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Increased tropospheric ozone levels as a public health issue during COVID-19 lockdown and estimation the related pulmonary diseases

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ABSTRACT

The aims of this study were to i) investigate the variation of tropospheric ozone (O_3) levels during the COVID-19 lockdown; ii) determine the relationships between O_3 concentrations with the number of COVID-19 cases; and iii) estimate the O_3-related health effects in Southwestern Iran (Khorramabad) over the time period 2019–2021. The hourly O_3 data were collected from ground monitoring stations, as well as retrieved from Sentinel-5 satellite data for showing the changes in O_3 levels pre, during, and after lockdown period. The concentration-response function model was applied using relative risk (RR) values and baseline incidence (BI) to assess the O_3-related health effects. Compared to 2019, the annual O_3 mean concentrations increased by 12.2% in 2020 and declined by 3.9% in 2021. The spatiotemporal changes showed a significant O_3 increase during COVID-19 lockdown, and a negative correlation between O_3 levels and the number of COVID-19 cases was found ($r = -0.59$, $p < 0.05$). In 2020, the number of hospital admissions for cardiovascular diseases increased by 4.0 per 10^5 cases, the mortality for respiratory diseases increased by 0.7 per 10^5 cases, and the long-term mortality for respiratory diseases increased by 0.9 per 10^5 cases. Policy decisions are now required to reduce the surface O_3 concentrations and O_3-related health effects in Iran.

1. Introduction

Air pollution in recent years became one of the major environmental and public health issues, with the industrialization, urbanization, and growing population (Amoatey et al., 2021; Anbari et al., 2022; Boogaard et al., 2019; Landrigan et al., 2018). Acute and chronic exposure to air pollutants has harmful effects on human health (Burnett et al., 2014; Khaniabadi et al., 2019a). Tropospheric ozone (O_3) is formed by photochemistry from nitrogen oxides (NO_X), volatile organic compounds (VOCs), and methane (De Marco et al., 2022; Héroux et al., 2015; Naghan et al., 2022). Over the last years, the urbanization and fossil fuel consumption led to rising background O_3 levels worldwide (Li et al., 2020a).

In cities worldwide, rising O_3 levels (Sicard, 2021) is associated with hospital admissions for cardiovascular and respiratory diseases, and mortality (Li et al., 2020b; Organization, 2021). To date, surface O_3 is a serious environmental problem in Europe, the United States, and across Asia, and is one of the most harmful air pollutants affecting human health, vegetation, materials, and biodiversity (Agathokleous et al., 2020; Carbajal-Arroyo et al., 2011; Sicard et al., 2019; Zhu et al., 2022).

The short-term O_3 effects on the respiratory systems is well-established, especially in sensitive people infected by obstructive chronic pulmonary diseases (Faustini et al., 2019). For the human health protection, the World Health Organization (WHO) set a limit value of 50 ppb calculated as daily 8-h running mean average. Previous studies e.g., in Beijing (Li et al., 2019), Isfahan (Abdolahnejad et al., 2018), Nice and Rome (Sicard et al., 2019), Suwon (Jeong, 2013), Hainan (Li et al., 2020a), Catalonia (Rovira et al., 2020), Shiraz (Bonyadi et al., 2020), Cape Town (Adebayo-Ojo et al., 2022), Kuwait (Al-Hemoud et al., 2021), and Paris (Filleul et al., 2006) have reported an increased rate of

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mortality and morbidity for exposure to the O_3. The chronic exposure to the O_3 levels led to 147,100 deaths within 13,000 cities in the world in 2019 (Malashock et al., 2022). The number of premature deaths due to O_3 exposure increased by 0.55 deaths per 10^6 people in the Europe Union in 2000–2017 (Stafoggia et al., 2022).

To assess the health effects due to O_3 exposure, few studies used the concentration-response functional model by using the relative risk, and baseline incidence values. This method was used in Ahvaz for estimating related respiratory diseases mortality by O_3 exposure in the period 2012–2018 (Naghan et al., 2022). Similar O_3 risk assessments were conducted based on concentration-response (C-R) function model in Texas (Zu et al., 2017), Surat City (Jariwala and Kapadia, 2022), Tabriz (Barzeghar et al., 2019), Canada (Zhao et al., 2021), Ahvaz (Amoatey et al., 2019), Mexico city and New York (Gurjar et al., 2010), Tehran (Hadei et al., 2017a), and Catalonia (Rovira et al., 2020).

During COVID-19 lockdown in 2020, reduced vehicle traffic and anthropogenic activities led to reduced emissions of NO_x and fine particles in cities, while the O_3 levels significantly increased e. g, in Suzhou (Wang et al., 2021a), Rome and Nice (Sicard et al., 2020), São Paulo (Nakada and Urban, 2020), Baghdad (Hashim et al., 2021), Barcelona and Madrid (Querol et al., 2021), Vienna (Brancher, 2021), Las Vegas and Los Angeles (Chen et al., 2020), San Fernando (Morales-Solís et al., 2021), and London (Vega et al., 2021). In China, a conducted study in 31 provincial cities illustrates a decline in NO_2 related to reducing of traffic emissions due to social distances for COVID-19 lockdown resulted in rising O_3 levels in 2020 (Nie et al., 2021).

In our study, we have investigated the spatial variation of surface O_3 levels during the COVID-19 lockdown; ii) determined the relationships between O_3 concentrations with the number of COVID-19 cases; and iii) estimated the O_3-related health effects in Southwestern Iran (Khorramabad) over the time period 2019–2021.

2. Materials and methods

2.1. The study area

Khorramabad (Fig. 1), with a semi-humid Mediterranean city is located in the southwest region of Iran (33°48′N; 48°35′E, 1147 m a.s.l.). This city with a population over 540,000 inhabitants, is the capital city of Lorestan Province. Due to the climate properties, the annual means of air temperature and precipitation are respectively 17.2 °C and 512 mm. Desert dust from neighbor countries in the Middle Eastern area is the main source of ambient pollution (Daryanoosh et al., 2018). Moreover, the petrochemical industries, heavy road traffic, and VOCs emissions have intensified poorly air quality in this area (Omidi Khaniabadi et al., 2019).

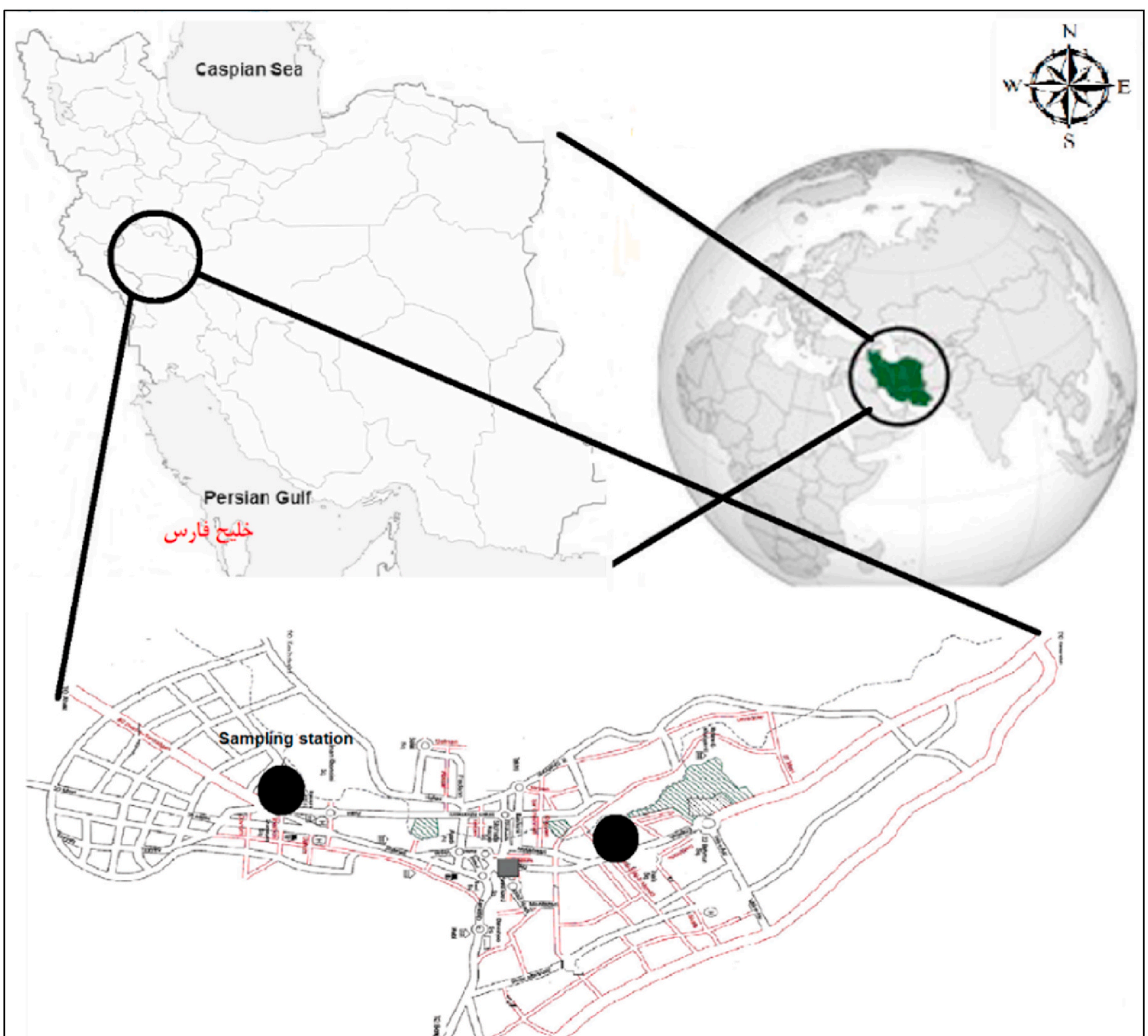

Fig. 1. Khorramabad city, southwest of Iran with the sampling stations.

2.2. 2. The O_3 concentrations and COVID-19 data collection

This study was performed using hourly O_3 data from two ground monitoring stations managed by the Environmental Protection Agency located in Khorramabad. The hourly data were collected from January 1, 2019 to December 31, 2021. The daily maximum 8-h moving average levels were obtained from at least 75% of valid hourly data per day. The number of COVID-19 cases was provided by the six public and private hospitals in Khorramabad, from March 21, 2020 to March 20, 2021 to assess the relationships with O_3 concentration.

2.3. Spatial variation

Sentinel-5 satellite images were used to analyze the spatial distribution of O_3 concentrations over Iran at 3.5 × 7.0 km of spatial resolution by using Google Earth Engine for geospatial analysis, and for three time periods: before (21st February to March 21, 2020), during (22nd March to April 21, 2020), and post-COVID-19 lockdown (22nd April to May 21, 2020).

2.4. Estimation of human health exposure

To estimate the population exposure to O_3 levels, at first standard Z-scores were identified in time series. The concentration-response (C-R) is a developed model from different meta-analyses and systematic reviews of health effects due to exposure to different air pollutants in the world (Amoatey et al., 2020; Jariwala and Kapadia, 2022; Naghan et al., 2022). For that, we need three parameters including the number of people at-risk (N) in a city, the baseline incidence (BI), and the relative risk (RR) values.

The RR value is the probability to develop a disease in a group in relation to the contact with a certain air pollutant, per each 5 ppb increase in concentration (Eq. (1)), and obtained from published epidemiological studies. The risk ratio taken from epidemiological studies in local or national scales are more robust for risk assessment (Khaniabadi and Sicard, 2021); nevertheless use of obtained RR from the different conducted studies in the other regions such as United States and Europe is acceptable by the WHO to investigate the risk of air pollution where epidemiological studies are missing. We used the RR values recommended by the WHO (Table 1) and published by Héroux et al. (2015).

$$RR = exp[\beta * (X - X_0)] \tag{1}$$

where β regulates the RR value increasing, also parameter X is the amount of air pollutant and X_0 is background level where there is not observed health effect (De Marco et al., 2018). The city-specific baseline incidence (BI per 10^5 inhabitants) was used for each health endpoint. The BI was provided by the six public and private hospitals in Khorramabad. Furthermore, the attributable proportion (AP%) among a population at risk, $P_{(c)}$, is a fraction of health effect statistically associated with exposure to a certain population in a given time period (Eq. (2)).

Table 1
The BI and RR values for the estimation of health effects in a 95% CI for assess of O_3 exposure more than 5 ppb (lag0-1 days).

Health outcomes	BI[a]	RR[b] per 5 ppb (95% CI[c])	Ref.
Hospital admission-CVD	436	1.009 (1.005–1.0127)	(De Marco et al., 2022), (Héroux et al., 2015)
Respiratory mortality (age≥30)	44–53	1.014 (1.005–1.024)	(De Marco et al., 2022), Héroux et al. (2015)
Long-term respiratory mortality	66	1.013 (1.005–1.024)	Jariwala and Kapadia (2022)

[a] Baseline incidence.
[b] Relative risk.
[c] Confidence interval.

$$AP = \sum([RR(c) - 1] * P(c)]) \Big/ \sum [RR(c) * P(c)] \tag{2}$$

where AP and RR(c) are respectively attributable proportion and relative risk for a certain health effect in category “c" acquired from the exposure-response functions resulting from published meta-analysis and epidemiological studies (Hadei et al., 2017b; Moradi et al., 2022; Rovira et al., 2020), and P(c) defined as the proportion of exposed population in category “c" exposure (Fattore et al., 2011; Gurjar et al., 2010; Khaniabadi et al., 2017b).

In an at-risk population the number of cases, NC_c per 10^5 people, correlated with a specific health effect because of exposure to air pollutant can be estimated by $NC_c = BI * AP$, as well as NE, refers to the number of excess cases among people attributed to air pollutant in category “c" exposure, can be estimated by $NE_c = 10^{-5} * [NC_c * N]$, where N is the number of at-risk population contacted with a certain air pollutant (Anbari et al., 2022; De Marco et al., 2018).

The indicator of Sum of Ozone Means Over 35 ppb (SOMO35), is recommended by the WHO to assess of the O_3-related health effect. The SOMO35 (in ppb days) is identified as the annually sum of maximum daily 8-h moving means of O_3 higher than 35 ppb (Eq. (3)).

$$SOMO35 = \sum_i max * [0.(C_i - 70)] \tag{3}$$

where C_i and i respectively refers to the maximum daily O_3 mean levels (in ppb) and the number of days over year (365 or 366). Since SOMO35 is sensitive to missing data, it is actually modified to full annual coverage by valid O_3 daily concentrations (N_{valid}) corresponding to following formula (Eq. (4)).

$$SOMO35 = SOMO35_{uncorrected} * N_{total}/N_{valid} \tag{4}$$

where, N_t is the day number during a year and N_v is the validated daily O_3 average concentrations. For long-term health effects estimation due to O_3 exposure, the log-linear function (Eq. (5)) was used.

$$RR = exp(\beta.SOMO35_{uncorrected} / N_{valid}) \tag{5}$$

For the health effects due to O_3 exposure, the model evaluates the integrated exposure-response (IER) function (Eq. (6)).

$$RR = \exp[\beta.\{C_{max\ 8} - X_0\}] \tag{6}$$

where β, C_{max8} (in ppb), and X_o are respectively the increase in the amount of RR, the maximum daily 8-h average O_3 concentrations, and the O_3 daily concentration.

The rate of incidence the specified health outcome at a population (BI, per 100,000) can be used for SOMO35 above a defined reference level by Eq. (7).

$$E = BI \times PAF \tag{7}$$

where BI is an obtained value of certain health effect in a population, PAF is the population attributable fraction which calculated by the amount of RR and multiplying the population (P) by following Eq. (8):

$$PAF = RR - 1/RR \tag{8}$$

The WHO recommended that to estimate the mortality and morbidity between people more than 30-year-old and at-risk population related to O_3 exposure, it can be used the C-R function model (Burnett et al., 2014; Faridi et al., 2018; Héroux et al., 2015).

In this study, the rate of hospital admissions due to cardiovascular diseases (HA-CVD), respiratory diseases mortality (MRD) for people over 30-year-old, and long-term respiratory diseases mortality (LT-MRD) were assessed by using O_3 concentrations between 2019 and 2021. Based on ICD-10, the codes J00-J98 are related to respiratory disease, J95 and J96 is related to respiratory mortality, I00–I99 are associated to cardiovascular disease.

Table 1 illustrates the amounts of BI and RR to assess the health effects attributed to O_3 exposure in Khorramabad (Iran). Normal distribution of data was analyzed by nonparametric test of the Kolmogorov-Smirnov 1-sample D-test. The analyses of correlations between the O_3 concentrations and the number of COVID-19 cases were performed by linear regression and Pearson correlation.

3. Results and discussion

3.1. Surface O_3 concentrations and COVID-19 cases

The city-averaged annual O_3 concentrations from 2019 to 2021 were assessed (Fig. 2). The highest annual mean concentration was observed in 2020 during COVID-19 (15.8 ppb). The O_3 concentrations in 2020 increased by 12.2% compared to 2019 and declined by 3.9% in 2021. High levels of sand from the Middle Eastern dessert (Broomandi et al., 2022) reached Southwestern Iran leading to high level of PM (Khaniabadi et al., 2017a, 2019b; Omidi Khaniabadi et al., 2019; Rashidi et al., 2022). The high PM levels led to lower solar radiation, and by consequence to lower O_3 formation (Anbari et al., 2022; Grange et al., 2021; Higham et al., 2021; Siciliano et al., 2020), but also to destruction of O_3 on PM surface (Sicard et al., 2020). In Hanoi, with 20% increase in the surface solar activity radiation due to lower $PM_{2.5}$ concentrations, 41% increase was observed in the O_3 levels during COVID-19 lockdown (Nguyen et al., 2022).

Our results showed that more than 8560 COVID-19 cases were admitted to hospitals in Khorramabad from March 21, 2020 to March 20, 2021. In 2020, the number of COVID-19 cases was significantly and negatively correlated ($r = -0.59$, $p < 0.05$) to surface O_3 levels (Fig. 2).

A similar negative association was observed between COVID-19 cases and O_3 concentration in 2020 in Baghdad (Iraq) (Hashim et al., 2021). The dramatically reducing in the human activities and social distances may be the reasons for decline in COVID-19 cases and the increase in O_3 levels in the cities (Chen et al., 2020). The O_3 concentrations in Milan increased in 2020, but they reported that there was a positive correlation among COVID-19 cases and O_3

Levels (Zoran et al., 2020), which do not confirm by the our paper.

3.2. Spatial variation in Iran during COVID-19 lockdown

The spatiotemporal changes of surface O_3 levels in Iran showed that there is an increase during COVID-19 lockdown, ranging from 0.1355 to 0.1444 mol O_3 m^{-2} (i.e., 32.58–34.73 ppb). Investigation for three-months pre- (21st February to 21st March), during (22nd March to 21st April), and post-lockdown period (22nd April to 21st May) in 2020 are in Fig. 3. We observed an increase of O_3 mean concentration (+6.5%) during lockdown compared to the previous month, and a reduction of 6.0% after lockdown. During lockdown, the highest change of O_3 levels was observed in the north-western Iranian provinces (Ardabil, West- and East-Azerbaijan, Urmia, and Zanjan), and the lowest change was observed in the southeastern areas (Sistan and Baluchistan province). The mechanism of related factors for high concentrations of O_3 not well studied but frequently increased during recent years with decreasing in the PM concentrations (Chen et al., 2021). The O_3 levels increased in March–April in Iran (Naqvi et al., 2021), then decreased in last days April 2020. In Tehran (Iran), the O_3 levels increased by 53.7% in 2020 compared to averaged 5-years ago at Aghdasieh station (Aghashariatmadari, 2021), and by 28.8% at Sharif-University station compared to the same period in 2019 (Broomandi et al., 2020). In Ardabil the monthly changes during lockdown increased O_3 levels by 51% compared to two months before COVID-19 lockdown (Rad et al., 2021).

In Khorramabad, the O_3 mean concentrations increased from 11.5 to 15.4 ppb during the lockdown. The results also indicated that O_3 mean concentrations increased during the lockdown (22nd March to April 21, 2020) by 33.8% in Khorramabad (Fig. 4). The annual O_3 mean concentrations increased by 12.2% between 2019 and 2020. This increase was also observed in Europe, the United States, and across Asia at the COVID-19 lockdown (Briz-Redón et al., 2021; Lian et al., 2020; Tobías et al., 2020). In six cities in the Europe and China, the O_3 concentration was increased during the lockdown (Sicard et al., 2020) which confirms our results.

Similar observations were reported in the conducted studies in Rio de Janeiro (Siciliano et al., 2020), Hanoi (Nguyen et al., 2022), Baghdad (Hashim et al., 2021), Tel-Aviv (Agami and Dayan, 2021), Barcelona (Tobías et al., 2020), Almaty (Kerimray et al., 2020), and Jiangsu (Bhatti et al., 2021). When the lockdown period started, the number of emission sources, volume of road traffic, and the air pollutants ($PM_{2.5}$ and NO_X) concentrations were dropped (Wang et al., 2021b); dropping in NO_X and VOC-limited background, resulted in higher O_3 levels (Tobías et al., 2020).

3.3. Human health effect

The number of hospital admissions for cardiovascular diseases (6.6, 10.6, and 8.3 per 10^5 cases), premature deaths for respiratory diseases (4.7, 5.4, and 4.9 per 10^5 cases), and long-term premature deaths for respiratory diseases (1.7, 2.6, and 2.2 per 10^5 cases) due to O_3 exposure above 5 ppb were obtained in 2019, 2020, and 2021, respectively (Table S1). HA-CVD, MRD, and LT-MRD increased by 60.6%, 14.8%, and 52.9% per 10^5 cases in 2020, due to higher O_3 exposure (Schwarz et al., 2021) and then decreased by 21.0%, 8.9%, and 15.4% per 10^5 cases in 2021 when annual mean of O_3 levels was reduced. In China, the authors observed a positive relationship between increase in O_3 levels and the risk of premature mortality, especial between exposure to O_3 and the risk of respiratory diseases and mortality (Liu et al., 2017). A study illustrated that the early O_3 exposure resulted in changes in the

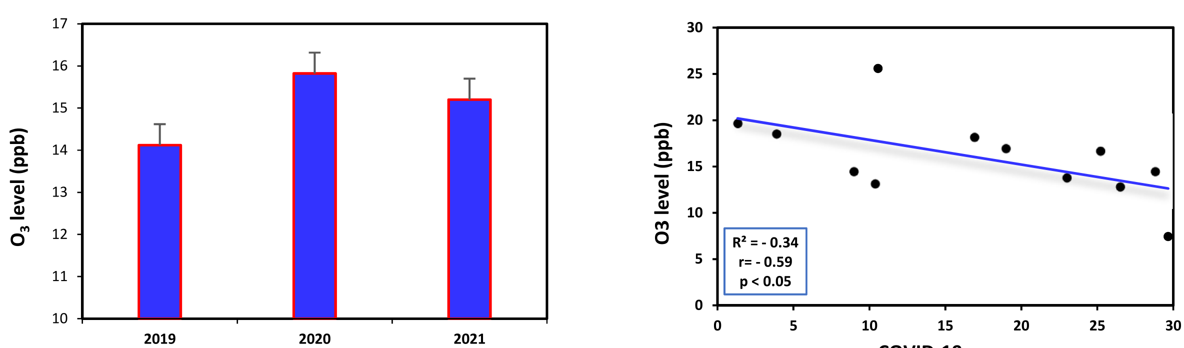

Fig. 2. (Left) The annual mean concentration of O_3 during 2019, 2020, and 2021 and (right) the relationship between COVID-19 cases and O_3 concentration during March 21, 2020 to March 20, 2021.

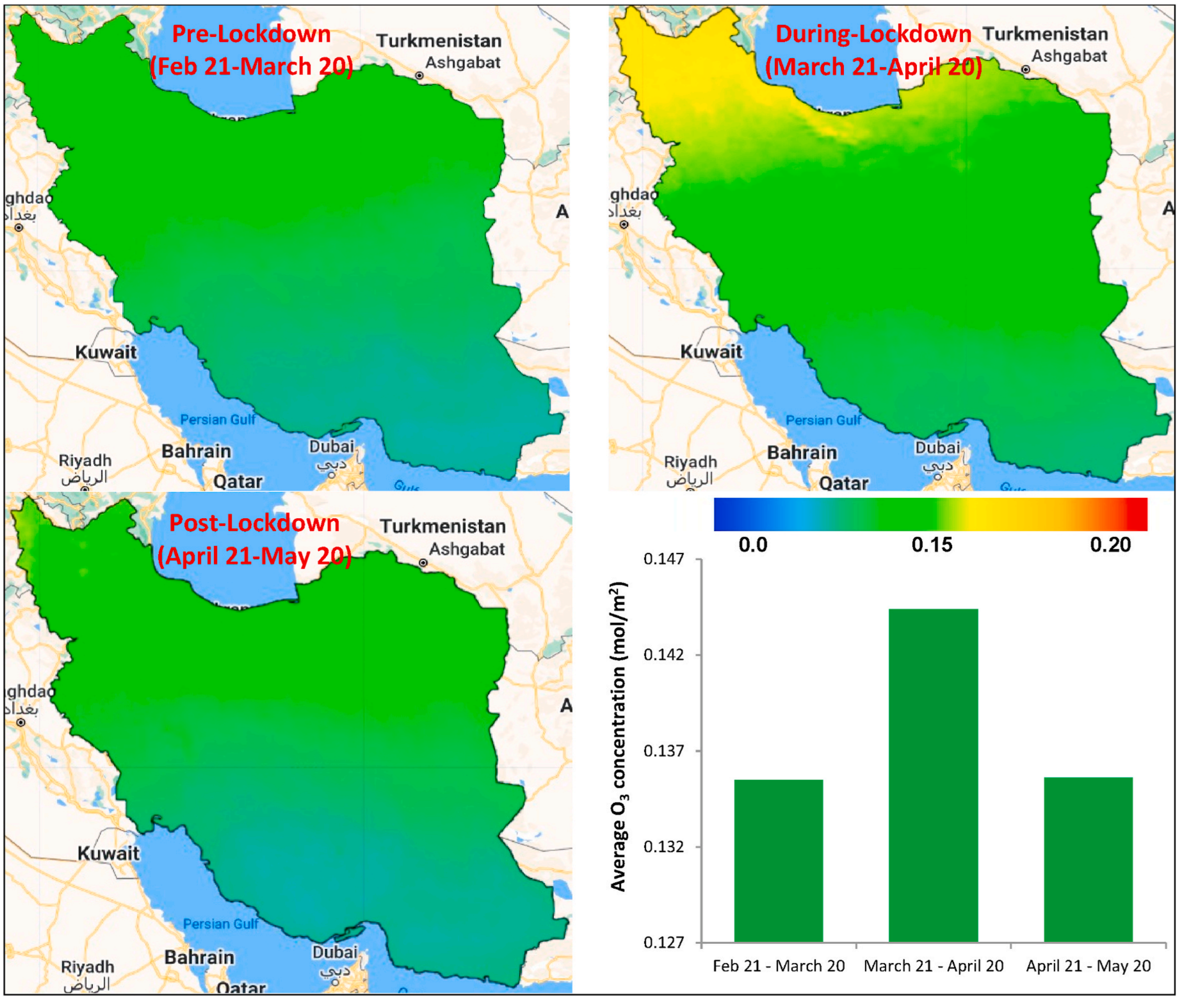

Fig. 3. The O_3 spatiotemporal variations in Iran during three monthly periods; before (21st February to March 21, 2020), during (22nd March to April 21, 2020), and after (22nd April to May 21, 2020) COVID-19 lockdown.

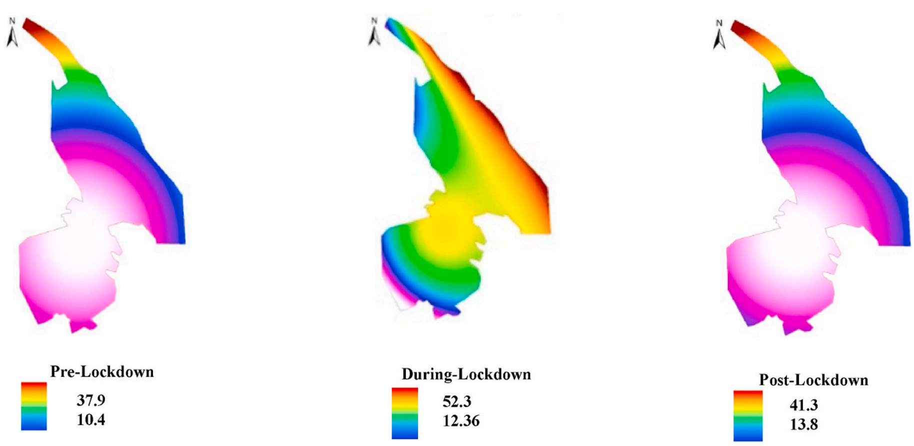

Fig. 4. Spatial distribution of O_3 concentrations before (21st February to March 21, 2020), during (22nd March to April 21, 2020), and after (22nd April to May 21, 2020) COVID-19 lockdown in Khorramabad, Iran.

conducting airways, and by increase in age, the potential relative risk increased due to O_3-related health effect (Vinikoor-Imler et al., 2014). The number of HA-CVD and MRD in 2020 and 2021, without COVID-19 effect, for O_3 exposure were HA-CVD = 6.1 and 4.9 per 10^5 cases at-risk population, MRD = 4.0 and 3.6 per 10^5 cases at-risk population, respectively. The number of MRD due to O_3 exposure in Ahvaz (Iran) with population higher than 1.3 million was 6.17 for people with ages ≥30 year-old (Karimi et al., 2019). In Surat (India), 24 HA-CVD, 1 MRD, and 4 LT-MRD per 10^5 people were attributed to O_3 exposure in 2018 (Jariwala and Kapadia, 2022).

Fig. 5 show more detailed the health effects estimation in Khorramabad due to O_3 exposure at different concentration intervals between

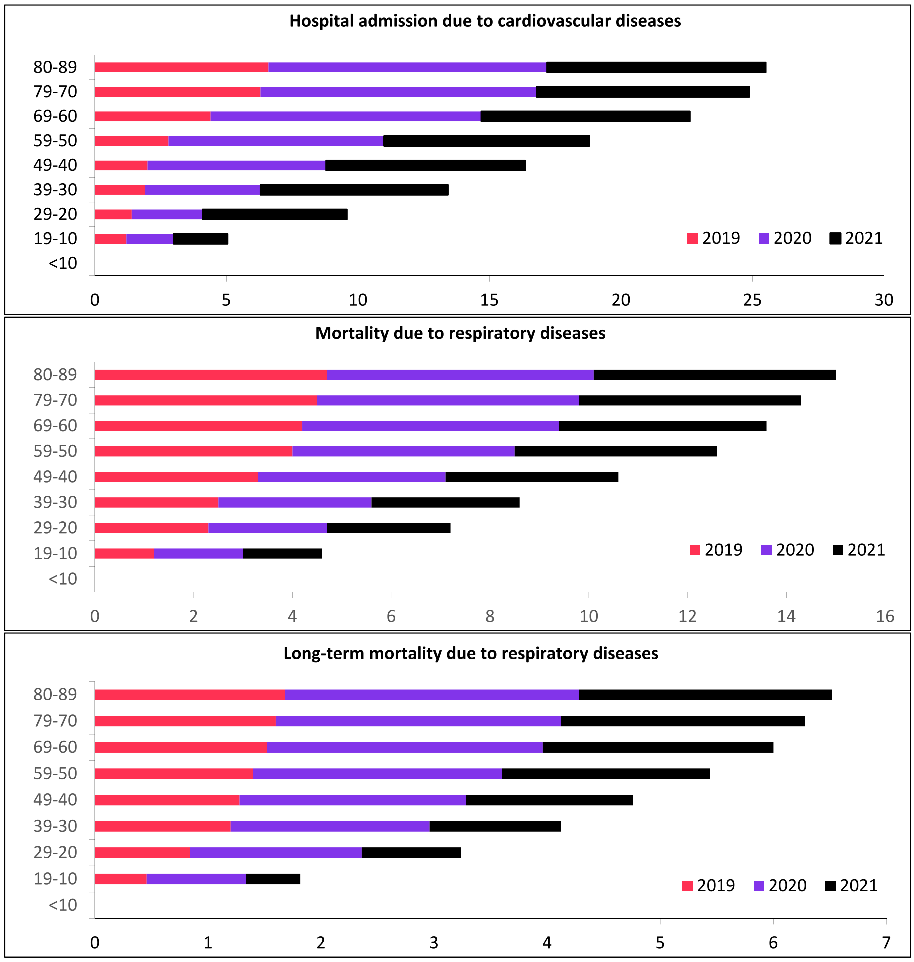

Fig. 5. Association between HA-CVD, MRD, and LT-MRD cases and categorical daily 8-h average O_3 concentrations in Khorramabad, during 2019–2021.

years 2019–2021. The number of estimated cases according to RR distribution (95% confidence interval) for different concentration classes were obtained (Tables S2–S4). The highest attributable number was observed at higher O_3 levels, however the 80–89 ppb interval resulted in a higher exposure, therefore to more attributable cases of HA-CVD, MRD, and LT-MRD. The highest HA-CVD, MRD, and LT-MRD was estimated in 2020, due to higher O_3 concentrations.

In Tehran, in an annual mean 20.3 ppb, 272 HA-CVD due to O_3 exposure was calculated by AirQ software in 2015–2016 (Hadei et al., 2017a). The premature respiratory diseases mortality of O_3 exposure was equal to 7.16 in Hamadan in Iran (Asl et al., 2018). In Portugal, the estimated health effects due to exposure to O_3 among under exposed population depicted that MRD = 242 cases in 2013, and decreased to 143 cases in 2019 (Brito et al., 2022). In China, a study showed that LT-MRD increased by 0.55% for each 5 ppb increase in the daily maximum 8-h mean concentration of ambient O_3 (Zhang et al., 2021). The mean of premature deaths due to exposure by O_3 pollution in the Hainan Island (China) was reported 391 (95%CI: 212–570) person as annually (Li et al., 2020a).

4. Conclusion

In the current study, we investigated the trends of O_3 levels during COVID-19 lockdown in Iran and Khorramabad city by using spatio-temporal variation, and attributable health risks including HA-CVD, MRD, and LT-MRD related to O_3 exposure from 2019 to 2021. Between 2019 and 2021, the annual O_3 mean concentrations ranged from 14.1 to 15.8 ppb. The number of COVID-19 cases was negatively correlated to surface O_3 levels ($r = -0.59$, $p < 0.05$). The results showed rising O_3 levels during 2020, led to a higher number of O_3-related diseases (+60.6%, +14.8%, and +52.9% for HA-CVD, MRD, and LT-MRD, respectively). The higher levels of O_3 in 2020 is a noticeable factor contributing to higher number of mortality and morbidity compared to 2019 and 2021.

In Iran, there is no valuable law for control of air pollution emissions from the industries, road traffic, and fossil fuel burning. Monitoring and enforcing are outside the government's power. In order to obtain policy response and the establishment of relevant policies for human health protection in Iran, we have to explore technological innovations such as monitoring by satellite, and provide more information about the links between air pollutants and health issues at local and national levels. Our results allow better knowledge and understanding about changes of O_3 concentrations during pandemic and its effect on human health.

Undergoing the rise in the emissions from anthropogenic activities as precursors of ground-level O_3, the number of cardiovascular and respiratory mortality per year increased in the different areas, while the land cover changes e.g., by increasing the vegetation and greenspace density may decline the effect of emission-driven related health; however, climate change especially global warming intensifies furtherly those impacts. Such outcomes highlight the emission control strategies associated with air oxidants such as O_3 can helps to reduce environmental pollution and public health issues (Fu and Tai, 2015).

Although our study has performed by the valuable methodology proposed by the WHO, but there are still limitations (Khaniabadi and Sicard, 2021). Due to unavailability information for interactions between different air pollutants, the health effect studies are generally focused on a single pollutant. Similar studies only reflect exposure to a specified outdoor air pollutants while people commonly spend more time indoors (Guan et al., 2021). In general, this recommended model assumes that exposure to air pollution concentration measured at the monitoring stations are representative to all people living within a certain city at the same time (Guan et al., 2021; Khaniabadi et al., 2017c).

Author statement

Khatereh Anbari: project administration, designed research, and funding acquisition, **Yusef Omidi Khaniabadi**: methodology, writing original draft, data curation and analysis, and writing-review and editing, **Pierre Sicard**: validation, writing-review, and editing, **Hasan Raja Naqvi**: software, and created the figures, **Rajab Rashidi**: monitoring, data collection, and designed research. All authors read and approved the final manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.apr.2022.101600.

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