Science of the Total Environment 927 (2024) 171997

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Dynamic and stationary monitoring of air pollutant exposures and dose during marathons

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HIGHLIGHTS

- Marathon athletes are exposed to air pollution impacting health and performance.
- Fixed and mobile monitors were deployed in 3 marathons and inhaled doses calculated.
- Fixed monitoring prior to the marathon is key to understand pollutant hourly trends.
- Mobile monitoring helps to identify hotspots and provides hyper-local exposures.
- Inhaled O_3 and PM doses may be higher for the slowest than for the fastest runners.

GRAPHICAL ABSTRACT

ARTICLE INFO

Editor: Pavlos Kassomenos

Keywords:
Athletics
Air quality
Amateur and elite runners
Urban race
Inhaled dose
Gender

ABSTRACT

Marathon running significantly increases breathing volumes and, consequently, air pollution inhalation doses. This is of special concern for elite athletes who ventilate at very high rates. However, race organizers and sport governing bodies have little guidance to support events scheduling to protect runners. A key limitation is the lack of hyper-local, high temporal resolution air quality data representative of exposure along the racecourse. This work aimed to understand the air pollution exposures and dose inhaled by athletes, by means of a dynamic monitoring methodology designed for road races. Air quality monitors were deployed during three marathons, monitoring nitrogen dioxide (NO_2), ozone (O_3), particulate matter (PMx), air temperature, and relative humidity. One fixed monitor was installed at the Start/Finish line and one mobile monitor followed the women elite runner pack. The data from the fixed monitors, deployed prior the race, described daily air pollution trends. Mobile monitors in combination with heatmap analysis facilitated the hyper-local characterization of athletes' exposures and helped identify local hotspots (e.g., areas prone to PM resuspension) which should be preferably bypassed. The estimation of inhaled doses disaggregated by gender and ventilation showed that doses inhaled by last

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https://doi.org/10.1016/j.scitotenv.2024.171997
Received 21 December 2023; Received in revised form 7 March 2024; Accepted 24 March 2024
Available online 31 March 2024

finishers may be equal or higher than those inhaled by first finishers for O_3 and PMx, due to longer exposures as well as the increase of these pollutants over time (e.g., 58.2 ± 9.6 and 72.1 ± 23.7 μg of $PM_{2.5}$ for first and last man during Rome marathon). Similarly, men received significantly higher doses than women due to their higher ventilation rate, with differences of 31–114 μg for NO_2, 79–232 μg for O_3, and 6–41 μg for PM_x. Finally, the aggregated data obtained during the 4 week- period prior the marathon can support better race scheduling by the organizers and provide actionable information to mitigate air pollution impacts on athletes' health and performance.

1. Introduction

Air pollution is known to cause cardiovascular and respiratory disease in general populations (Brunekreef et al., 2021; Chen and Hoek, 2020; Burnett et al., 2018; Cohen et al., 2017; Héroux et al., 2015; among others). The European Environment Agency estimated at least 307,000 premature deaths in the EU-27 in 2019 due to exposure to particulate matter ($PM_{2.5}$) above the World Health Organization (WHO) guideline level of 5 μg/m^3 (EEA, 2021). Nitrogen dioxide (NO_2) pollution caused 40,400 premature deaths in the EU and ozone (O_3) exposure caused 16,800 premature deaths (EEA, 2021). In urban settings, the health impacts from air pollutant exposures have been addressed in the scientific literature for the general population as a function of age groups, as well as for populations performing physical activities such as commuting by cycling or on foot (Cole-Hunter et al., 2018; de Nazelle et al., 2012; Tainio et al., 2021). Active populations performing physical activities, such as cycling and running, whether recreationally or professionally, are at higher risk of air pollutant side-effects than general populations due to increased ventilation (Alves Pasqua et al., 2018), which in addition changes from nasal to oral breathing patterns, resulting in particles bypassing the usual nasal filtration mechanism (Saibene et al., 1978; Niinimaa et al., 1980). Furthermore, the increased velocity with which pollutants are transported in the breathing flow carries them deeper into the respiratory tract and increases deposition (Sun et al., 2016; Zoladz and Nieckarz, 2021; Hussein et al., 2019). The ventilation rates are dependent on the type and intensity of exercise (Schiffer, 2008), and higher maximum ventilation levels achieved for elite athletes (Pasqua et al., 2018). Thus, populations performing physical activities have a higher air pollutant inhalation dose intake than non-active populations.

This is particularly relevant in endurance sports and, especially elite athletes, given the potential harmful effect of air pollution on both athletic performance and health (McKenzie and Boulet, 2008; Morici et al., 2020; Hodgson et al., 2021; Guo and Fu, 2019; Hofman et al., 2018; Luengo-Oroz and Reis, 2019; Reche et al., 2020; Zoladz and Nieckarz, 2021; Riediker, 2022; Cusick et al., 2023; Beavan et al., 2023). Several studies have associated exposure to O_3 and PM to reduced performance during exercise (Rundell and Sue-Chu, 2013; El Helou et al., 2012; Hodgson et al., 2022; Cusick et al., 2023), likely due to reduced lung function and the pro-inflammatory characteristics of these pollutants which have been correlated with apoptosis of neutrophils and bronchial epithelial cells (Chimenti et al., 2009). However, the number of studies available is scarce, probably due to the relatively limited number of individuals impacted by these exposures during elite athletic competitions. Conversely, one example of high-level international competitions, which involves a large number of athletes, is urban road races (major marathon races attracting up to 50,000 runners/race). The interest in urban marathons is rapidly increasing in large and densely populated cities around the globe (London, Berlin, New York, Tokyo, Chicago, Boston), generating significant revenue for the cities. Marathons attract elite as well as recreational athletes from around the world, in what is recently known as "marathon tourism". However, these major cities frequently suffer from high air pollution levels which impact athletes' health and performance (El Helou et al., 2012; Pascal et al., 2013). Marathon running has been linked to serious, acute functional respiratory issues during and after a marathon race (Tiller, 2019). This is caused by the stress the respiratory system is experiencing during competition, due to increased ventilation, which can exceed 110 L per minute in elite athletes (Hausswirth et al., 1997).

Several major and regional sports events have been affected in recent years by strong air pollution episodes (Viana et al., 2022, and references therein), and this is expected to increase in coming years, partly due to climate-change driven pollution. Thus, sport governing bodies and race organizers are starting to pay more attention to air pollution during sports events. However, they have little guidance to support events scheduling or protect runners, and guidelines to minimize air pollution exposure in sports are mostly non-existing (Bunds et al., 2019). One of the main limitations is the lack of local and high temporal resolution air quality data representative of exposure at competition venues and locations, which would facilitate selecting the optimal time of the day or route for competitions (e.g., road races). Regulatory air quality monitoring stations (EU Air Quality Directive 2008/50/EC) monitor air pollutant concentrations to which the general population is exposed at a low spatial resolution, and thus are generally not representative of specific locations. This is also an issue when estimating exposure to vulnerable populations such as in schools, hospitals or elderly homes. In the case of road races, 3 main challenges need to be considered if realistic exposures to air pollutants are to be quantified: 1) air quality data should be collected as close as possible to the runners and at high temporal resolution, 2) portable instruments are necessary to track exposure trends, and 3) average exposures are not representative for individual athletes, as runners have different performances and exposure times; thus, data acquisition time must be consistent with the runner's chronometric performance. This has a major influence on the inhalation doses received by the different athletes, given that they are exposed during different periods to the air pollutants and the fact that air pollutant concentrations vary throughout the day (i.e., O_3 concentrations increase towards midday). In this framework, mobile monitoring could address some of these challenges by tracking exposure at hyperlocal level during sport events, such as marathons. Air quality sensor-based monitoring systems are a non-regulatory technology, lower in cost, smaller and more portable than regulatory instrumentation (Clements et al., 2022; Castell et al., 2017; Peltier et al., 2021). Air sensors typically provide real-time data and allow measuring the air quality at higher spatial resolution than the regulatory stations (Clements et al., 2022). However, sensor technologies have well-known limitations (Bi et al., 2022; Borrego et al., 2016; Desouza et al., 2022; Gerboles et al., 2017; Jayaratne et al., 2018; Li et al., 2020; Ripoll et al., 2019; WMO, 2018) such as effects due to ambient conditions, cross-sensitivities and drifts over time, and the need for calibration under local ambient and aerosol mix conditions to ensure data quality. In this regard, the CEN working group WG42 is in the process of providing standardized guidance for sensor technologies (European Committee for Standardization, 2021), and tools are being developed to assess compliance of sensors with the data quality objectives of the EU Air Quality Directive (Yatkin et al., 2022). Specifically, in the framework of sports-related exposure monitoring, cost-effective air quality monitors have been tested in several indoor and ambient air settings including sports facilities (Viana et al., 2022; Reche et al., 2020; Sun et al., 2016).

This work aims to understand the air pollution exposures and dose inhaled by elite and recreational athletes during urban marathons, by implementing a combined mobile and fixed monitoring strategy. The final goal is to provide road race organizers with an evidence-based methodology to assess participant's exposure and use this information to mitigate the health risk.

2. Methodology

2.1. Strategy, instrumentation and monitoring locations

Air quality, temperature and humidity were monitored during three urban marathons in 2021 and 2022: Rome (19/09/2021) and Nice (28/11/2021) in Europe, and Eugene, Oregon, (17/07/2022) in USA. These events were selected because they are representative of different race configurations, types of environments (urban, rural, coastal) and pollution levels. A large metropolitan inland area (Rome) with relatively high pollutant levels, which exceeds yearly WHO guidelines for NO_2, O_3, C_6H_6 and PM_{10} and $PM_{2.5}$ (Battista et al., 2016). Two small to medium size urban area (OECD, 2023) with lower pollutant levels, one of them along the coast (Nice) with pollutants coming mostly from traffic and an airport (Mazenq et al., 2017; Suissa et al., 2013) and in the country-side (Eugene). The information regarding the locations and type of marathon is detailed in Table 1. The Eugene and Rome marathons were designed with circular routes, Rome with 1 loop and Eugene with 3 loops. Conversely, the Nice marathon was designed as an A to B race (starting in Nice and finishing in Cannes) running along the southeast coast of France. Whereas the Nice marathon runs along a highway, the Eugene and Rome marathons are urban. Both Nice and Rome marathons have been gathering over the last years >10,000 runners. The Eugene marathon was part of the 18th World Athletics Championships 2022, which gathered exclusively 63 elite participants from 33 countries. Finally, data quality was the last selection criterion applied. Data were collected during other marathons (e.g., Barcelona 2021 and Zagreb 2022) but are not reported in this work, because of insufficient data quality assurance. Specifically, the main issues affecting these monitoring campaigns were lack of enough time fort acclimatization of sensors to local conditions (due to late arrival of the sensors), short duration of the monitoring period (owing to lack of access to the start/finish lines) and/or incompleteness of the datasets (missing data due to technical difficulties). As a result, the data from these marathons were excluded from the analysis. In sum, based on data quality, population, air pollution and race design criteria, 3 different marathons were selected for the present work: an urban race with higher pollution levels and single-loop design (Rome), a more rural area with lower pollution levels and a multiple loop design (Oregon), and a seaside race along a highway (A-to-B race, in Nice).

The monitoring instruments used were commercial sensor-systems designed for fixed and mobile deployment. Both models integrate Alphasense monitors for $PM_{2.5}$ and PM_{10} (OPC–N3), NO_2 (NO2–B43F), and ozone (O_3; OX-B431), as well as sensors for temperature and relative humidity. The sensor-systems specifications are summarized in Supplementary Table S1. Complete technical specifications can be downloaded from the manufacturer's website (https://www.kunakair.com). One unit of each model was deployed respectively in two locations: a fixed/stationary and a mobile position. The fixed monitor was located at the starting point of the marathon, at a < 4 m distance from the start line where the athletes concentrated, representative of the air quality exposure concentrations at the start of the race. The same strategy was used in previous studies (Sun et al., 2016). The mobile monitor was mounted on the back of a bicycle which followed the leading female athlete group of the race throughout the entire racecourse (Supplementary Fig. S1) to be a closer representation of the athletes' exposures during the marathon. Thus, the fixed monitor was representative of the air pollutant concentrations at the start and finish of the marathons in Eugene and Rome (loop races), whereas it was only representative of the starting conditions in Nice (point A). The mobile monitor was representative of the exposure of the elite women athletes, while it was not able to fully represent that of the slower runners (the difference between the fastest and slowest athletes in urban marathons can be of 4 to 5 h). Finally, baseline air pollutant concentrations were recorded prior the race (4 weeks for Eugene and 1 week for Rome) with the fixed monitors to facilitate the understanding and representativity of the pollutant concentrations during the actual marathon day. This option was not possible for Nice, where, due to access limitations, the monitor was only installed 1 day prior to the marathon.

The monitors were calibrated by the manufacturer following their internal quality assurance/quality control (QA/QC) procedures, and were subsequently shipped to the marathon cities and installed by local staff. Once at their destination, an adjustment of the baseline and sensitivity (span) of the monitors was carried out remotely by the manufacturer to account for the different meteorological conditions they were deployed in. According to the experimental design, the units were not intercompared or calibrated against local air pollutant or meteorological reference data because this work aimed to test an air quality monitoring strategy for marathon organizers, not for air quality researchers or networks, and marathon organizers do not typically have access to local air quality monitoring networks. The aim of this work is to describe the spatio-temporal variability of air pollutants monitored with individual sensing units, in absence of comparability across units or with reference data. This approach was presented and discussed in detail in Viana et al. (2022). In this framework, local calibration is in general best practice when dealing with low-cost sensor-systems (WMO, 2018), and the lack thereof is acknowledged by the authors as a limitation of this work. However, the results in this work are assessed in terms of the temporal variability of air pollutants in each individual monitor, and the results are neither used for comparison across monitors nor for compliance-checking purposes. In addition, data quality was based on testing of the same monitors in the framework of two previous studies (Reche et al., 2020; Viana et al., 2022), where the performance of several units was tested for gaseous pollutants and PMx at the Barcelona EU-reference air quality monitoring station in Palau Reial during 30 and 5-day periods, respectively. In addition, one of the monitors was intercompared during 3 months at a local reference station in North America (the exact location cannot be disclosed due to confidentiality issues) following non-EU quality procedures. Results from Reche et al. (2020) and Viana et al. (2022), shown in Supplementary Fig. S2, evidenced statistically significant comparability between monitors and reference data for NO_2 ($R^2 = 0.94$ and 0.90) and O_3 ($R^2 = 0.93$ and 0.85), while the correlation was lower for PM_{10} ($R^2 = 0.70$) and $PM_{2.5}$ ($R^2 = 0.58$ and 0.82).

Table 1
Location, monitoring dates, type of marathon, start and finish times, and available fixed data prior to the race.

Location	Eugene	Rome	Nice
Country	USA	Italy	France
Date	17/07/2022	19/09/2021	28/11/2021
Start time	6:15	6:45	8:00
Finish time (1st elite man)	8:21	8:53	10:11
Finish time (1st elite woman)	8:33	9:14	10:56
Finishing time (Last athlete)	9:19	13:51	14:14
Marathon racecourse design	3 loops	1 loop	A to B
Data prior marathon	20/06/2022–16/07/2022	11/09/2021–18/09/2021	27/11/2021
Nr. valid datapoints (mobile monitor)	848 (10 s resolution)	900 (10 s resolution)	848 (10 s resolution)[a]
Nr. participants	63 professional participants	7500 registered participants	9500 registered participants

[a] Only 30 data points for $PM_{2.5}$ and PM_{10}.

2.2. Data analysis

Time series of PM_{10}, $PM_{2.5}$, NO_2 and O_3 were obtained for each marathon, from one fixed and one mobile monitor, with 5-minute and 10-second time-resolution, respectively. In total about 900 valid datapoints were obtained per pollutant and for each city with the mobile monitor (Table 1). Subsequently, 1-minute averages (except for Nice $PM_{2.5}$ and PM_{10}; 5-minute averages) were calculated for the mobile monitor data, and 5-minute averages (except for Oregon $PM_{2.5}$ and PM_{10}; 10-minute averages) were calculated for the fixed monitor. Hourly averages were calculated and plotted to assess the variability of air pollutant concentrations prior to the marathon, in order to understand the representativity of the concentrations recorded during the day of the marathon. Heatmaps were plotted to identify pollution hotspots linked to the design of the racecourse (e.g., street canyons, vicinity of major roads, etc.).

Inhalation doses were estimated based on the monitored concentrations and the exposure time, based on Eq. (1) (Borghi et al., 2021):

$$Dose\ (\mu g) = C_i\left(\frac{\mu g}{m^3}\right)*V_E*\left(\frac{m^3}{min}\right)*t\ (min) \tag{1}$$

where C_i is the concentration of the pollutant, V_E is the minute ventilation and t is the exposure time.

The minute ventilation (V_E) was estimated using Eq. (4), derived from Eqs. (2) and (3). Ventilation (V_E) was estimated based on the ventilatory equivalent for oxygen (ERO_2), defined as the ratio of the volume of air ventilating the lungs (V_E) to the volume of oxygen consumed (VO_2) (Eq. (2)), and the correlation between VO_2 and velocity (Eq. (3)) found in (Léger and Mercier, 1984; Ellens et al., 2019).

$$ERO_2\,(-) = \frac{V_E\ (L/min)}{VO_2\ (L/min)};\ V_E = ERO_2*VO_2 \tag{2}$$

$$VO_2(ml/kg/min) = 3.5*v\ (km/h) \tag{3}$$

$$V_E(l/min) = ERO_2*v*3.5*BW(kg)*0.001 \tag{4}$$

BW is the body weight expressed in kg, which was assumed to be 60 and 75 kg for women and men athletes, respectively. The ventilatory equivalent for oxygen ERO_2 was assumed to be 31 for the fastest man and woman (arriving 1st), and 28 for the last man and woman. Velocity v (km/h) was calculated based on the average finishing times in the three marathons (2.16 h for the first man, 2.5 h for the first woman, and 6.66 h for the last man and woman).

The total inhalation doses were estimated for each full marathon and separately for men and women athletes, based on the data from the fixed and mobile monitors to represent two different scenarios:

- Mobile monitor: was used to estimate the dose inhaled by elite athletes (1st man and woman athletes to finish the race). This was considered the most representative setup given that concentrations were recorded hyper-locally, in close proximity to the elite women athletes along the full racecourse. Mobile monitoring was only available for the elite (fastest) runners.
- Fixed monitor: in absence of mobile data for the slower athletes, the data from the fixed monitor were used to estimate their exposures assuming that the concentrations recorded at the Start line were representative of the full racecourse. While this is an over-simplification and a limitation, this approach was considered to be more adequate than using data from a local air quality monitoring station in each city, which would have been found at a larger distance from the racecourse.

Statistically significant differences between doses inhaled as a function of gender (1st man vs. 1st woman, and last man vs. last woman) were assessed per pollutant and location (mobile vs. fixed devices). Moreover, within the same gender, differences between 1st (elite) and last athletes per pollutant and location (fixed only) were assessed. Data normality and lognormality was assessed using the Shapiro-Wilk test. Overall data did not follow normal and lognormal distribution thus, differences between groups were assessed using a non-parametric ANOVA (Brown-Forsythe and Welch ANOVA, $P < 0.05$) followed by Dunnett's post-hoc multiple comparison test (alpha <0.05). GraphPad Prism version 10 GraphPad Software, San Diego, California USA) was used.

3. Results and discussion

3.1. Air quality baseline prior the marathons

The fixed monitors were deployed in Eugene and Rome, 4 and 1 weeks (respectively) prior to the marathons in each city. The purpose of this deployment was to determine the baseline air quality trends during the weeks before each event, to facilitate understanding the representativity of the actual marathon day in terms of air pollutant concentrations. It should be noted that the baseline aimed to represent only the weeks before the marathons, and was not meant to describe the annual variability of air pollutants in each city. The mean hourly evolution of NO_2, O_3, PM_{10} and $PM_{2.5}$ concentrations in each city is shown in Fig. 1.

Results show that, in both cities, the marathons took place on relatively low air pollution days, in comparison to the previous weeks. This may be an effect of the re-routing of the city traffic in preparation for the marathon. A notable reduction of pollutants was observed for O_3 concentrations in Rome, where a decreasing trend was observed between 11/09/2021 (reaching 140 μg/m^3 as daily maximum) and 19/09/2021, marathon day (reaching 70 μg/m^3 as daily maximum). When comparing to previous weekends, Eugene gases concentrations show comparable or higher concentrations, whereas PM concentrations show comparable or lower concentrations. In terms of hourly trends, NO_2 and O_3 concentrations followed the characteristic urban cycle in Eugene and Rome, with O_3 maxima coinciding with the highest insolation hours and an inverse NO_2 pattern (Clapp and Jenkin, 2001).

PM_{10} and $PM_{2.5}$ daily maximum concentrations were also lower in Eugene during the marathon day (7 μg/m^3 and 3 μg/m^3, respectively) when compared to previous weeks, when hourly maxima of 18 and 7 μg/m^3 were recorded at the location of the fixed monitor (Fig. 1). Hourly PM trends also followed certain patterns, even if they were not as marked as those observed for the gaseous pollutants, with a larger variability of hourly concentrations probably resulting from the variety of PM emission sources in each of the urban environments. As expected, PM concentrations were strongly influenced by meteorology (e.g., low concentrations during a rainfall episode in Eugene, between 5th and 6th of July 2022, Fig. 1c (Menne et al., 2012)) as well as by short-term local sources (e.g., peak PM_{10} and $PM_{2.5}$ concentrations recorded on 15/09/21 in Rome).

The baseline air pollutant concentrations described in Fig. 1 allowed us to conclude on the representativity of the air quality situation in each city during each of the marathons. In addition, monitoring hourly air pollutant trends over a representative period of time during the weeks before the marathon, provided added value in the form of targeted information which may be used by event organizers for scheduling the marathon and the activities around it to minimize athlete's air pollution exposures. This approach was previously proposed for athletics stadia to support scheduling of athletics competitions (Reche et al., 2020; Sun et al., 2016; Viana et al., 2022).

3.2. Temperature and humidity during marathons

Average temperature recorded during the marathons with the mobile monitor was 16.3 ± 0.5, 21.0 ± 0.4, and 7.0 ± 1.1 °C in Eugene, Rome, and Nice, respectively. Average humidity values recorded with the mobile monitor were 61.9 ± 2, 64.3 ± 2.4, and 42.7 ± 4.2 % for Eugene,

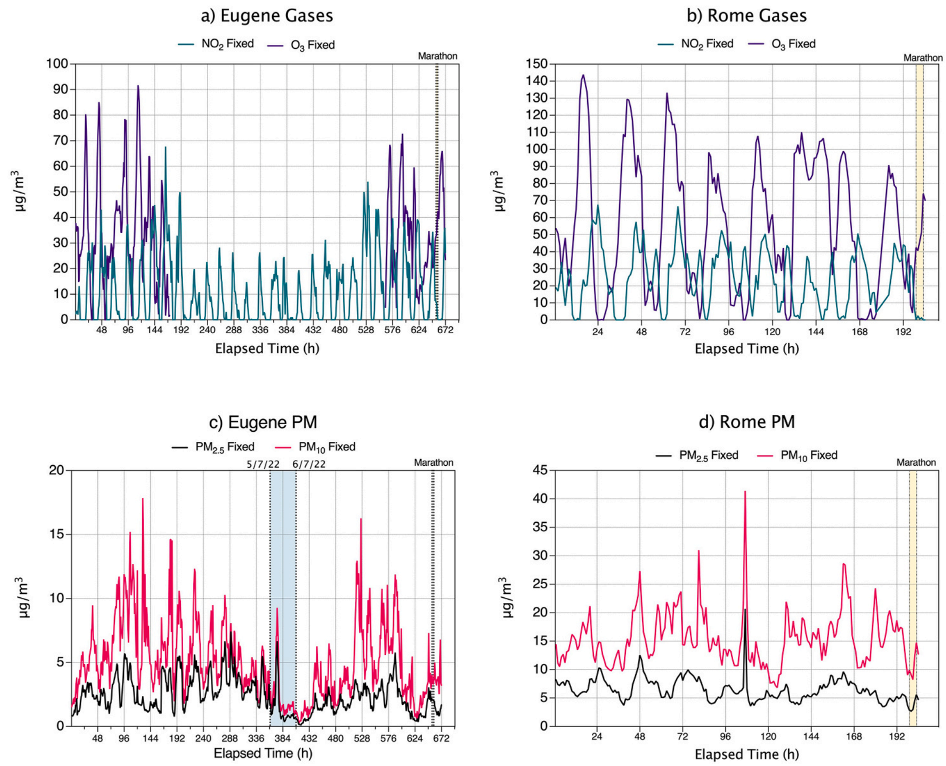

Fig. 1. Hourly evolution of NO_2, O_3, PM_{10} and $PM_{2.5}$ prior the marathon at the fixed position. The marathon start and finishing time is indicated in the time series with dotted lines and yellow shading. The start and finish of a rainfall episode is indicated with dotted lines and blue shading.

Rome, and Nice, respectively. Similar temperature and humidity (± 2) were recorded with the fixed monitors. High temperature and humidity during exercise have been linked to severe health problems (Zhao et al., 2013; Rodrigues Júnior et al., 2020; Hodgson et al., 2021, 2022), as well as decrease in athletic performance (Vihma, 2010; El Helou et al., 2012; Hodgson et al., 2022; Cusick et al., 2023). Thus, recording temperature and humidity at different points along the route, or using mobile monitors to track changes in these parameters before and during the marathon, is a simple way of providing essential information to ensure the safety of participants.

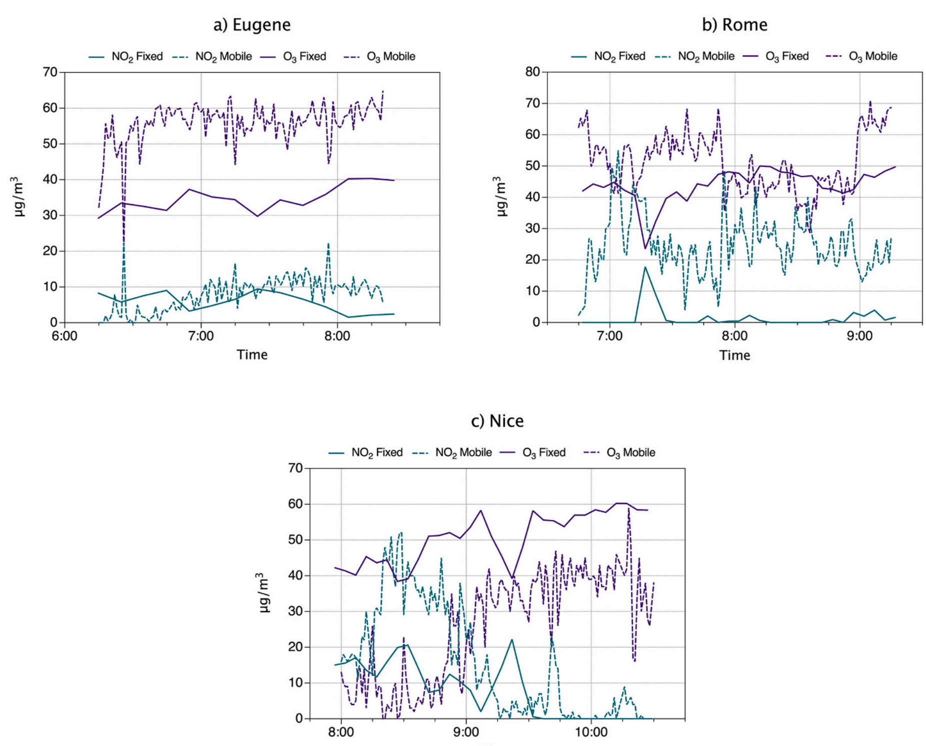

Fig. 2. NO_2 and O_3 time series during the three marathons monitored at the fixed and mobile locations.

3.3. Exposure to gaseous pollutants during marathons

Mean 10-min NO_2 and O_3 concentrations are shown for each marathon in Fig. 2, from the fixed and mobile monitors. In all three marathons, O_3 steadily increased over time with increasing ambient temperature, as expected, and independently of the monitor (fixed or mobile). This is especially clear for Nice where O_3 concentrations at the end of the marathon were 2 and 4 times higher than at the start according to the fixed and mobile monitors, respectively. This is linked to the photochemical formation of O_3 with increasing sunlight and temperatures, as well as to the rapid decrease of NO_2 (Fig. 2c). For longer races, the increase in O_3 concentrations towards the central hours of the day is a key challenge for the slower athletes as they receive higher doses of this hazardous pollutant than the elite athletes (who cross the finish line before ozone maxima are recorded).

The high variability in O_3 concentrations in Nice is evident in the heatmaps in Supplementary Fig. S3. The difference between the concentrations reported by the fixed and mobile monitors (e.g., the case of O_3 in Eugene) results from the fact that the monitors were not intercompared locally, as discussed in the Methods section.

3.4. Exposure to particulate pollutants during marathons

In the Eugene and Rome marathons, PM_{10} and $PM_{2.5}$ measured with both fixed and mobile monitors showed relatively constant concentrations across the duration of the race (Fig. 3a and b). The data from the mobile monitors, however, were markedly noisier and showed numerous peaks over time which were probably dependent on the influence from local sources at specific times and locations throughout the racecourse (Fig. 3 and heatmaps in Fig. 4). The effects of local sources on PM can be clearly observed in the heatmaps (e.g., Eugene Fig. 4). Overall, $PM_{2.5}$ and PM_{10} followed quite similar temporal patterns, especially in Rome. A few exceptions were recorded with the mobile monitors, where PM_{10} showed clear increases in concentrations not observed for $PM_{2.5}$ (e.g., Eugene at 7:15 to 7:30, and 8:15, and Rome at 6:55 and 7:45 to 8:00; Fig. 3a and b) which were probably linked to local dust re-suspension by the runners and/or other participants/visitors. In the case of Eugene, a 3-loop marathon, it is interesting to observe how PM_{10} and $PM_{2.5}$ concentrations peaked at 3 moments in time coinciding with approximately 10 min after the start of the marathon, 50 and 90 min later. Considering that the racecourse was a loop, these results suggest that the increases were caused by a specific source where the passage of the runners increased PM concentrations (an unpaved area prone to dust resuspension, for example, given that this trend was not observed for gaseous pollutants, Fig. 2 and Fig. 4). This can be clearly identified by means of the heatmaps (Fig. 4), which demonstrates a clear $PM_{2.5}$ hotspot when the athletes ran through two unpaved park areas, and likely the cause of the increased $PM_{2.5}$ concentrations monitored. Similar results were obtained for PM_{10}. These results show the potential of hyper-local monitoring to identify air pollutant hotspots along the racecourse, and thus provide actionable information for race organizers to minimize athletes' exposures to air pollutants by altering the design of the course.

In Rome, the highest PM_{10} and $PM_{2.5}$ concentrations were monitored at the start of the marathon. From that moment onwards, concentrations decreased and remained mostly constant with the exception of certain local concentration peaks. In Nice, the only marathon studied with an "A to B" configuration (as opposed to a loop, the cases of Rome and Eugene), particle concentrations from the mobile monitor remained relatively constant over time, with a slight tendency to increase towards the finish line. The fixed monitor, on the other hand, showed a steep increase of both PM fractions at around 9:15, not visible in the mobile unit. Once again, this result shows the major differences between exposure concentrations estimated based on fixed vs. mobile monitors, evidencing the inability of the fixed monitor to reproduce the hyper-local exposure concentrations monitored by the mobile monitor, due to the influence of emissions in direct proximity to the monitor.

To better understand the variability in PM sources and the differences between fixed and mobile measurements, $PM_{2.5}/PM_{10}$ ratios were calculated. Whereas absolute PM concentrations should not be compared between the fixed and mobile monitors across cities because they were not calibrated locally (see section Methods), the $PM_{2.5}/PM_{10}$

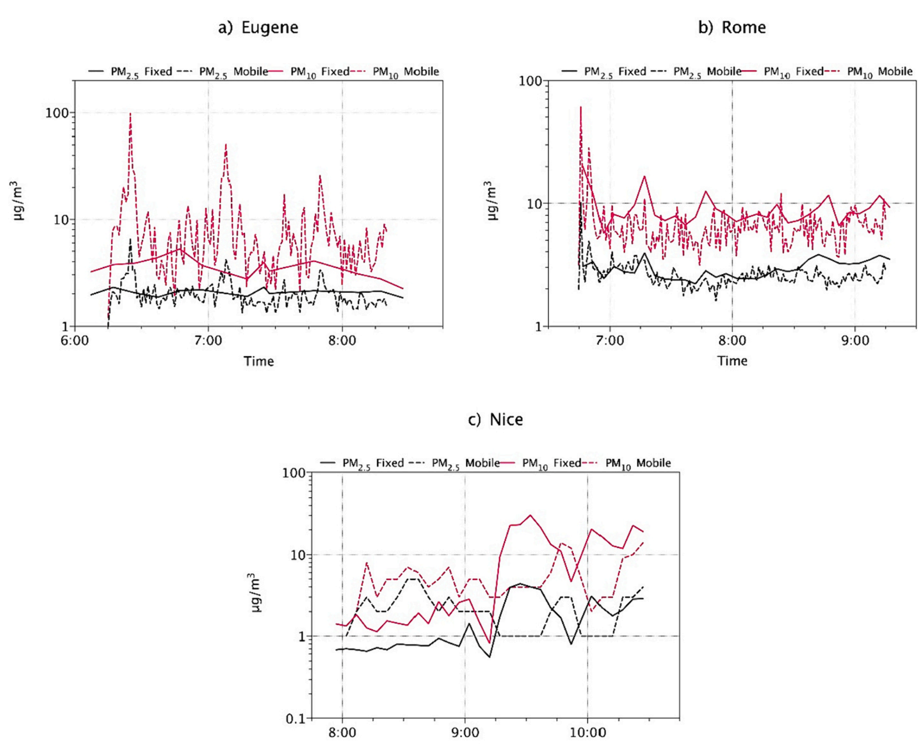

Fig. 3. PM_{10} and $PM_{2.5}$ evolution during marathon at the fixed and mobile positions.

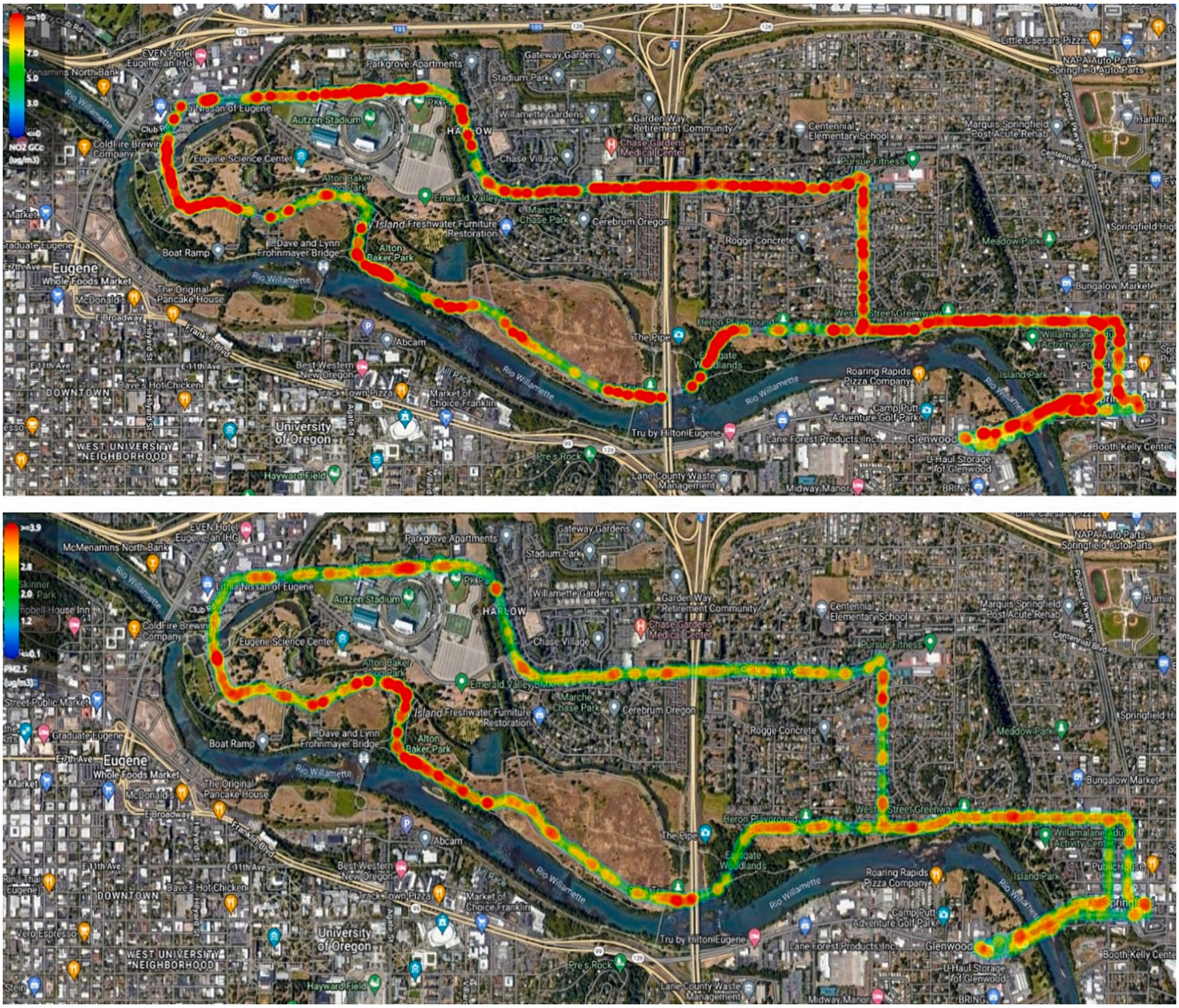

Fig. 4. NO_2 (top) and $PM_{2.5}$ (bottom) concentrations during Eugene, OR marathon.

ratios may be considered comparable. In Rome, the mean $PM_{2.5}/PM_{10}$ ratios for the full race were similar for both monitors (0.32 for the fixed and 0.38 for the mobile monitor). Conversely, mean $PM_{2.5}/PM_{10}$ ratios were markedly different in Nice (0.19 fixed, 0.40 mobile) and Eugene (0.58 fixed and 0.23 mobile). These results demonstrate clear differences in the type of aerosols the mobile and fixed monitors were challenged with, which showed different size distributions and probably originated from different emission sources. As a result, because the mobile monitors reported exposures in close proximity to the (elite) runners, it may be concluded that the data generated by the fixed monitors were not as representative of athlete exposures as those from the mobile monitors. Whereas the data from the fixed monitors was probably more representative of local exposure variability (not absolute concentrations) than those generated by central air quality monitoring stations in each of the cities (Viana et al., 2022), results evidence that the mobile data provide higher added value when assessing the potential health and performance impacts of inhaled particles.

3.5. Air pollutants inhalation doses during marathons

Inhalation doses of air pollutants throughout the duration of the marathon were estimated based on Eqs. (1)–(4), disaggregated by gender, with the aim of understanding the differential impact of air pollutant exposures on athletes' health. The fact that data were collected in close proximity to the athletes (specifically, the elite runners) provided added value given that inhalation doses could also be estimated as a function of the duration of the race and taking into account the influence of local emission sources (e.g., the potential area with high dust re-suspension identified in Eugene; Fig. 3 and Fig. 4).

Inhalation dose results are shown in Fig. 5 and Supplementary Table S2, for each pollutant separately. According to this analysis, the trends obtained in terms of air pollutant potential doses inhaled varied largely across marathons. Overall, for the elite athletes and using mobile monitoring data, air pollutant potential doses inhaled were consistently and statistically significantly lower, except in Nice for O_3 and PM_x pollutants, for the women athletes due to their lower ventilation rate (V_E) which in turn depends on the velocity and gender. When comparing the effect of the total run time, i.e., the potential doses received by elite runners in comparison to recreational athletes (using fixed monitoring data and assuming fixed concentrations to be representative of average concentrations throughout the racecourse, which is a limitation of this approach), a larger variability was observed and only statically significant differences were observed for $PM_{2.5}$ in Rome (58.2 ± 9.6 μg elite vs. 72.1 ± 23.7 μg last men athletes and 48.1 ± 8.0 μg elite vs. 57.6 ± 19.0 μg last women athletes) and O_3 in Nice (845.2 ± 112.5 μg elite vs. 775.4 ± 91.9 μg last women athletes).

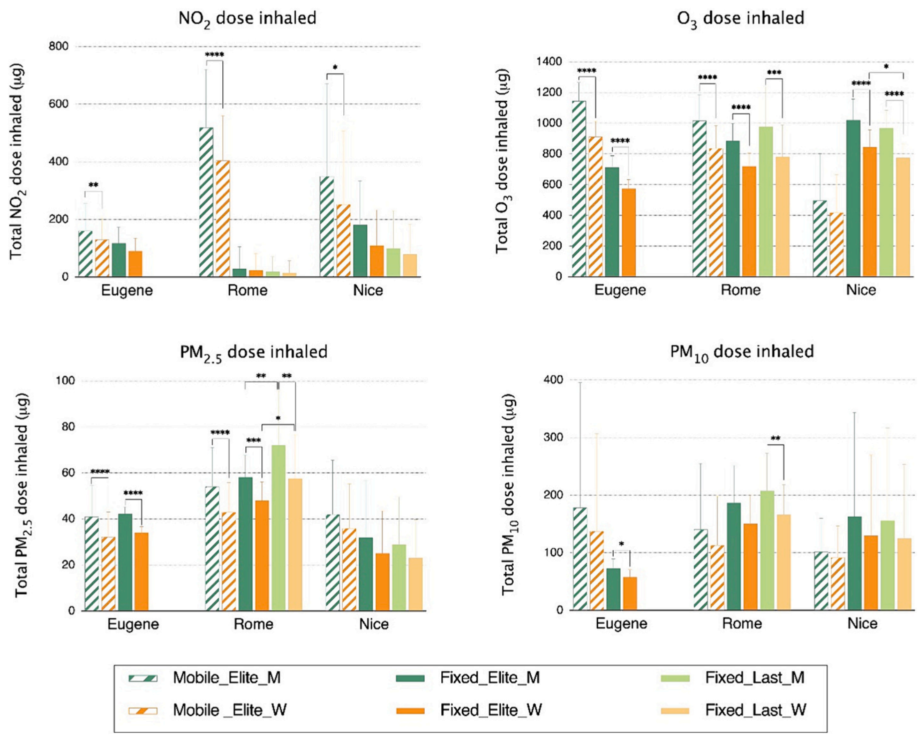

Fig. 5. Inhalation doses estimation (µg) for NO_2, O_3, PM_{10} and $PM_{2.5}$ for the full duration of each marathon for elite (fastest) and last (slowest) athletes, disaggregated by gender (M: men; W: women). * Denotes statistical differences between groups.

For NO_2 exposures (Fig. 5 and Supplementary Table S2), potential inhalation doses for women were statistically significantly lower than for men (130.3 ± 73.3 µg vs. 160.8 ± 94.1 µg in Eugene, 404.9 ± 155.2 µg vs. 519.0 ± 201.6 µg in Rome, and 252.5 ± 254.6 µg vs. 349.6 ± 320.9 µg in Nice), and the faster athletes received higher doses than the slower ones. This is explained by the decreasing NO_2 concentrations over time from the start of the marathon (Fig. 2) and higher ventilation displayed by faster runners. The slower runners were exposed to lower NO_2 concentrations over a longer period of time. These results were consistent for the three marathons, irrespective of the absolute concentrations monitored. Conversely, the longer exposure time of slower runners was detrimental in terms of potential inhaled dose for other pollutants such as O_3 and PMx, with increasing concentrations over time. Clear examples of this are observed in Rome, where the slower runners received estimated inhalation doses of $PM_{2.5}$ of up to 72.1 ± 23.7 µg (men) and 57.6 ± 19.0 µg (women) whereas the faster runners received significantly lower doses of 58.2 ± 9.6 µg and 48.1 ± 8.0 µg (men and women, respectively; Fig. 5 and Table S2). Similar patterns were obtained for O_3 and PM_{10} dose during the Rome marathon. This effect (higher exposure doses on slower runners due to increase of concentrations over time) was likely underestimated given the fact that inhaled doses of the slowest runners could not be calculated considering the actual running period (e.g., 5 h used, instead of 6 h). Thus, for those pollutants with an increasing trend such as O_3 and PMx, the differences between faster and slower runners were likely even higher than calculated and shown in Fig. 5.

As a result, it may be concluded that, at equal ventilation and gender but different speed and exposure concentrations (e.g., fastest man vs. last man and fastest woman vs. last woman), potential inhalation doses of last finishers may be equal or higher than those of first finishers for certain pollutants. These results are especially relevant for marathons taking place in cities with high mean O_3 and PM concentrations, and should be taken into account for marathon scheduling (by carefully selecting the period of the year and time of day with the lowest ambient O_3 concentrations).

These findings are in line with current available literature. Both, the potential of first finishers to be exposed to higher doses of pollutants due to increased ventilation, and increased exposure doses to pollutants which increase over time have been previously discussed (Hodgson et al., 2021, 2022; Viana et al., 2022). The current results show how both (ventilation and changes in pollutant concentrations) are relevant in the athletes' total exposure doses. This should be taken into consideration when planning a marathon, and select the hours of lowest gaseous and PM pollutants considering first and last finishers, given that several authors have reported effects on performance (Cusick et al., 2023; El Helou et al., 2012; Hodgson et al., 2021; Rundell et al., 2018; Rundell and Sue-Chu, 2013) as well as health impacts (Tiller, 2019). In the particular cases of this study, the advice would be to run the marathons early in the morning with last finishers ending no later than midday to avoid O_3 and PM peaks. This is already the case for Eugene and Rome, however, in Nice, last arriving runners would be exposed to the potential O_3 and PM concentration peaks. These results highlight the benefits which would derive from expanding the presented methodology and using two mobile monitors, one following first elite woman and a second one following last finisher. Future work would benefit from estimating, in addition to inhalation doses, PM deposition in the different regions (head-airways, tracheobronchial and alveolar) of the athletes' respiratory tracts by collecting particle size distribution data. While this would require the use of more sophisticated instrumentation (i.e., particle sizers), it would provide significant added value to understand the hazardous potential of inhaled particles as they deposit along different regions of the respiratory tract, by linking inhalation deposition modelling with biological effects. Regional inhalation deposition models have been successfully applied for this kind of assessment in previous works (Martins et al., 2015; Ribalta et al., 2019, among others). Individual GPS and physical activity data (e.g., heart rate) tracking would also increase the accuracy of dose estimation.

As long as the necessary QA/QC procedures are implemented to ensure data quality, portable monitors have been proven useful in different environments such as disadvantaged communities (Lu et al., 2022; Caplin et al., 2019), children in schools (Varaden et al., 2021; Chen et al., 2020), hospitals (Palmisani et al., 2021), or commuters (Motlagh et al., 2021). Specifically, in the sports community, the use of these monitors has been previously tested in athletic competitions such as in Athletics stadia or marathons (Viana et al., 2022; Reche et al., 2020; Sun et al., 2016). Previous exposure characterization during marathon running has been conducted using hourly air quality data from urban background stations (Hodgson et al., 2021, 2022) or static sensors deployed across the marathon route (Sun et al., 2016). In this work we propose a combination of fixed and mobile monitors to better understand and characterize runners' exposure during marathons at hyper-local level and with high time-resolution. The proposed methodology can be applicable to other outdoor sports activities which require mobile hyper-local data due to continuous movement such as different running competitions or cycling races.

A number of key limitations of this approach and potential improvements for future studies should be highlighted:

- Lack of local calibration of the mobile and fixed monitors: none of the monitors were calibrated locally against reference air quality monitoring instrumentation; they were only adjusted remotely to account for the local environmental conditions. This implies that the concentrations reported should not be assessed in absolute terms or compared with regulatory limit values. However, it is noted that this was not the aim of this work, which aimed to focus on the temporal variability of the pollutants monitored. In addition, due to the applied nature of this approach, where the air quality monitors are meant to be used by event organizers, it does not seem likely that the monitors would be locally calibrated following a scientific intercomparison. Therefore, this limitation was intrinsic to the study design of this work.

- Lack of local intercomparison between the fixed and mobile monitors in each city: future work should include the intercomparison between the fixed and mobile monitors prior to the start of the marathon, which would enable the quantitative estimation of the added value of using the mobile monitor by comparing the inhaled doses based on the mobile monitor and those based on the fixed monitor.
- Lack of drift assessment for the individual sensors: while the study period was short for each marathon (between 2 days and 4 weeks), drifts in sensor performance may occur, which should be accounted for. Drifts typically occur over longer periods of time (>1 year; WMO, 2018; Spinelle et al., 2015; Ripoll et al., 2019).
- Improved estimation of the dose inhaled by the slowest athletes: the inhaled dose of the slowest runners could not be calculated considering the actual running period (e.g., 5 h used, instead of 6 h). For future work and a better estimation of actual doses of the slowest athletes, the fixed monitor should be kept until the last runner finishes.
- Improved estimation of exposure of all types of athletes: a second mobile monitor following the 'tail/last' runners would help provide a better perspective of the exposure of all athletes.

4. Conclusions

This study aimed to describe the exposure concentrations and air pollutant doses inhaled by athletes during marathons, by means of a combined fixed and mobile monitoring strategy. Three marathons in three different cities were the subjects of study. The results from this work may be applied to provide sports events or marathon organizers with a strategy to assess participant's exposure to particulate and gaseous air pollutants.

The use of fixed monitors prior the race (longer-term monitoring) can help understand the local air pollutant concentrations baseline and during the marathon day. For both cities in which fixed monitors were installed prior the race, marathons took place on relatively low air pollution days (probably owing to traffic restrictions in view of the marathons). An additional benefit of understanding the local daily hourly pollutants evolution is that this information may be used for event organizers to schedule and plan the race minimizing pollutant exposures.

The mobile monitors were able to capture pollutant concentrations hyper-locally and identify pollutant hotspots along the races, not identifiable by the fixed monitor. The high resolution heatmaps generated were key to understand and identify the sources of the pollutants. In combination, they provide relevant information for race organizers to identify racecourse areas which should preferably be bypassed.

Inhaled doses of air pollutants were estimated as a function of gender and running speed, with the aim of understanding the differential impact of air pollutant exposures on athletes' health. The fact that pollutant concentrations were measured close to the athletes provided additional value since inhalation doses were estimated as a function of duration and considering local emission sources. As expected, athletes with higher ventilation rates (e.g., men vs. women) inhaled higher doses. Conversely, at equal ventilation capacity and gender, last finishers were exposed to equal or higher doses of certain pollutants such as O_3 and PM, which increase over time, than faster runners. Thus, this shows the relevance of 1) understanding pollutant hourly variability in order to schedule the races to avoid air pollutant peaks (for both faster and slower runners), and 2) monitoring pollutant concentrations using one or preferably 2 mobile monitors, which would allow to estimate inhaled doses of the faster and slower athletes. An improvement to this methodology would be to monitor particle size distribution data, which would allow to estimate particle deposition across different areas of the respiratory tract and link to health effects.

The main limitation stems from the fact that the air quality monitors were not calibrated locally nor intercompared with each other. Thus, pollutant concentrations should not be assessed in absolute terms or compared to regulatory values, and should only be used to assess temporal variability of pollutant concentrations. This is intrinsic to the study design.

Funding

This study was partly funded by the World Athletics Health and Science department, the Spanish Ministry of Science and Innovation (Project CEX2018-000794-S) and by AGAUR (project 2017 SGR41).

CRediT authorship contribution statement

Carla Ribalta: Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis, Data curation, Conceptualization. **Fréderic Garrandes:** Writing – review & editing, Supervision, Project administration, Methodology, Funding acquisition, Formal analysis, Conceptualization. **Stéphane Bermon:** Writing – review & editing, Supervision, Methodology, Funding acquisition, Conceptualization. **Paolo Emilio Adami:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Edurne Ibarrola-Ulzurrun:** Writing – review & editing, Visualization, Formal analysis, Data curation. **Ioar Rivas:** Writing – review & editing. **Mar Viana:** Writing – review & editing, Writing – original draft, Supervision, Methodology, Funding acquisition, Formal analysis, Conceptualization.

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.

Data availability

Data will be made available on request.

Acknowledgements

We thank Roma, Eugene and Nice marathon organizers for facilitating the air quality data collection.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.171997.

References

Battista, G., Pagliaroli, T., Mauri, L., Basilicata, C., De Lieto Vollaro, R., 2016. Assessment of the air pollution level in the city of Rome (Italy). Sustainability 8 (9), 838. https://doi.org/10.3390/su8090838.

Beavan, A., Härtel, S., Spielmann, J., Koehle, M., 2023. Air pollution and elite adolescent soccer players' performance and well-being; an observational study. Environ. Int. 175, 107943.

Bi, J., Carmona, N., Blanco, M.N., Gassett, A.J., Seto, E., Szpiro, A.A., Larson, T.V., Sampson, P.D., Kaufman, J.D., Sheppard, L., 2022. Publicly available low-cost sensor measurements for PM2.5 exposure modeling: guidance for monitor deployment and data selection. Environ. Int. 158, 106897 https://doi.org/10.1016/j.envint.2021.106897.

Borrego, C., Costa, A.M., Ginja, J., Amorim, M., Coutinho, M., Karatzas, K., Sioumis, Th., Katsifarakis, N., Konstantinidis, K., De Vito, S., Esposito, E., Smith, P., André, N., Gérard, P., Francis, L.A., Castell, N., Schneider, P., Viana, M., Minguillón, M.C., Reimringer, W., Otjes, R.P., von Sicard, O., Pohle, R., Elen, B., Suriano, D., Pfister, V., Prato, M., Dipinto, S., Penza, M., 2016. Assessment of air quality microsensors versus reference methods: the EuNetAir joint exercise. Atmos. Environ. 147, 246–263. https://doi.org/10.1016/J.ATMOSENV.2016.09.050.

Borghi, F., Spinazzè, A., Mandaglio, S., Fanti, G., Campagnolo, D., Rovelli, S., Keller, M., Cattaneo, A., Cavallo, D.M., 2021. Estimation of the Inhaled Dose of Pollutants in Different Micro-Environments: A Systematic Review of the Literature. Toxics 9, 140. https://doi.org/10.3390/toxics9060140.

Brunekreef, B., Strak, M., Chen, J., Andersen, Z.J., Atkinson, R., Bauwelinck, M., Hoek, G., 2021. Mortality and Morbidity Effects of Long-term Exposure to Low-level PM2. 5, Bc, NO2, and O3: An Analysis of European Cohorts in the ELAPSE Project. Research Reports, Health Effects Institute, 2021.

Bunds, K.S., Casper, J.M., Frey, H.C., Barrett, M., 2019. Air pollution at college football games: developing a methodology for measuring air pollutant exposure in a sport event microenvironment. Event Manag. 23 (3), 399–412.

Burnett, R., Chen, H., Szyszkowicz, M., Fann, N., Hubbell, B., Pope III, C.A., Spadaro, J. V., 2018. Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. Proc. Natl. Acad. Sci. 115 (38), 9592–9597.

Caplin, A., Ghandehari, M., Lim, C., Glimcher, P., Thurston, G., 2019. Advancing environmental exposure assessment science to benefit society. Nat. Commun. 10 (1), 1236.

Castell, N., Dauge, F.R., Schneider, P., Vogt, M., Lerner, U., Fishbain, B., Bartonova, A., 2017. Can commercial low-cost sensor platforms contribute to air quality monitoring and exposure estimates? Environ. Int. 99, 293–302.

Chen, J., Hoek, G., 2020. Long-term exposure to PM and all-cause and cause-specific mortality: a systematic review and meta-analysis. Environ. Int. 143, 105974.

Chen, L.W.A., Olawepo, J.O., Bonanno, F., Gebreselassie, A., Zhang, M., 2020. Schoolchildren's exposure to PM 2.5: a student club–based air quality monitoring campaign using low-cost sensors. Air Qual. Atmos. Health 13, 543–551.

Chimenti, L., Morici, G., Paternò, A., Bonanno, A., Vultaggio, M., Bellia, V., Bonsignore, M.R., 2009. Environmental conditions, air pollutants, and airway cells in runners: a longitudinal field study. J. Sports Sci. 27 (9), 925–935.

Clapp, L.J., Jenkin, M.E., 2001. Analysis of the relationship between ambient levels of O3, NO2 and NO as a function of NOx in the UK. Atmos. Environ. 35 (36), 6391–6405.

Clements, A., Duvall, R., Green, D., Dye, T., 2022. The Enhanced Air Sensor Guidebook. Washington, DC, USA, US Environmental Protection Agency.

Cohen, A.J., Brauer, M., Burnett, R., Anderson, H.R., Frostad, J., Estep, K., Forouzanfar, M.H., 2017. Estimates and 25-year trends of the global burden of disease attributable to ambient air pollution: an analysis of data from the Global Burden of Diseases Study 2015. Lancet 389 (10082), 1907–1918.

Cole-Hunter, T., de Nazelle, A., Donaire-Gonzalez, D., Kubesch, N., Carrasco-Turigas, G., Matt, F., Foraster, M., Martínez, T., Ambros, A., Cirach, M., Martinez, D., Belmonte, J., Nieuwenhuijsen, M., 2018. Estimated effects of air pollution and space-time-activity on cardiopulmonary outcomes in healthy adults: a repeated measures study. Environ. Int. 111, 247–259. https://doi.org/10.1016/j.envint.2017.11.024.

Cusick, M., Rowland, S.T., DeFelice, N., 2023. Impact of air pollution on running performance. Sci. Rep. 13 (1), 1832.

Desouza, P., Kahn, R., Stockman, T., Obermann, W., Crawford, B., Wang, A., Crooks, J., Li, J., Kinney, P., 2022. Calibrating networks of low-cost air quality sensors. Atmos. Meas. Tech. 15, 6309–6328. https://doi.org/10.5194/amt-15-6309-2022.

El Helou, N., Tafflet, M., Berthelot, G., Tolaini, J., Marc, A., Guillaume, M., Hausswirth, C., Toussaint, J.F., 2012. Impact of environmental parameters on Marathon running performance. PLoS One 7. https://doi.org/10.1371/journal.pone.0037407.

Ellens, T., Kaur, R., Roehl, K., Dubosky, M., Vines, D.L., 2019. Ventilatory equivalent for oxygen as an extubation outcome predictor: a pilot study. Canad. J. Resp. Therap. 55, 65.

European Committee for Standardization, 2021. Air quality - performance evaluation of air quality sensor systems - part 1: gaseous pollutants in ambient air. In: CEN/TS 17660-1:2021 (https://dx.doi.org/10.31030/3260371).

European Environment Agency, 2021. Air quality in Europe 2021 - Health impacts of air pollution in Europe, 2021. In: Briefing no. 19/2021. Published 15 Nov 2021 - Last modified 09 Feb 2023. Accessed: 13/11/2023. https://doi.org/10.2800/08097.

Gerboles, M., Spinelle, L., Borowiak, A., 2017. Measuring Air Pollution with Low-cost Sensors. Thoughts on the Quality of Data Measured by Sensors. Joint Research Centre, EU Science Hub: Brussels, Belgium.

Guo, M., Fu, S., 2019. Running with a mask? The effect of air pollution on marathon runners' performance. J. Sports Econ. 20 (7), 903–928.

Hausswirth, C., Bigard, A.X., Guezennec, C.Y., 1997. Relationships between running mechanics and energy cost of running at the end of a triathlon and a marathon. Int. J. Sports Med. 18 (05), 330–339.

Héroux, M.E., Anderson, H.R., Atkinson, R., Brunekreef, B., Cohen, A., Forastiere, F., Walton, H., 2015. Quantifying the health impacts of ambient air pollutants: recommendations of a WHO/Europe project. Int. J. Public Health 60, 619–627.

Hodgson, J.R., Chapman, L., Pope, F.D., 2021. The Diamond League athletic series: does the air quality sparkle? Int. J. Biometeorol. https://doi.org/10.1007/s00484-021-02114-z.

Hodgson, J.R., Chapman, L., Pope, F.D., 2022. Amateur runners more influenced than elite runners by temperature and air pollution during the UK's Great North Run half marathon. Sci. Total Environ. 842, 156825.

Hofman, J., Samson, R., Joosen, S., Blust, R., Lenaerts, S., 2018. Cyclist exposure to black carbon, ultrafine particles and heavy metals: an experimental study along two commuting routes near Antwerp, Belgium. Environ. Res. 164, 530–538.

Hussein, T., Saleh, S.S.A., dos Santos, V.N., Boor, B.E., Koivisto, A.J., Löndahl, J., 2019. Regional inhaled deposited dose of urban aerosols in an eastern Mediterranean city. Atmosphere 10 (9), 530.

Jayaratne, R., Liu, X., Thai, P., Dunbabin, M., Morawska, L., 2018. The influence of humidity on the performance of a low-cost air particle mass sensor and the effect of atmospheric fog. Atmos. Meas. Tech. 11, 4883–4890. https://doi.org/10.5194/amt-11-4883-2018.

Júnior, J.F.C.R., Mckenna, Z., Amorim, F.T., Sena, A.F.D.C., Mendes, T.T., Veneroso, C. E., Júnior, M.N.S.D.O., 2020. Thermoregulatory and metabolic responses to a half-marathon run in hot, humid conditions. J. Therm. Biol. 93, 102734.

Léger, L., Mercier, D., 1984. Gross energy cost of horizontal treadmill and track running. Sports Med. 1, 270–277.

Li, J., Zhang, H., Chao, C.Y., Chien, C.H., Wu, C.Y., Luo, C.H., Chen, L.J., Biswas, P., 2020. Integrating low-cost air quality sensor networks with fixed and satellite monitoring systems to study ground-level PM2.5. Atmos. Environ. 223 https://doi.org/10.1016/j.atmosenv.2020.117293.

Lu, T., Liu, Y., Garcia, A., Wang, M., Li, Y., Bravo-Villasenor, G., Han, B., 2022. Leveraging citizen science and low-cost sensors to characterize air pollution exposure of disadvantaged communities in Southern California. Int. J. Environ. Res. Public Health 19 (14), 8777.

Luengo-Oroz, J., Reis, S., 2019. Assessment of cyclists' exposure to ultrafine particles along alternative commuting routes in Edinburgh. Atmos. Pollut. Res. 10 (4), 1148–1158.

Martins, V., Minguillón, M.C., Moreno, T., Querol, X., de Miguel, E., Capdevila, M., Lazaridis, M., 2015. Deposition of aerosol particles from a subway microenvironment in the human respiratory tract. J. Aerosol Sci. 90, 103–113.

Mazenq, J., Dubus, J.C., Gaudart, J., Charpin, D., Nougairede, A., Viudes, G., Noel, G., 2017. Air pollution and children's asthma-related emergency hospital visits in southeastern France. Eur. J. Pediatr. 176, 705–711. https://doi.org/10.1007/s00431-017-2900-5.

McKenzie, D.C., Boulet, L.P., 2008. Asthma, outdoor air quality and the Olympic Games. CMAJ 179 (6), 543–548.

Menne, M.J., Durre, I., Korzeniewski, B., McNeill, S., Thomas, K., Yin, X., Anthony, S., Ray, R., Vose, R.S., Gleason, B.E., Houston, T.G., 2012. Global Historical Climatology Network - Daily (GHCN-Daily), Version 3. [Eugene, OR US Jul 2022 subset]. NOAA National Climatic Data Center. https://doi.org/10.7289/V5D21VHZ [18/09/2023].

Morici, G., Cibella, F., Cogo, A., Palange, P., Bonsignore, M.R., 2020. Respiratory effects of exposure to traffic-related air pollutants during exercise. Front. Public Health 8, 575137.

Motlagh, N.H., Zaidan, M.A., Fung, P.L., Lagerspetz, E., Aula, K., Varjonen, S., Tarkoma, S., 2021. Transit pollution exposure monitoring using low-cost wearable sensors. Transp. Res. Part D: Transp. Environ. 98, 102981.

de Nazelle, A., Fruin, S., Westerdahl, D., Martinez, D., Ripoll, A., Kubesch, N., Nieuwenhuijsen, M., 2012. A travel mode comparison of commuters' exposures to air pollutants in Barcelona. Atmos. Environ. 59, 151–159. https://doi.org/10.1016/j.atmosenv.2012.05.013.

Niinimaa, V.P.S.R.J., Cole, P., Mintz, S., Shephard, R.J., 1980. The switching point from nasal to oronasal breathing. Respir. Physiol. 42 (1), 61–71.

OECD, 2023. Urban Population by City Size (Indicator). https://doi.org/10.1787/b4332f92-en (Accessed on 12 September 2023).

Palmisani, J., Di Gilio, A., Viana, M., de Gennaro, G., Ferro, A., 2021. Indoor air quality evaluation in oncology units at two European hospitals: low-cost sensors for TVOCs, PM2. 5 and CO2 real-time monitoring. Build. Environ. 205, 108237.

Pascal, M., Corso, M., Chanel, O., Declercq, C., Badaloni, C., Cesaroni, G., Aphekom Group, 2013. Assessing the public health impacts of urban air pollution in 25 European cities: results of the Aphekom project. Sci. Total Environ. 449, 390–400.

Pasqua, L.A., Damasceno, M.V., Cruz, R., Matsuda, M., Garcia Martins, M., Lima-Silva, A. E., Bertuzzi, R., 2018. Exercising in air pollution: the cleanest versus dirtiest cities challenge. Int. J. Environ. Res. Public Health 15 (7), 1502.

Peltier, R.E., Castell, N., Clements, A.L., Dye, T., Hüglin, C., Kroll, J.H., von Schneidemesser, E., 2021. An Update on Low-cost Sensors for the Measurement of Atmospheric Composition (December 2020).

Reche, C., Viana, M., van Drooge, B.L., Fernández, F.J., Escribano, M., Castaño-Vinyals, G., Nieuwenhuijsen, M., Adami, P.E., Bermon, S., 2020. Athletes' exposure to air pollution during world athletics relays: a pilot study. Sci. Total Environ. 717 https://doi.org/10.1016/j.scitotenv.2020.137161.

Ribalta, C., López-Lilao, A., Estupiñá, S., Fonseca, A.S., Tobías, A., García-Cobos, A., Viana, M., 2019. Health risk assessment from exposure to particles during packing in working environments. Sci. Total Environ. 671, 474–487.

Riediker, M., 2022. Low transmission of coronavirus via aerosols during outdoor running races and athletic events. Aerosol Air Qual. Res. 22, 220069 https://doi.org/10.4209/aaqr.220069.

Ripoll, A., Viana, M., Padrosa, M., Querol, X., Minutolo, A., Hou, K.M., Barcelo-Ordinas, J.M., Garcia-Vidal, J., 2019. Testing the performance of sensors for ozone pollution monitoring in a citizen science approach. Sci. Total Environ. 651 https://doi.org/10.1016/j.scitotenv.2018.09.257.

Rundell, K.W., Sue-Chu, M., 2013. Air quality and exercise-induced bronchoconstriction in elite athletes. Immunol. Allergy Clin. N. Am. 33, 409–421. https://doi.org/10.1016/j.iac.2013.02.009.

Rundell, K.W., Smoliga, J.M., Bougault, V., 2018. Exercise-induced bronchoconstriction and the air we breathe. Immunol. Allergy Clin. N. Am. 38, 183–204. https://doi.org/10.1016/j.iac.2018.01.009.

Saibene, F., Mognoni, P., Lafortuna, C.L., Mostardi, R., 1978. Oronasal breathing during exercise. Pflugers Arch. 378, 65–69.

Schiffer, J., 2008. Children and youths in athletics. New Stud. Athlet. 23 (3), 7–18.

Spinelle, L., Gerboles, M., Villani, M.G., Aleixandre, M., Bonavitacola, F., 2015. Field calibration of a cluster of low-cost available sensors for air quality monitoring. Part A: ozone and nitrogen dioxide. Sensors Actuators B Chem. 215, 249–257.

Suissa, L., Fortier, M., Lachaud, S., Staccini, P., Mahagne, M.H., 2013. Ozone air pollution and ischaemic stroke occurrence: a case-crossover study in Nice, France. BMJ Open 3 (12), e004060. https://doi.org/10.1136/bmjopen-2013-004060.

Sun, L., Wong, K.C., Wei, P., Ye, S., Huang, H., Yang, F., Westerdahl, D., Louie, P.K., Luk, C.W., Ning, Z., 2016. Development and application of a next generation air sensor network for the Hong Kong Marathon 2015. Air Quality Monitoring. Sensors 16, 211.

Tainio, M., Jovanovic Andersen, Z., Nieuwenhuijsen, M.J., Hu, L., de Nazelle, A., An, R., Garcia, L.M.T., Goenka, S., Zapata-Diomedi, B., Bull, F., de Sá, T.H., 2021. Air pollution, physical activity and health: a mapping review of the evidence. Environ. Int. 147, 105954 https://doi.org/10.1016/j.envint.2020.105954.

Tiller, N.B., 2019. Pulmonary and respiratory muscle function in response to marathon and ultra-marathon running: a review. Sports Med. 49 (7), 1031–1041.

Varaden, D., Leidland, E., Lim, S., Barratt, B., 2021. "I am an air quality scientist"–using citizen science to characterise school children's exposure to air pollution. Environ. Res. 201, 111536.

Viana, M., Karatzas, K., Arvanitis, A., Reche, C., Escribano, M., Ibarrola-Ulzurrun, E., Adami, P.E., Garrandes, F., Bermon, S., 2022. Air quality sensors systems as tools to support guidance in athletics stadia for elite and recreational athletes. Int. J. Environ. Res. Public Health 19. https://doi.org/10.3390/ijerph19063561.

Vihma, T., 2010. Effects of weather on the performance of marathon runners. Int. J. Biometeorol. 54, 297–306.

WMO, 2018. Low-cost Sensors for the Measurement of Atmospheric Composition: Overview of Topic and Future Applications.

Yatkin, S., Gerboles, M., Borowiak, A., Davila, S., Spinelle, L., Bartonova, A., Signorini, M., 2022. Modified target diagram to check compliance of low-cost sensors with the data quality objectives of the European air quality directive. Atmos. Environ. 273, 118967.

Zhao, J., Lorenzo, S., An, N., Feng, W., Lai, L., Cui, S., 2013. Effects of heat and different humidity levels on aerobic and anaerobic exercise performance in athletes. J. Exerc. Sci. Fit. 11 (1), 35–41.

Zoladz, J.A., Nieckarz, Z., 2021. Marathon race performance increases the amount of particulate matter deposited in the respiratory system of runners: An incentive for "clean air marathon runs". PeerJ 9, 1–22. https://doi.org/10.7717/peerj.11562.