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Long-term trends and health impact of $PM_{2.5}$ and O_3 in Tehran, Iran, 2006–2015

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A R T I C L E I N F O

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A B S T R A C T

The main objectives of this study were (1) investigation of the temporal variations of ambient fine particulate matter ($PM_{2.5}$) and ground level ozone (O_3) concentrations in Tehran megacity, the capital and most populous city in Iran, over a 10-year period from 2006 to 2015, and (2) estimation of their long-term health effects including all-cause and cause-specific mortality. For the first goal, the data of $PM_{2.5}$ and O_3 concentrations, measured at 21 regulatory monitoring network stations in Tehran, were obtained and the temporal trends were investigated. The health impact assessment of $PM_{2.5}$ and O_3 was performed using the World Health Organization (WHO) AirQ+ software updated in 2016 by WHO European Centre for Environment and Health. Local baseline incidences in Tehran level were used to better reveal the health effects associated with $PM_{2.5}$ and O_3. Our study showed that over 2006–2015, annual mean concentrations of $PM_{2.5}$ and O_3 varied from 24.7 to 38.8 $\mu g\,m^{-3}$ and 35.4 to 76.0 $\mu g\,m^{-3}$, respectively, and were significantly declining in the recent 6 years (2010–2015) for $PM_{2.5}$ and 8 years (2008–2015) for O_3. However, Tehran citizens were exposed to concentrations of annual $PM_{2.5}$ exceeding the WHO air quality guideline (WHO AQG) (10 $\mu g\,m^{-3}$), U.S. EPA and Iranian standard levels (12 $\mu g\,m^{-3}$) during entire study period. We estimated that long-term exposure to ambient $PM_{2.5}$ contributed to between 24.5% and 36.2% of mortality from cerebrovascular disease (stroke), 19.8% and 24.1% from ischemic heart disease (IHD), 13.6% and 19.2% from lung cancer (LC), 10.7% and 15.3% from chronic obstructive pulmonary disease (COPD), 15.0% and 25.2% from acute lower respiratory infection (ALRI), and 7.6% and 11.3% from all-cause annual mortality in the time period. We further estimated that deaths from IHD accounted for most of mortality attributable to long-term exposure to $PM_{2.5}$. The years of life lost (YLL) attributable to $PM_{2.5}$ was estimated to vary from 67,970 to 106,706 during the study period. In addition, long-term exposure to O_3 was estimated to be responsible for 0.9% to 2.3% of mortality from respiratory diseases. Overall, long-term exposure to ambient $PM_{2.5}$ and O_3 contributed substantially to mortality in Tehran megacity. Air pollution is a modifiable risk factor. Appropriate sustainable control policies are recommended to protect public health.

1. Introduction

Air pollution is a major environmental risk factor affecting human health in both developed and developing countries (Abajobir et al., 2017; Cohen et al., 2017; Forouzanfar et al., 2016; Münzel et al., 2017; West et al., 2016). Exposure to air pollutants can cause a variety of health effects depending on the pollution composition (variety of particles and gases), exposure level, duration and frequency of exposure, and associated toxicity of the specific pollutant (Amini et al., 2017b; Brunekreef and Holgate, 2002; Hassanvand et al., 2017; Landrigan,

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2017). These exposures are associated with a broad range of acute and chronic health effects varying from sub-clinical effects to premature mortality (Arnold, 2014; Calderón-Garcidueñas et al., 2015). Although ambient air pollutants are many, the most studied markers with independent effects are fine particulate matter ($PM_{2.5}$: particles with an aerodynamic diameter of < 2.5 μm) and O_3 (Aksoyoglu et al., 2014; Brauer et al., 2012; Broome et al., 2015; Jacob and Winner, 2009; Sofen et al., 2015). In 2015, the Global Burden of Diseases (GBD) study estimated that long-term exposures to ambient $PM_{2.5}$ caused 4.2 million deaths, making it the fifth-ranking mortality risk factor globally, and 103.1 million disability-adjusted life-years (DALYs), representing 7.6% of total global deaths and 4.2% of global DALYs (Cohen et al., 2017). In 2015, ambient ozone contributed to 254,000 additional deaths, making it the 33rd highest ranking risk factor globally for deaths (Cohen et al., 2017). Research on ambient $PM_{2.5}$ and O_3 and their health impacts are more distinguished for megacities (Cheng et al., 2016) such as Los Angeles (Hasheminassab et al., 2014), Beijing (Zhang et al., 2007), Tokyo (Hara et al., 2013), Delhi (Amann et al., 2017), Mumbai (Joseph et al., 2012), Paris (Bressi et al., 2013) and Mexico City (Calderón-Garcidueñas et al., 2015). Quantitative health impact assessments of air pollution have been extensively applied since one of the first international studies that quantified the public-health impact of air pollution in Austria, France, and Switzerland (Künzli et al., 2000). These assessments greatly support decisions and policy-makers in understanding the health benefits that would be associated with improved air quality (Anenberg et al., 2016; Arranz et al., 2014; Dhondt et al., 2012; Krzyzanowski et al., 2002; Likhvar et al., 2015; Naddafi et al., 2012).

To date, several air pollution health impact assessment studies have been published that estimated the health effects based on previous versions of the WHO's AirQ tool (Anenberg et al., 2016; Conti et al., 2017; Fattore et al., 2011; Ghozikali et al., 2016; Hadei et al., 2017; Kermani et al., 2017; Moustris et al., 2017; Naddafi et al., 2012; Orru et al., 2009). However, dose-response functions and relative risks (RR) have been updated based on new epidemiological studies in the WHO AirQ+ software updated in 2016 by WHO European Centre for Environment and Health (Héroux et al., 2015; WHO, 2017). The main objectives of the current study were: 1) to investigate temporal variations of $PM_{2.5}$ and O_3 concentrations over a 10-year period (2006–2015), and 2) to assess health impacts of long-term exposure to $PM_{2.5}$, and O_3 by using the WHO AirQ+ software in Tehran megacity. Tehran is the largest Middle Eastern megacity with about 9 million residents and a day time population of over 10 million people owing to high number of commuters from other cities. Inhabitants in Tehran are mostly exposed to high levels of $PM_{2.5}$ and O_3 including frequent smog episodes where schools and governmental offices become closed due to sever air pollution events (Abdo et al., 2016; Faridi et al., 2015; Faridi et al., 2017; Hassanvand et al., 2014; Hassanvand et al., 2015; Hosseini and Shahbazi, 2016; Naddafi et al., 2012; Shahbazi et al., 2016). In spite of the significance of ambient $PM_{2.5}$ and O_3 in Tehran, there is little information published on temporal trend and long-term health effects of $PM_{2.5}$ and O_3 in this area. High air pollution levels in Tehran are likely the result of vehicles and industrial emissions, as well as specific geographic conditions, with the Alborz Mountains in the north and a desert in the south (Shahbazi et al., 2016).

2. Material and methods

2.1. Air quality data gathering and processing

Real-time hourly concentrations of $PM_{2.5}$ and O_3 were obtained from 21 air quality monitoring stations (AQMSs) in Tehran during the 10-year period 2006–2015. All AQMSs belonged to Tehran Air Quality Control Company (TAQCC) (Fig. 1). The air pollutant data measured at the AQMSs have been recently publicly available via the Internet. At each AQMSs, $PM_{2.5}$ and O_3 are measured using the beta-attenuation and UV-spectrophotometry methods, respectively. Table S1 shows information regarding Tehran AQMSs.

The air quality data were processed in the following ways: To remove spatial-temporal outliers, hourly concentrations were compared with values from neighboring monitoring stations and with the time trends data (Song et al., 2017). Our removal criterion consisted of four conditions as used in previous research (Barrero et al., 2015; Song et al., 2017). First, the series data were transformed into standard scores (i.e. z scores). The points in the transformed time series meeting the conditions (1) having an absolute z score larger than 4 ($|z_t| > 4$), (2) the increment from the previous value being larger than 9 ($z_t - z_{t-1} > 9$), (3) the ratio of the value to its centered rolling mean of order 3 (RM3) being larger than 2 ($z_t/RM3(z_t) > 2$), and (4) individual monitoring station's increment from the previous value being two times larger than belonged city's all monitoring station's averaged increment ($city(z_t - z_{t-1})$) from the previous value (i.e., $(z_t - z_{t-1}) / city(z_t - z_{t-1}) > 2$), were then removed from the hourly raw data. Minimum of 75% of valid data per year was set as an inclusion criterion for each station; so, if > 25% of hourly concentrations for one of the $PM_{2.5}$ and O_3 for a given station was missing, that specific air pollutant was excluded from the further analyses. Finally, based on the mentioned inclusion and exclusion criteria, there were 10 and 6 eligible stations for $PM_{2.5}$ and O_3, respectively (Table S1). Daily average concentrations were calculated from the hourly data when > 75% of the data per day were valid measurements. Hourly city levels were calculated for each air pollutant by averaging all available data across the selected monitoring stations. To compute the annual and monthly concentrations of $PM_{2.5}$ and O_3, we used hourly concentrations. Maximum daily 8-h O_3 concentrations were calculated based on hourly O_3 concentrations and the long-term O_3 mean was calculated based on the mean of daily max 8-h means. To learn more about the temporal trends of each air pollutants, the $PM_{2.5}$ and O_3 data were compared after being sorted into diurnal, monthly, seasonal and annual $PM_{2.5}$ and O_3 groups. SOMO35 metric (annual sum of maximum daily 8-h ozone means over 70 μg m^{-3}) is an important indicator for estimating premature deaths due to O_3 exposures in health impact assessment calculations (Aksoyoglu et al., 2014; Lacressonnière et al., 2016; Malley et al., 2015; Nawahda et al., 2013; Sicard et al., 2016; Sofen et al., 2015). We calculated SOMO35 ((μg m^{-3})·days) for the study period as follows (Aksoyoglu et al., 2014; Ellingsen et al., 2008; Lacressonnière et al., 2016):

$$SOMO35_{uncorrected} = \sum_{i=1}^{N_{total}} \max\left[0, (C_i - 70\ \mu g\ m^{-3})\right] \tag{1}$$

$$SOMO35 = SOMO35_{uncorrected}\frac{N_{total}}{N_{Valid}} \tag{2}$$

where $[C_i]$ is the maximum daily 8-h running average O_3 concentration (μg m^{-3}); N_{total} is the number of days for the whole year (365 or 366 for a year); and N_{Valid} represents the number of valid daily concentrations ($N_{Valid} > 273$). The function "maximum" ensures that only concentrations exceeding 70 μg m^{-3} are taken into calculation. Due to lack of $PM_{2.5}$ measurements in some monitoring stations during 2006 to 2010, daily ratio of $PM_{2.5}/PM_{10}$ was calculated for each station according to the available data from 2011 to 2015. Then the $PM_{2.5}$ concentrations were estimated based on the acquired ratios for the associated monitoring station. The calculated annual mean of daily $PM_{2.5}/PM_{10}$ ratios ranged from 0.38 to 0.44 during 2011–2015, as compared to 0.33–0.44 for Delhi, 0.73 for Europe, 0.65 for America, and 0.63 for Beijing (Maji et al., 2017). The results observed are relatively lower than the value of 0.50 recommended by Ostro (Ostro, 2004) for developing countries. The correlation between $PM_{2.5}$ and PM_{10} was high (r = 0.86), as illustrated in Fig. S1.

Diurnal meteorological data (temperature, dew point, wind speed and precipitation) over the study period were obtained from Iran Meteorological Organization data stored in the University of Wyoming archive (http://weather.uwyo.edu/upperair).

The non-parametric Mann-Kendall trend test was used to detect