an unparalleled chance
82 to capture the full-coverage ambient benzene level at the global scale. Although CTMs generally
83 showed various biases owing to high uncertainties in initial conditions, input variables, and
84 parameterizations (Ivatt and Evans, 2020), the machine-learning bias-correction method could
85 significantly reduce bias in air quality models (Bocquet et al., 2015). Up to date, some studies have
86 developed multiple machine-learning models to estimate the concentrations of $\text{PM}_{2.5}$ (Wei et al.,
87 2021; Wei et al., 2020), NO_2 (Wei et al., 2023), and O_3 (Wei et al., 2022) around the world.

88 Unfortunately, no study employed the ensemble technique to analyze the change of global ambient
89 benzene after COVID-19 outbreak. Besides, nearly all of the current studies only used original
90 observation data to assess the impact of COVID-19 lockdown on ambient benzene level (Pakkattil
91 et al., 2021). Actually, the concentrations of air pollutants were not only controlled by emission, but
92 also modulated by complex meteorological conditions (Hammer et al., 2021). For instance, some
93 pioneering studies have revealed that several severe haze episodes still occurred even with the strict
94 restrictions put in place in China (Chang et al., 2020; Huang et al., 2021). Hence, it is necessary to
95 remove the effects of meteorological parameters and then to further quantify the isolated
96 contribution of emission reduction to global ambient benzene level and health risks during COVID-
97 19 lockdown period.

98 In our study, the machine-learning model coupled with CTMs was applied to estimate the
99 global ambient benzene concentrations from 23 January to 30 June in 2019 and 2020. At first, the
100 CTMs output, emission inventory, meteorological parameters, and many other geographical
101 covariates were integrated into the ensemble decision tree model to obtain global full-coverage
102 benzene concentrations in the atmosphere. Then, we also examined the synergetic impacts from the
103 anthropogenic emissions and meteorological factors during the pre-lockdown and lockdown periods.
104 Finally, we estimated the emission-induced benzene concentrations before and after COVID-19
105 lockdown and quantified the benzene-related health benefits due to COVID-19 lockdown in major
106 regions around the world. This study shows important implications for developing control strategies
107 to alleviate global atmospheric benzene pollution.

108 **2. Data and methods**

109 2.1 Data preparation

110 2.1.1 Ground-level benzene observation

111 Our analysis was performed based on the recent development of unprecedented public access to
112 ground-level air quality observations. In our study, we collected an air quality dataset of hourly
113 surface benzene observations at 669 sites at the global scale during 23 January-30 June in 2019 and
114 2020 (Figure S1). The start date of COVID-19 lockdown in China was January 23th and the national
115 lockdown lasted for about one month. However, the deblocking date in Wuhan was April 8th. The
116 start and end dates of lockdown in India were March 25th and April 25th, respectively. The

117 lockdown in the United States occurred firstly in California in March 19th, and then the lockdown
118 lasted for 1-2 months. The lockdown dates in most European countries lasted from March to May.
119 The detailed spatial distribution of these sites in India, Europe, and the United States are depicted
120 in Figure S1. The surface benzene dataset in India was downloaded from the Central Pollution
121 Control Board (CPCB) online database, which has been widely utilized in previous studies (Mahato
122 et al., 2020; Mor et al., 2021; Sharma et al., 2020). The CPCB database provides data quality
123 assurance (QA) or quality control (QC) programs by developing strict procedures for sampling,
124 analysis, and calibration (Gurjar et al., 2016). The ground-level benzene observations in Europe and
125 the United States were collected from air quality data portal of the European Environment Agency
126 (EEA) and United States Environmental Protection Agency (EPA), respectively. Only days with
127 more than 12 h of available data are included in the analysis. All of the hourly data was average to
128 the daily scale.

129 2.1.2 Independent variables

130 The daily benzene concentrations at global scale were simulated using GEOS-Chem model
131 (v12-01), which included the full gaseous HO_x-O_x-NO_x-CO-NMVOC chemistry and online aerosol
132 calculations. The simulation used assimilated meteorological observations (GEOS MERRA-2) at 2°
133 x 2.5° horizontal resolution with 72 vertical levels for the year 2019 and 2020. The anthropogenic
134 emission inventory in 2019 was collected from Community Emissions Data System (CEDS). Then,
135 the emission inventory in 2020 was calculated based on that in 2019 and updated adjustment factor
136 proposed by Doumbia et al. (2021).

137 The meteorological parameters were obtained from the NASA Goddard Earth Observing
138 System Composition Forecast (GEOS-CF) model (Keller et al., 2021b). GEOS-CF integrates the
139 GEOS-Chem atmospheric chemistry model into the GEOS Earth System Model (Hu et al., 2018;
140 Long et al., 2015) and provides global hourly analyses of meteorological variables at 0.25° spatial
141 resolution (Keller et al., 2021b). Meteorological parameters including surface pressure (PS), relative
142 humidity (RH), 2-m air temperature (T2M), total precipitation (TPREC), 10-m latitudinal wind
143 component (U10M), 10-m longitudinal wind component (V10M), and boundary layer height (BLH)
144 obtained from GEOS-CF were used to develop the model (Figure S2). In addition, cropland, forest,
145 grassland, shrubland, and barren land also have been integrated into the final model (Liu et al., 2020).

146 All of the independent variables collected from multiple sources were regridded to 0.25° grids
 147 using spatial interpolation algorithms. During the process of model development, the most important
 148 procedure was to remove some redundant explanatory variables and then to determine the optimal
 149 variable group. The basic principle of the variable selection was to eliminate the less important
 150 predictors. These variables generally suggested that the R² value of the submodel did not experience
 151 a significant decrease or even suffered from a slight increase when these redundant ones were
 152 removed from the model. At last, a total of 64001 samples and 7 variables were utilized to predict
 153 the ambient benzene concentrations at the global scale.

154 2.2 Model development

155 2.2.1 The ensemble model development for atmospheric benzene estimates

156 In the pioneering studies, random forest (RF), extreme gradient boosting (XGBoost), and light
 157 gradient boosting machine (LightGBM) exhibited the better estimation accuracy (Li et al., 2021).
 158 RF model holds a great deal of decision trees, and each one experiences an independent sampling
 159 procedure and all of these trees show the same distributions (Breiman). RF model often displays
 160 excellent prediction performance owing to the injected randomness. The model accuracy is strongly
 161 dependent on the number of trees, splitting features, and the variable group. The detailed procedures
 162 are summarized as follows:

$$163 \quad f(x) = \sum_{z=1}^Z c_z R(x \in Q_z) \quad (1)$$

$$164 \quad \overset{\Delta}{c}_z = \text{average}(y_i | x_i \in Q_z) \quad (2)$$

$$165 \quad BR_1(p, q) = \{X | X_j \leq q\} \& BR_2(p, q) = \{X | X_j > q\} \quad (3)$$

$$166 \quad \min_{p,q} \left[\min_{M_1(p,q)} \sum (y - c_1)^2 + \min_{M_2(p,q)} \sum (y - c_2)^2 \right] \quad (4)$$

$$167 \quad \overset{\Delta}{c}_1 = \text{average}(y_i | x_i \in Q_1(p, q)) \& \overset{\Delta}{c}_2 = \text{average}(y_i | x_i \in Q_2(p, q)) \quad (5)$$

168 where (x_i, y_i) is the sample for i = 1, 2, ..., N in Q regions (Q₁, Q₂, ..., Q_Z); R denotes the weight
 169 of each branch; BR represents decision tree branch; c_m is the response to the model; $\overset{\Delta}{c}_z$ represents
 170 the optimal value, p is the feature variable; c₁ is the average of left branch; c₂ is the average of right

171 branch; q represents the split point.

172 XGBoost model is an improved algorithm of gradient boosting decision tree (GBDT) model and
173 loss functions have been extended to the second order function. The detailed XGBoost algorithm is
174 shown as the following formula (Zhai and Chen, 2018):

$$175 \quad Y^{(t)} = \sum_{i=1}^n [l(y_i, \hat{y}^{(t-1)}) + \partial_{y^{(t-1)}} l(y_i, \hat{y}^{(t-1)}) f_t(x_i) + \frac{1}{2} \partial_{y^{(t-1)}}^2 l(y_i, \hat{y}^{(t-1)}) f_t^2(x_i)] + \varepsilon(f_t) \quad (6)$$

176 where $Y^{(t)}$ is the cost function at the t -th period; ∂ represents the derivative of the original function;
177 $\partial_{y^{(t-1)}}^2$ is the second derivative of the original function; l is the differentiable convex loss function
178 that reflects the minus of the predicted value (\hat{y}) of the i -th instance at the t -th period and the target
179 value (y_i); $f_t(x)$ represents the increment; $\varepsilon(f_t)$ is the regularizer.

180 LightGBM model is an update version of XGBoost method, and significantly improve the
181 running speed of modelling process. Moreover, this method could decrease the cache miss by a large
182 margin and further improved the predictive accuracy. The detailed algorithms are as follows (Sun
183 et al., 2020):

$$184 \quad \hat{f} = \arg \min_f L(y, f(x)) \quad (7)$$

$$185 \quad f_T(X) = \sum_{t=1}^T f_t(X) \quad (8)$$

$$186 \quad \Gamma_t = \sum_{i=1}^n (g_i f_t(x_i) + \frac{1}{2} h_i f_t^2(x_i)) \quad (9)$$

187 where \hat{f} is the least value of cost function; $L(y, f(x))$ is the cost function; $f_T(X)$ denotes the total
188 regression trees; $f_t(X)$ represents each regression tree; g_i and h_i represent the first- and second-order
189 gradient statistics of the cost function, respectively.

190 Although all of these models showed the better performance in predicting air pollutants, nearly
191 all of these submodels still suffered from some weaknesses in the prediction accuracy. Hence, it was
192 necessary to collocate these models using back-propagation neural network (BPNN) to further
193 simulate daily ambient benzene concentrations at the global scale. As depicted in Figure 1, three
194 submodels including RF, XGBoost, and LightGBM were stacked through BPNN model to simulate
195 the daily atmospheric benzene levels at the global scale. Firstly, a 5-fold cross-validation method

196 was utilized to train each submodel to determine the optimal hyperparameter. Then, the BPNN
197 method was employed to further train the estimated concentrations of three submodels against the
198 observations (Figure 1). Lastly, the global ambient benzene concentrations were predicted on the
199 basis of the ensemble model.

200 2.2.2 The meteorology-normalized benzene estimates

201 The ambient benzene concentration was influenced by both of meteorological parameters and
202 emissions. To isolate the contribution of emission, the impacts of meteorological conditions should
203 be removed. In our study, the XGBoost approach was utilized to eliminate the impacts of
204 meteorological conditions. The simulated benzene concentration in each grid (0.25°) based on the
205 method in section 2.2.1 was treated as the dependent variable. The daily benzene emission,
206 meteorological factors, month of year (MOY), and day of year (DOY) in each grid were regarded
207 as the explanatory variables. The raw dataset was randomly classified into a training dataset (90%
208 of input dataset) for developing the XGBoost model and the remained samples were regarded as the
209 test dataset. After the development of the XGBoost model, the weather normalized technique was
210 employed to predict the ambient benzene concentration at a specific time point. The detailed
211 deweathered algorithms was introduced by Grange and Carslaw (2019) firstly. The meteorology-
212 normalized benzene level served as the concentrations contributed by emission alone. The
213 differences of total and deweathered benzene concentrations were regarded as the concentration
214 contributed by meteorology. In addition, the CV R^2 value of model using for the separation of
215 meteorology and emission also should be higher than 0.50, otherwise the model could be considered
216 to be unreliable.

217 2.3 Health effect assessment

218 In our study, the carcinogenic and non-carcinogenic risks of ambient benzene were assessed
219 based on the standard methodology of United States Environmental Protection Agency (USEPA).
220 The carcinogenic and non-carcinogenic risks induced by benzene exposure for were evaluated based
221 on the lifetime carcinogenic risks (LCR) and hazard index (HI). The formulas for calculating
222 benzene intake (BI), LCR, and HI are as follows (Table S1):

$$223 \quad BI=(C \times ET \times EF \times ED) / (365 \times 24 \times AT) \quad (10)$$

$$224 \quad HI=BI / RfC \quad (11)$$

225
$$\text{LCR} = \text{BI} \times \text{IUR} \quad (12)$$

226 where C ($\mu\text{g}/\text{m}^3$) denotes the concentration of the corresponding ambient benzene; ET is the
227 exposure time; EF represents the annual exposure frequency (d a^{-1}); ED is the exposure duration
228 (a); AT_{nca} and AT_{ca} denotes the average exposure time for carcinogenic and non-carcinogenic risks
229 (a), respectively. BI means the benzene intake; RfC represents the reference dose ($\mu\text{g}/\text{m}^3$); IUR is
230 the inhalation risk ($1/\mu\text{g}/\text{m}^3$). The non-carcinogenic risk of the ambient benzene is considered to be
231 high when HI was above 1.0, whereas the health risk is not obvious when HI is below 1.0. The
232 carcinogenic risk was regarded as definite risk when LCR was higher than 1×10^{-4} , while it was
233 treated as the possible risk when this indicator was located between 1×10^{-6} and 1×10^{-4} . The risk
234 was treated as negligible when the indicator was lower than 1×10^{-6} (Dumanoglu et al., 2014; Li et
235 al., 2017).

236 3. Results and discussion

237 3.1 The model fitting and validation

238 The ensemble model was utilized to estimate the ambient benzene concentrations at the global scale
239 during 23 January-30 June in 2019 and 2020. The cross-validation (CV) R^2 value of the ensemble
240 model ($R^2 = 0.60$) was significantly higher than that of RF (0.52), XGBoost (0.53), and LightGBM
241 (0.55) (Figure S3). Nevertheless, both of the root-mean-square error (RMSE) ($1.18 \mu\text{g}/\text{m}^3$) and the
242 mean absolute error (MAE) ($0.59 \mu\text{g}/\text{m}^3$) of the ensemble model were significantly lower than those
243 of RF (RMSE and MAE: 1.41 and $0.72 \mu\text{g}/\text{m}^3$), XGBoost (RMSE and MAE: 1.37 and $0.70 \mu\text{g}/\text{m}^3$),
244 and LightGBM (RMSE and MAE: 1.34 and $0.69 \mu\text{g}/\text{m}^3$). The higher R^2 value and the lower RMSE
245 and MAE suggested the higher accuracy of the ensemble model in air quality simulation. In the
246 pioneering studies, Wolpert (1992) confirmed that the joint use of multiple statistical models could
247 decrease the probability of overfitting and strengthen the predictive accuracy and transferability of
248 final models. Besides, our previous studies also demonstrated that the stacking of various decision
249 tree models could significantly outperform individual model because each decision tree model could
250 suffer from some weaknesses (Li et al., 2021). For instance, the dataset in the RF model appeared
251 to be over-fitted when much noise existed in the training data of regression problems (Breiman,
252 2001). Besides, RF model might underestimated/overestimated the extremely values of ambient
253 benzene (Xue et al., 2019), which could be neutralized by the XGBoost algorithm through the

254 boosting method (Li et al., 2020). For XGBoost algorithm, excessive leaf nodes often showed low
255 splitting gain, while the LightGBM model could make up this defect (Nemeth et al., 2019). Overall,
256 the combination of these decision tree models could overcome these weaknesses of these individual
257 models and enhance the robustness of the final model.

258 Although 10-fold CV has verified that the modelling performance of ensemble model was
259 superior to the individual models, this method cannot examine the spatial transferability of this
260 model. In our study, many regions except India, Europe, and the United States were lack of
261 monitoring sites of ambient benzene. Fortunately, the CTMs output provided a strong proxy to
262 predict the daily ambient benzene concentrations before and after COVID-19 outbreak. In order to
263 examination the spatial transferability of the ensemble model, the cross validation was performed.
264 In each round, two-thirds of the benzene dataset in India, Europe, and the United States were applied
265 to train the model and the remained one was utilized to examine the model (e.g., India+Europe for
266 training and the United States for testing). After three rounds, all of the simulated benzene
267 concentrations were compared with the corresponding observed values. As shown in Figure S4, the
268 out-of-bag R^2 value reached 0.58, which was slightly lower than the R^2 value (0.60) of training
269 model. In addition, RMSE and MAE of the fitting equation for the out-of-bag data were 1.18 and
270 0.62, respectively. The result was in good agreement with those based on CV database, indicating
271 the ensemble model showed satisfied spatial generalization.

272 The ensemble model can capture the spatiotemporal variation of ambient benzene during
273 COVID-19 lockdown period, while the impact of COVID-19 lockdown cannot be quantified
274 because the contribution of meteorological parameters cannot be removed based on this model alone.
275 Therefore, it is proposed to employ the XGBoost algorithm to isolate the contribution of emission
276 reduction to global atmospheric benzene. As depicted in Figure S5, the CV R^2 value and slope of
277 fitting curve reached 0.65 and 0.62, respectively. The result suggested that meteorology-normalized
278 model was robust because the CV R^2 value was much higher than 0.50.

279 3.2 The impact of COVID-19 lockdown on global atmospheric benzene level

280 The ensemble model was developed to expand the ground-observed benzene measurement to
281 the global scale and capture the global spatial variability of ambient benzene. As shown in Figure
282 S6, the global simulated (total) benzene concentration during Jan. 23-Jun. 30 in 2019 and 2020

283 ranged from 0.52 to 6.36 $\mu\text{g}/\text{m}^3$, with the average value of $0.92 \pm 0.23 \mu\text{g}/\text{m}^3$. At the regional scale,
284 the benzene concentration displayed significantly spatial variability. The benzene concentration
285 followed the order of India ($1.44 \pm 0.14 \mu\text{g}/\text{m}^3$) > China ($1.17 \pm 0.13 \mu\text{g}/\text{m}^3$) > Europe (1.02 ± 0.08
286 $\mu\text{g}/\text{m}^3$) > United States ($0.96 \pm 0.09 \mu\text{g}/\text{m}^3$) during Jan. 23-Jun. 30 in 2019 and 2020. Besides, the
287 global simulated mean benzene level suffered from slight decrease from 0.93 ± 0.06 in 2020 to 0.90
288 ± 0.06 in 2019. However, the inter-annual variation of ambient benzene exhibited remarkable spatial
289 discrepancy at the global scale. As depicted in Figure S7, the change ratio of simulated (total)
290 benzene level during the COVID-19 lockdown period (the difference of the benzene level before
291 COVID-19 lockdown and that during COVID-19 lockdown period) in 2020 was in the order of
292 India (-18.5%) > Europe (-16.7%) > China (-11.7%) > United States (-11.5%). Compared with 2020,
293 the change ratio of benzene level during the same period in 2019 followed the order of India (-
294 16.3%) > Europe (-6.62%) > United States (-6.46%) > China (-4.18%). It should be noted that the
295 simulated ambient benzene concentration suffered from the higher decreasing ratio in 2020
296 compared with the same period in 2019 in nearly all of the major countries around the world, which
297 might be associated with the local COVID-19 lockdown measures in 2020.

298 Due to the interference of meteorological conditions, we cannot quantify the direct impact of
299 COVID-19 lockdown on ambient benzene based on the comparison of simulated (total) benzene
300 levels. Thus, the meteorology-normalized method was employed to decouple the separated
301 contributions of emission reduction and meteorology to ambient benzene. In our study, both of the
302 change ratio and detrended change ratio were applied to evaluate the impact of COVID-19
303 lockdown on global ambient benzene level. The change ratio represents the variation of ambient
304 benzene level during lockdown period in 2020 compared with pre-lockdown period in 2020.
305 However, the detrended change ratio reflects the difference of the change ratio in 2020 and the
306 change ratio during the same period in 2019, which could avoid the inter-annual system error and
307 contingency of a single year. As summarized in Figure 2 and 3, the change ratio of deweathered
308 benzene concentration from pre-lockdown to lockdown period in 2020 was in the order of India (-
309 23.6%) > Europe (-21.9%) > United States (-16.2%) > China (-15.6%). Meanwhile, the change ratio
310 of deweathered benzene concentration during the same time in 2019 followed the order of Europe
311 (-10.2%) > United States (-8.04%) > India (-7.40%) > China (-2.31%). The large gap in the change

312 ratio of deweathered benzene level between 2019 and 2020 confirmed that the drastic and
313 consequential quarantines significantly decreased the ambient benzene concentrations in nearly all
314 of the regions with lockdown measures. Among all of the major countries, India suffered from the
315 most dramatic benzene decrease during 24 March 2020-24 April 2020 (-23.6%) compared with the
316 same period in 2019 (-7.4%). During this period, the prohibition of industrial activities and mass
317 transportation was proposed to curb the spread of COVID-19 pandemic, leading to the tremendous
318 reduction of anthropogenic benzene emission (Pathakoti et al., 2021; Zhang et al., 2021). Sahu et al.
319 (2022) revealed that the substantial increase of OH radical during COVID-19 period also facilitated
320 the ambient benzene removal due to the photooxidation reaction. The decrease ratio of deweathered
321 benzene level in India was close to that of PM_{2.5} (-26%), while it is was markedly lower than that
322 of NO₂ (-50%) (Zhang et al., 2021). Although both of Europe and the United States also performed
323 stringent lockdown restrictions in some regions such as Italy, Spain, and California (Guevara et al.,
324 2021a; Keller et al., 2021a), while the detrended change (P*: change ratio in 2020-change ratio in
325 2019) for deweathered benzene in Europe (P* = -13.9%) and the United States (P* = -6%) between
326 2020 and 2019 was still lower than that of India (P* = -16.2%) (Table 1). It was assumed that the
327 absolute concentration of ambient benzene in Europe and the United States were much lower than
328 that in India. It should be noted that the China displayed relatively gentle decreasing ratio (-15.6%)
329 after COVID-19 outbreak, which was even lower than the ratio in the United States. As the first
330 epidemic epicenter country, Chinese government imposed a rapid lockdown measure in Wuhan and
331 other cities across China in an effort to prevent the spread of the COVID-19 pandemic (Wu et al.,
332 2020). These restrictions interrupted a wide array of economic activities and reduced primary air
333 pollutant emissions, and thus resulted in the remarkable decreases of deweathered NO₂ (-43.6%)
334 and PM_{2.5} (-22%) (Dai et al., 2021). The gentle decreasing ratio of ambient benzene compared with
335 other pollutants might be linked with the source apportionment of atmospheric benzene. It was well
336 known that industrial source (e.g., chemical industry and solvent use) was major emission sector of
337 benzene (Li et al., 2019). Although the contribution from solvent use exhibited substantial decreases
338 in some cities (Qi et al., 2021; Wang et al., 2021), the chemical industry was not entirely interrupted
339 even during the COVID-19 lockdown period (Dai et al., 2021). Zheng et al. (2021a) also
340 demonstrated that the reduction of non-methane volatile organic compounds (NMVOCs) emission

341 from industry sector was much less than other pollutants.

342 Although the deweathered benzene concentrations in nearly all of the major countries
343 experienced obvious decreases during COVID-19 lockdown period, the change ratios of
344 deweathered benzene in different regions of these countries still showed large spatial variability. In
345 China, most of the cities in East China such as Beijing (-30.6%), Shanghai (-6.25%), and Wuhan (-
346 45.3%) experienced dramatic decreases of deweathered benzene levels (Figure S8), which was
347 mainly contributed by the simultaneous emission reduction of industry and transportation sectors.
348 Besides, enhanced atmospheric oxidation capacity could accelerate the benzene removal due to the
349 unbalanced decreases of VOC and NO_x emissions (Jensen et al., 2021). However, the deweathered
350 benzene concentrations in Northeast China and Yunnan province even exhibited slight increases
351 after COVID-19 outbreak (Figure 4). Dai et al. (2021) also found that the deweathered PM_{2.5}
352 concentration in Kunming increased ~20% after COVID-19 outbreak. At first, the contribution of
353 residential combustion source (62.1%) to atmospheric benzene in Yunnan province was higher than
354 other sectors (Guevara et al., 2021b; Kuenen et al., 2021). Moreover, the increase of domestic
355 emission due to home quarantine order further increased the ambient benzene concentration (10%)
356 in this province, which has been demonstrated by the updated emission inventory in 2020 (Dolumbia
357 et al., 2021). The slight increases of deweathered benzene levels in Northeast China after COVID-
358 19 outbreak could be linked with the earlier work
359 resumption(<https://baijiahao.baidu.com/s?id=1658138056285012986&wfr=spider&for=pc>). Based
360 on the simulation result, the deweathered ambient benzene level in Northeast China rebounded
361 sharply after the third week, and then returned back to normal in the late February. In India, the
362 decreasing ratios of deweathered benzene in Delhi, Mumbai, Kolkata, Bengaluru, Hyderabad,
363 Chennai, Ahmedabad, and Lucknow reached 21.6%, 20.9%, 73.7%, 26.9%, 38.0%, 33.7%, 25.1%,
364 and 33.3% during COVID-19 lockdown period (Figure S9), respectively. Among all of the major
365 cities in India, the ambient benzene level in Kolkata suffered from the most drastic decrease. It was
366 assumed that Kolkata is designated as dusty city and filled with vehicle emission. Fortunately, the
367 city experienced complete stop of vehicles movement, burning of biomass and dust particles from
368 the construction works, which were important sources for ambient benzene (Bera et al., 2021;
369 Kumar and and Singh, 2003). In Europe, the deweathered benzene levels in nearly all of the cities

370 displayed marked decreases because most European countries have imposed lockdowns to combat
371 the spread of the COVID-19 pandemic (Guevara et al., 2021a). For example, the private car use and
372 heavy good vehicles (HGVs) on the road in London reduced by 80% and 30-40%, respectively (Shi
373 et al., 2021). The drastic decrease of transportation emission triggered the P* value in London
374 between 2020 and 2019 reaching -43.6%. In the United States, the decreasing ratios of deweathered
375 benzene levels in the cities of Eastern United States and California were generally higher than those
376 in Central United States, which was in good agreement with the spatial variability of PM_{2.5} decrease
377 (Hammer et al., 2021). It was closely associated with the length of lockdown period
378 (https://en.wikipedia.org/wiki/COVID-19_lockdowns).

379 In addition, ambient benzene levels were also strongly affected by meteorological conditions
380 that alter photochemical production, advection, and depositional loss. Hence, we examined how
381 meteorological parameters influenced the temporal variability of ambient benzene during COVID-
382 19 lockdown period. In 2020, most of the major countries including China (3.9%), Europe (5.2%),
383 and the United States (4.7%) suffered from slight unfavorable meteorological conditions, which was
384 in good agreement with the impact of meteorological conditions on ambient NO₂ concentrations
385 (Shi et al., 2021). Among all the meteorological parameters, air temperature was the most important
386 factor for ambient benzene in nearly all of the regions around the world during the study period.
387 Compared with 2019, the air temperatures in China, India, Europe, and the United States increased
388 by 0.4, 0.9, 0.4, and 0.2°C during the same period in 2020, respectively. Jia and Xu (2014)
389 demonstrated that the increased air temperature generally suppressed the benzene photooxidation
390 and secondary organic aerosol (SOA) formation. Thus, the increased air temperature was not
391 beneficial to the further reduction of ambient benzene. Except air temperature, some other factors
392 such as RH, rainfall amount, and wind speed might affect the ambient benzene level. For instance,
393 the increased RH could be favorable to the benzene oxidation and the higher rainfall amount
394 promoted the benzene removal. However, in the machine learning model, the importance values of
395 these variables were much lower than that of air temperature. Overall, the result suggested that the
396 unfavorable meteorological conditions (air temperature) weakened the health benefits of ambient
397 benzene due to drastic lockdown measures around the world.

398 3.3 The effect of COVID-19 lockdown on global health risks

399 The global average LCR during 23 January-30 June in 2019 and 2020 were 4.89×10^{-7} and
400 4.51×10^{-7} after removing the contributions of meteorological conditions, respectively (Figure S10).
401 Although the COVID-19 lockdown decreased the LCR value slightly, both of the LCR values during
402 two periods were lower than the threshold level of 10^{-6} , suggesting that dwellings in most regions
403 could avoid carcinogenic risk through inhalation exposure to benzene (Li et al., 2017). However,
404 the LCR values showed significant spatial difference in different regions. For instance, North China
405 often suffered from the relatively higher benzene pollution, and the LCR value in this region
406 decreased from 1.03×10^{-6} (possible risk) during the study period in 2019 (the same period to 2020)
407 to 7.37×10^{-7} during COVID-19 lockdown period. The result verified that the stringent emission
408 control measures significantly decreased the health risk due to benzene exposure. The LCR value
409 across India only decreased from 6.55×10^{-7} to 6.42×10^{-7} during the study period, whereas the
410 northern part of India such as Bihar decreased from 1.14×10^{-6} to 1.09×10^{-6} due to the impact of
411 COVID-19 lockdown (Figure 5). As the most populous province of India, Bihar possessed more
412 than 124 million people (<http://kolkata.china-consulate.org/chn/lqgk/t1331638.htm>). The result
413 suggested that the COVID-19 lockdown certainly obtained remarkable short-term health benefits
414 through decreasing the ambient benzene exposure. The LCR values in Europe and the United States
415 decreased from 4.99×10^{-7} and 4.77×10^{-7} to 4.57×10^{-7} and 4.63×10^{-7} , respectively. Compared
416 with China and India, Europe and the United States suffered from relatively low carcinogenic risk
417 of benzene exposure even before COVID-19 lockdown. Although the COVID-19 lockdown further
418 decreased the LCR values in these regions, the overall carcinogenic risk was negligible.

419 Additionally, the non-carcinogenic risk around the world during the period was also assessed
420 based on HI. The average HI of ambient benzene exposure in China, India, Europe, and the United
421 States reduced from 8.92×10^{-3} , 7.45×10^{-3} , 6.32×10^{-3} , and 5.76×10^{-3} in 2019 to 8.53×10^{-3} ,
422 7.13×10^{-3} , 5.81×10^{-3} , and 5.59×10^{-3} during COVID-19 lockdown period in 2020 (Figure 6),
423 respectively. Although HI value in some regions including Bihar (1.52×10^{-2} to 1.41×10^{-2}) and
424 Uttar Pradesh (1.04×10^{-2} to 1.03×10^{-2}) in India and Beijing-Tianjin-Hebei (BTH) (1.25×10^{-2} to
425 1.14×10^{-2}) in China still experienced decreases during COVID-19 lockdown period, the HI values
426 in these regions were still significantly lower than the risk threshold (HI = 1). Therefore, the impact
427 of COVID-19 lockdown on non-carcinogenic risk of benzene exposure was insignificant.

428 **4. Conclusions and limitations**

429 The drastic lockdown measures largely reduced the air pollutant emissions. The meteorology-
430 normalized ambient benzene concentrations in China (-15.6%), India (-23.6%), Europe (-21.9%),
431 and the United States (-16.2%) experienced dramatic decreases after COVID-19 outbreak.
432 Furthermore, the decreasing ratios in these major regions during COVID-19 lockdown period were
433 much higher than the same period in 2019, indicating the aggressive emission control measures
434 efficiently decreased ambient benzene concentrations. Emission reductions from industrial activities
435 and transportation were major drivers for the decreasing of ambient benzene level during lockdown
436 period, while the relatively stable solvent use emission could restrict the further decrease of benzene
437 pollution. Besides, the slight increase of domestic emission during this period might be an important
438 reason for the benzene increase in some regions (e.g., Yunnan province). There is also an urgent
439 need to control the household combustion and solvent use emissions apart from the emissions from
440 industry and transportation sectors.

441 Besides, substantial decreases of atmospheric benzene levels could save sufficient health
442 benefits. Dramatic decreases of benzene emissions in Europe and the United States cannot save
443 effective health benefits because the ambient benzene levels in both of these regions during
444 business-as-usual scenario were significantly lower than the risk threshold. However, the benzene
445 decreases in North China Plain (NCP), China and Bihar, India could save abundant health benefits
446 because these regions often suffered from severe atmospheric benzene pollution during business-
447 as-usual scenario. Thus, more targeted abatement measures are needed to reduce the benzene
448 emission in these areas. For instance, the stricter industrial and vehicle emission standards for VOC
449 control should be implemented in China and India. Moreover, some measures including limiting the
450 amount of coal-fired power plants, adding environmentally friendly cars and clean fuels for vehicles
451 and vessels, and strengthening the labeling system for vehicles in use should be strengthened.

452 It should be noted that our study still suffered from some limitations. First of all, the monitoring
453 sites were not evenly distributed around the world, and thus the simulation result might show the
454 higher uncertainty in the regions lack of monitoring sites. Besides, the GEOS-Chem model still
455 suffered from some uncertainties due to imperfect chemical mechanism and inaccurate emission
456 inventory. In the future work, the model should be further improved.

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459 **Data availability**

460 The CEDS emission inventory are available at the website of
461 <https://zenodo.org/record/3754964#.YwrJL8jfmfU>. The global meteorological parameters
462 (reanalysis dataset) are obtained from the website of <http://geoschemdata.wustl.edu/ExtData/>.

463 **Author contributions**

464 LCH and LR wrote the manuscript. LR and CLL contributed to the conceptualization of the study.

465 LCH and LR conducted the research, and visualized the results. CLL revised the manuscript.

466 **Competing interests**

467 The contact author has declared that neither they nor their co-authors have any competing interests.

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Figure 1 The workflow of global atmospheric benzene modelling. CTM output represents the simulated benzene concentration based on GEOS-Chem model. Meteo denotes the meteorological parameters derived from GEOS-CF reanalysis. Emission represents the daily emission of benzene. MOY and DOY are the month of year and day of year, respectively. Simulated benzene represents the predicted benzene concentrations based on the ensemble model. Deweathered benzene denotes the benzene concentration after removing meteorological effects.

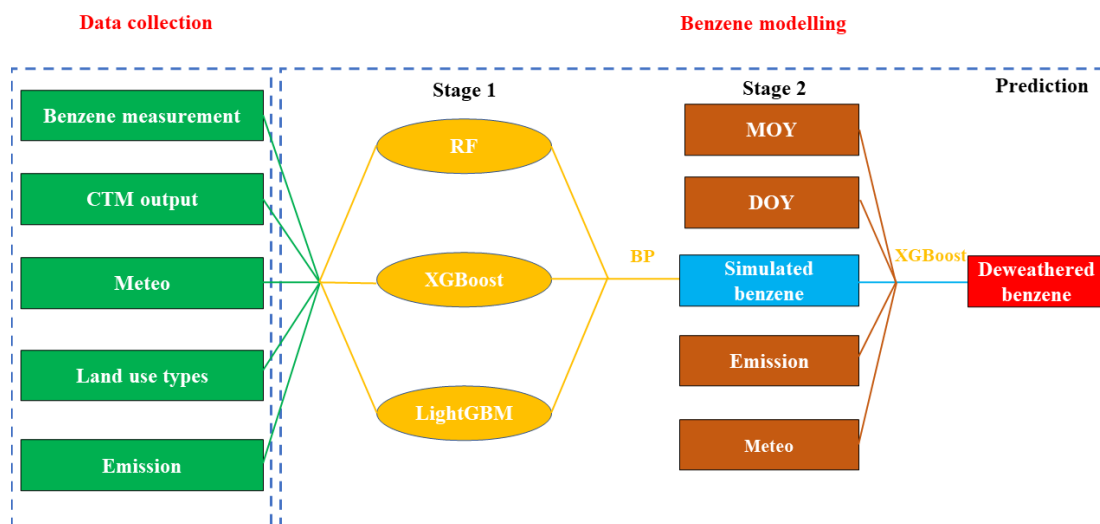


Figure 2 The global average deweathered benzene concentrations in 2019 (Jan. 23-Jun. 30) (a) and 2020 (Jan. 23-Jun. 30) (b). (c) represents the difference of deweathered benzene concentrations in 2020 and 2019 (Unit: $\mu\text{g}/\text{m}^3$).

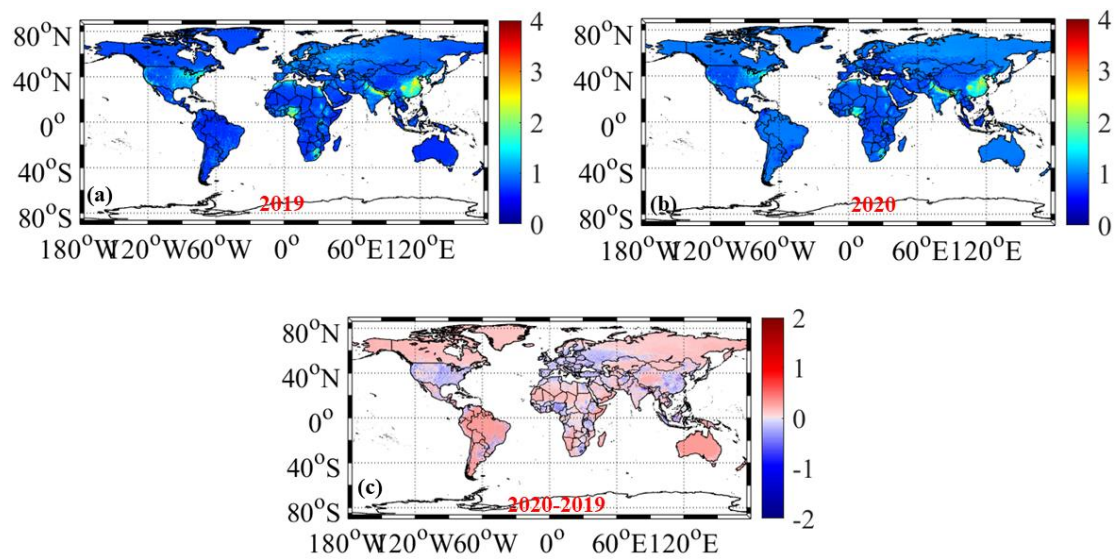


Figure 3 The weekly variations of atmospheric benzene concentrations ($\mu\text{g}/\text{m}^3$) in some major regions around the world during Jan. 23-Jun. 30. The red line and background denote mean values and standard deviation of deweathered weekly benzene concentrations in 2020. The cyan line and background denote mean values and standard deviation of deweathered weekly benzene levels in 2019. The dashed vertical red line suggests COVID-19 restriction dates, and the black line indicates the beginning of easing measures.

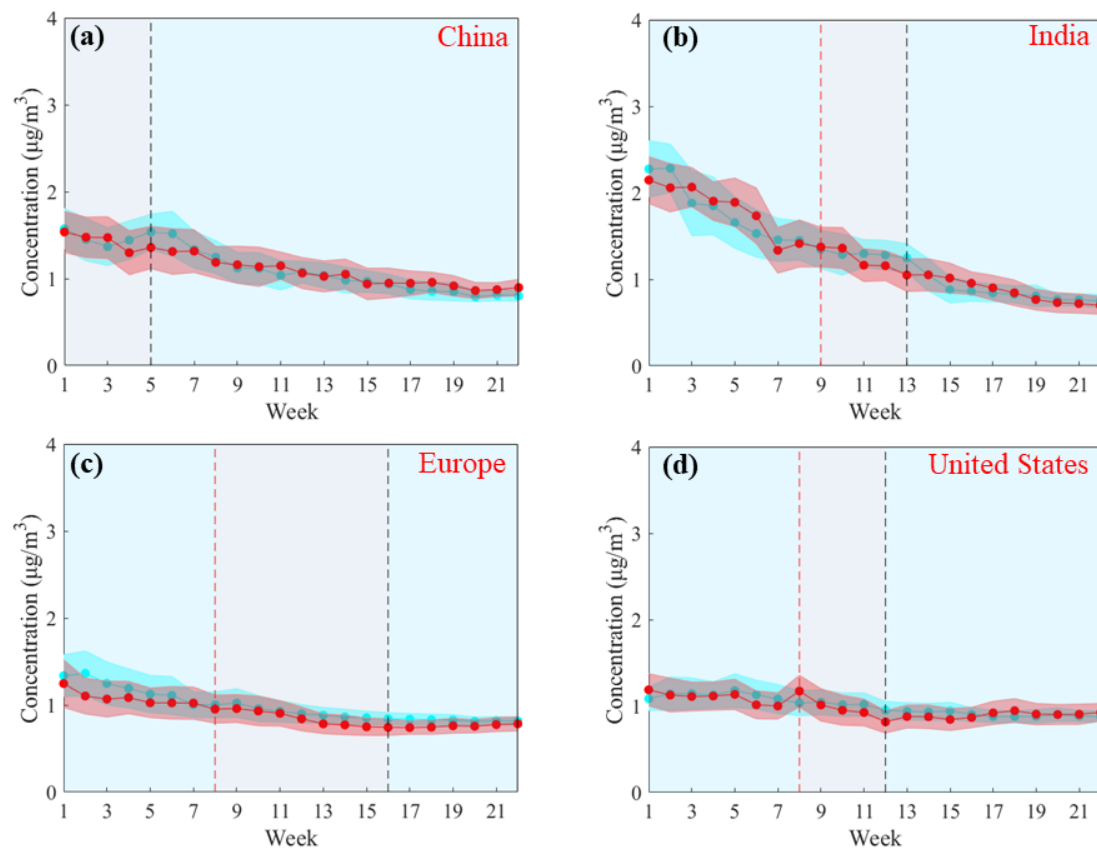


Figure 4 The concentration difference for deweathered benzene between COVID-19 period in 2020 and the same period in 2019 in East Asia, South Asia, Europe, and North America (Difference = deweathered benzene concentration in 2020-deweathered benzene concentration in 2019) (Unit: $\mu\text{g}/\text{m}^3$).

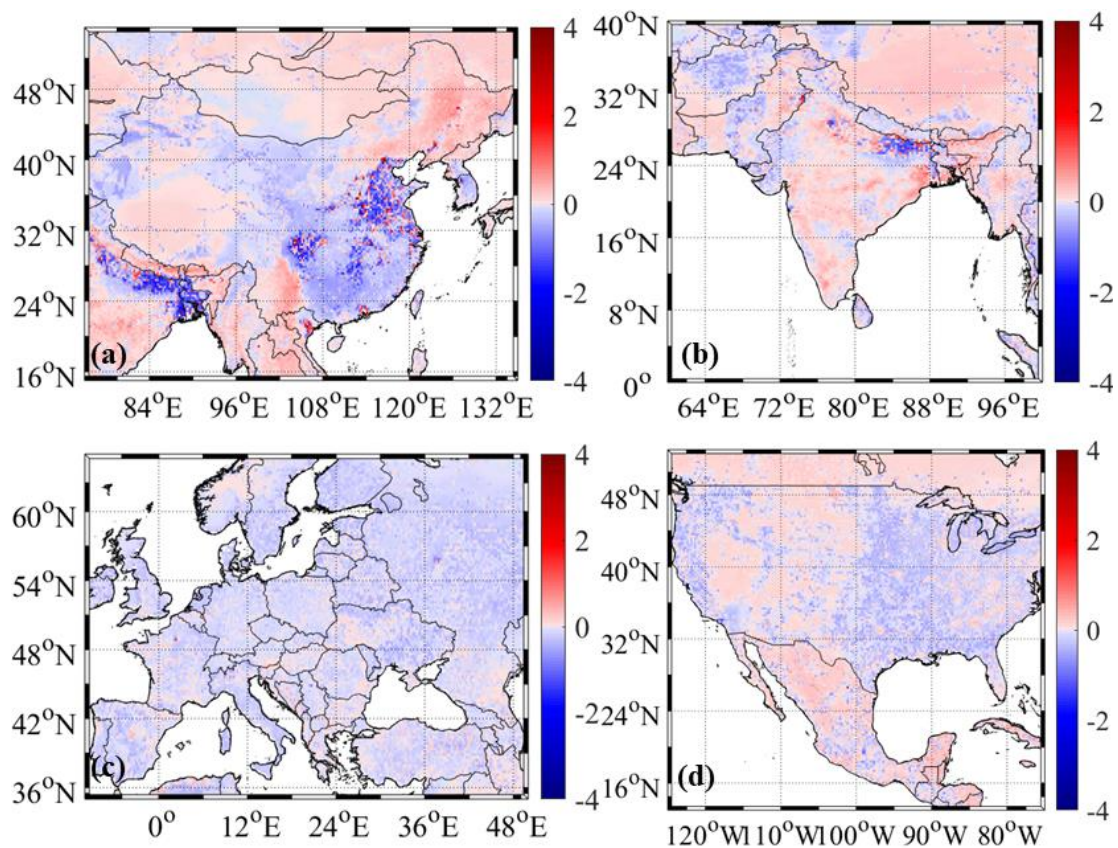


Figure 5 The carcinogenic risk differences (Unit: 10^{-7}) for atmospheric benzene between COVID-19 period in 2020 and the same period in 2019 in East Asia, South Asia, Europe, and North America (Difference = benzene concentration in 2020-benzene concentration in 2019).

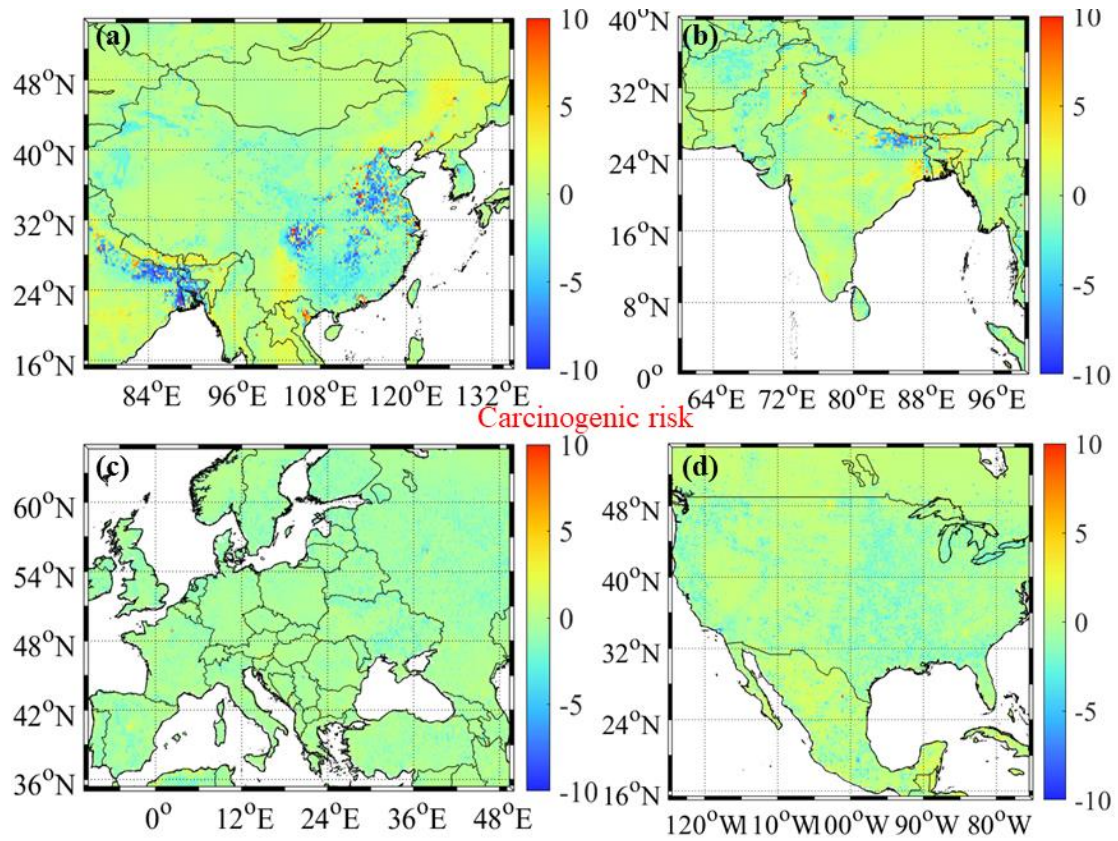


Figure 6 The non-carcinogenic risk differences (Unit: 10^{-3}) for atmospheric benzene between COVID-19 period in 2020 and the same period in 2019 in East Asia, South Asia, Europe, and North America (Difference = benzene concentration in 2020-benzene concentration in 2019).

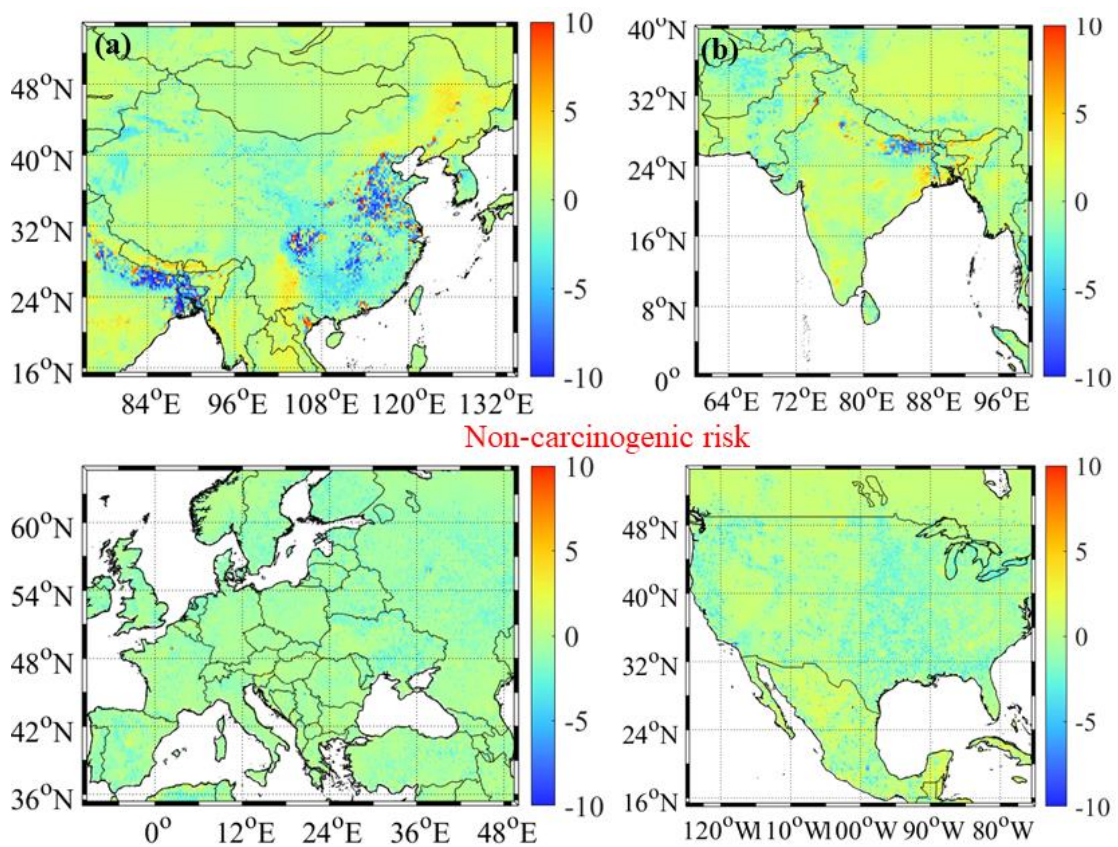


Table 1 The change ratio (%) of deweathered (P_{dew}) and detrended (P^*) benzene concentrations in major regions around the world.

Change ratio	China	India	Europe	United States
P_{dew} in 2020	-15.6	-23.6	-21.9	-16.2
P_{dew} in 2019	-2.31	-7.40	-8.04	-10.2
P^*	-13.3	-16.2	-13.9	-6.00

