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Mitigation impact of roadside trees on fine particle pollution

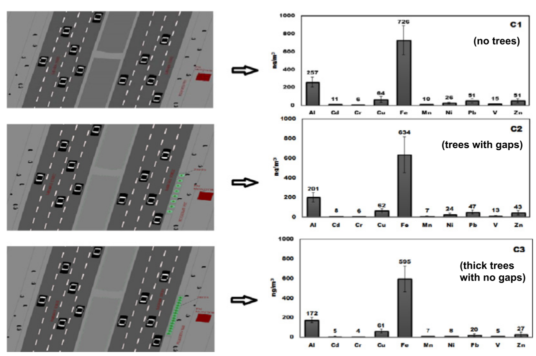
Huseyin Ozdemir

Department of Civil Engineering, Bahcesehir University, 34353 Besiktas, Istanbul, Turkey
Now at Atmospheric Modeling Section, Department of Environmental Science, Aarhus University, Frederiksborgvej 399, Roskilde, Denmark

HIGHLIGHTS

- Fine particles and respiratory health problems are increasing in Istanbul.
- Mediterranean Cypress trees were used to reduce the roadside vehicle pollution.
- Exposure decreased about >50% for cadmium and lead.
- Roadside trees reduced the fine particle concentrations to urban background level.
- Vegetation barrier design changed the roadside particle reduction amount.

GRAPHICAL ABSTRACT



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ABSTRACT

Fine particulate matter ($PM_{2.5}$) is an important air pollutant due to its adverse health effects. Vehicle emissions make a large contribution to particle concentrations in urban areas. Exposure to particles especially near roadways increases the risk of public health problems. Planting vegetation might be used to capture the fine particles at the roadside, which can lower the health risks for the urban population. Istanbul is the most populous city in Turkey, where the number of non-electric cars is increasing rapidly, resulting in decreasing air quality, especially at the roadsides. Recent studies show that cardiovascular, respiratory and total non-accidental mortality is increasing with short-term exposure to outdoor air pollution in Istanbul. In this study, roadside trees were investigated for the mitigation effect on vehicle-related $PM_{2.5}$ and heavy metal (HM). *Cupressus sempervirens* (Mediterranean Cypress trees) were planted in three different cases (i.e., no trees-C1, trees with gaps-C2 and thick trees with no gaps-C3) at the study site. Location of the site is on a dense-traffic roadside in Istanbul, where thousands of people are living, working and walking through two sides of this road. $PM_{2.5}$ samples and tree leaves were examined in the performed experiments. C2 and C3 showed the importance of roadside tree plantation by reducing the exposure to significantly low levels. Roadside $PM_{2.5}$ concentrations were reduced by 17% in C3, equivalent to urban background levels in the city. Maximum removal of HMs is observed in nickel from 26.4 ± 7.8 to 7.5 ± 2.4 μg m^{-3}. Pedestrian exposure is calculated with the measured data in three experiments and exposure is significantly reduced (e.g., >50% for cadmium and lead exposure) in experiment C3. In conclusion, three experiments showed that Mediterranean Cypress trees significantly decreased particle pollution at roadsides in Istanbul.

1. Introduction

In a world where >50% of the population is living in urban environments (UNFPA, 2015), air pollution and specifically particulate matter

E-mail address: huseyin@envs.au.dk.

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(PM) has become one of the critical issues for human health. Fine particles, which are <2.5 μm in aerodynamic diameter ($PM_{2.5}$) can reach the alveolar region and even enter the bloodstream. Exposure to $PM_{2.5}$ pollution has different health effects e.g. cardiovascular and respiratory diseases, premature mortality, reduced birth weight, lung cancer and inflammatory illnesses (Pope et al., 1995; Laden et al., 2006; Hoek et al., 2013; Sun et al., 2016). These are caused by the toxic heavy metals (HMs) in PM. HMs are defined as elements with specific weights of higher than 5 g/cm^3 (Hogan, 2010). Although they are naturally occurring elements, some of them are potentially toxic even at extremely low concentrations and likely initiators of cancer (Needleman and Landrigan, 2004; Kurt-Karakus, 2012; Liu et al., 2015).

Road traffic is one of the most important sources of fine particle emissions in urban areas. It is directly affecting the pedestrians, cyclists and even those inside homes or workplaces (Hofman et al., 2013). This has important impacts especially in the city centers, where the traffic congestion generally occurs (Pant and Harrison, 2013). Epidemiological studies have shown that living in busy streets is associated with chronic respiratory disease in children. There is also an association between long-term exposure to traffic-related air pollution and mortality rates of elderly people (Hoek et al., 2002). Mortality risk is larger for people living within 100 m of a major roadway or 50 m of a bus route (Medina-Ramón et al., 2008). Cohort studies on traffic-related air pollution show that there is a causal association between exposure to traffic-related air pollution with exacerbation of asthma, non-asthma respiratory symptoms, impaired lung function, total and cardiovascular mortality, and cardiovascular morbidity (HEI, 2010). Traffic density and long-term mortality relation were compared with combustion-related metals in $PM_{2.5}$. Results showed that traffic-density is significantly associated with mortality (Lipfert et al., 2006). Roadside trees can potentially mitigate particle pollution and protect residents from these adverse health conditions. Therefore, it is consequently important to find out the true effectiveness of trees in reducing exposure of the roadside dwellers (Al-Dabbous and Kumar, 2014).

Trees are considered one of the most appropriate solutions to decrease air pollution in urban areas (Beckett et al., 2000; Sgrigna et al., 2015). Beckett et al. (2000) have quantified the effectiveness of five tree species and found out that two conifers *P. nigra* and *C. leylandii* showed much greater effectiveness at capturing particles due to their finer and more complex structure. Pugh et al. (2012) showed that street vegetation reduces the concentration of PM_{10} even in street canyons up to 60%. Biomonitoring of traffic-derived air pollutants in Rome showed high concentrations near roads and railways (Moreno et al., 2003) due to the significant surface accumulation of roadside tree leaves (Maher et al., 2008). Plant species can act as natural filters by trapping and retaining the particles on their leaf surfaces. Maher et al. (2013) describes the impact of roadside tree lines on the indoor concentration of vehicle-derived particles. Thirty young birch trees were temporarily installed at the curbside of four houses adjacent to a heavily trafficked urban road for a period of 13 days and another four houses in the line were left treeless. PM reduced up to 50% in those houses screened by the temporary tree line. Roganovic and Đurovic (2013) have used Cypress tree bark as a bio-indicator of different HMs (Cd, Cu, Mn, Ni, and Zn) and this suggests that Cypress tree barks can be used as a bio-indicator of these pollutants. Brantley et al. (2014) studied the effect of a vegetation barrier on roadside black carbon and reported up to a 22% reduction in black carbon concentrations behind the barrier. Al-Dabbous and Kumar (2014) describes the effect of a roadside vegetation barrier on freshly emitted nanoparticles. The vegetation barrier reduced the number concentrations of nanoparticles up to 37% and exposure was found to be reduced by 36%. Studies conducted in Turkey have mainly focused on biomonitoring characteristics of plant species (Akguc et al., 2010; Osma et al., 2012; Yasar et al., 2012). *Cedrus libani* (A. Rich) was used as an indicator of airborne HM pollution in Konya (Onder and Dursun, 2006). Leaves of *Robinia pseudo-acacia* L. (Fabaceae) showed to be an effective biomonitor of environmental quality in areas subjected to industrial and traffic pollutions in Denizli (Celik et al., 2005). There is no study in Turkey regarding the mitigation effect of plants on the traffic-related $PM_{2.5}$ and HM emissions.

Vegetation barriers have been found to be beneficial in improving the roadside air quality at busy roads. However, detailed investigations are necessary in order to understand the effectiveness of vegetation barriers under varying meteorological, traffic and road conditions (Baldauf et al., 2008; Janhall, 2015). The efficiency of roadside trees in removing $PM_{2.5}$ and its HM components is less studied. Therefore, extensive field studies for optimizing their design are needed. A number of monitoring and modeling studies have investigated the effect of roadside trees on particles (Hagler et al., 2012; Maher et al., 2013). However, field studies referring to the influence of roadside trees on the traffic-derived $PM_{2.5}$ and its HMs are rare. Consequently, there is a need for field studies with well-structured experimental data (Janhall, 2015).

Vehicle traffic is getting worse in Istanbul with over 3.6 million vehicles on-road (TUIK, 2016). This is leading to prolonged exposure to high concentrations of $PM_{2.5}$ and its HM content (Summak et al., 2018). Past findings at the same study site also presented high $PM_{2.5}$ concentrations (Ozdemir et al., 2012a, 2012b, 2014). Another motivation of this study is based on the recent health studies in Istanbul. Significant associations were found between $PM_{2.5}$ and respiratory-related hospital admissions in the city (Capraz et al., 2017). Turkey also emerges as a country with one of the highest rates of premature deaths (28,924) due to air pollution in Europe from ambient PM and ozone exposure (OECD, 2014). According to the European Environment Agency (EEA), 97.2% of the urban population in Turkey is exposed to unhealthy levels of particulate matter (EEA, 2014). Cardiovascular, respiratory and other air-pollution-related diseases pose a major threat for people's health due to the rapid economic development and increase in emission sources in Istanbul. Recent studies show that exposure to higher levels of outdoor air pollution in Istanbul resulted with increased cardiovascular, respiratory and total non-accidental mortality rates in the city during 2007–2012 (Capraz et al., 2016).

In this study, the impact of trees on the reduction of $PM_{2.5}$ and its HM concentrations were examined by temporarily installing a line of young *Cupressus sempervirens* (Mediterranean Cypress) trees adjacent to an urban road in Istanbul with heavy traffic. Proximity to pollutant sources such as traffic is an important factor that affects personal exposure. Pedestrians are exposed to much more contaminated air from vehicles while walking along the roadside. This study also investigated the effects of planted roadside trees on the exposure levels to freshly emitted fine particles from the road traffic. Investigation of roadside trees on the reduction of HMs is crucial due to the potential and dangerous health risks of HMs. However, existing data is found to be insufficient in the literature except for a number of studies and HMs are investigated mainly in biomonitoring studies. Young trees are selected for the field study due to easy transportation and planting. Sampling campaigns were performed for three different cases; no trees, trees with gaps, and thick trees with no gaps. Concentrations are then measured and reported to understand the reduction level of roadside pedestrian exposure.

2. Material and methods

2.1. Study area

Istanbul is a megacity with a population of 14.7 million, which is about 18.6% of Turkey's total population. There are 3.6 million vehicles on the road and that number is increasing with 656 vehicles day^{-1} (TUIK, 2016). The city has a unique geographical location spanning two continents, Europe and Asia (Fig. 1). The climate is Mediterranean, characterized by warm-dry summer and cold-wet winter. Due to the cultural and financial features of the city, large human flux is generating urbanization and transforming Istanbul into a megacity (Ozdemir et al., 2012a, 2012b). Air quality is influenced adversely by growing urbanization and local pollution sources e.g. vehicle-traffic. The city is also

Fig. 1. Location of the sampling site (Barbaros street in Istanbul; 41.0465° N, 29.0080° E).

affected by the trans-boundary aerosol transport from the eastern part of Europe (Kindap et al., 2006).

The sampling site is a transition point in the city, where the people travel from the European and the Anatolian sites through this location. Moreover, it is close to the touristic places and business centers, which contribute to the traffic jam in every day of the week. Consequently, air pollution is high here due to dense traffic. Although there are some other pollution sources such as trans-boundary pollution, ship emissions, restaurants, etc., vehicle traffic is the dominant and closest source to the sampling point (Ozdemir et al., 2014). Re-suspension dust is also coming from the street, which is also a type of traffic-related emission. Quantifying the effect of other air pollution sources at the site will need a modeling study, which is out of the scope of this study.

Air sampling device and trees were located on the sidewalk of the urban road with heavy traffic (~44,000 vehicles day^{-1}). Young Mediterranean Cypress trees (n = 15, height = 2.2 ± 0.1 m) were used (Table 1). Mediterranean Cypress is a medium-sized coniferous evergreen tree and native species in Turkey. Coniferous evergreens are considered to be more effective in PM accumulation than broad-leaved species due to their high surface areas. Because evergreen conifers have limited seasonal changes, pollutants can accumulate throughout the year (McDonald et al., 2007; Sæbø et al., 2012). Trees were grown and stored in the plantation garden of Istanbul Metropolitan Municipality prior to the study. Although they are not exposed to an air pollution source beforehand, they are washed for the first sampling. Elemental analyses were performed with clean and polluted leaves to see the difference after the roadside exposure.

Table 1
Information about the field study and mean meteorological conditions ("±" shows the standard deviation).

	C1	C2	C3
Time span (2015)	08/28–10/04	10/05–11/09	11/10–12/07
$PM_{2.5}$ filter samples (N)	34	29	23
Traffic (vehicles day^{-1})	37,653 ± 6411	44,399 ± 4402	49,877 ± 6130
Tree diameter (m)	–	0.37 ± 0.03	0.37 ± 0.03
Tree height (m)	–	2.2 ± 0.1	2.2 ± 0.1
Tree number	–	15	15
Tree density (trees m^{-2})	–	3.7	6.7
Wind speed (m s^{-1})	5.6 ± 2.73	5.0 ± 3.37	5.0 ± 3.11
Temperature (°C)	18.5 ± 5.44	22.8 ± 3.94	15.1 ± 4.5
Relative humidity (%)	77.9 ± 8.54	74.9 ± 8.44	83.1 ± 12.34
Rainfall (mm)	1.7 ± 4.45	0.1 ± 0.67	1.4 ± 2.51

Trees have been planted at a distance of 3.6 m from the street and three different cases (C1, C2, and C3) were tested for a consecutive series of sampling campaigns in spring 2015. Because trees and sampling device are close to the road (Fig. 2), vehicle traffic becomes the main source of fine particles at the sampling site. Each case was conducted over approximately one month to minimize the effect of weather variations. As presented in Fig. 2a, C1 stands for the no-tree case and serves as a comparison for the other two cases. Secondly, C2 represents the trees with gaps, and they were installed with a space of 0.8 m (Fig. 2b). Finally, case 3 (C3) represents a temporary line of thick trees with no gaps (Fig. 2c). Three cases provide an opportunity to see the most efficient way of planting trees at the roadside for the accumulation of $PM_{2.5}$ and HMs. One sampling instrument was available during the study and daily measurements were made for the filter sampling. Further information for each campaign and the sampling site can be found in Table 1.

One of the air quality monitoring stations of the Ministry of Environment and Urbanization (close to the sampling point) provided concurrent PM_{10} concentrations ($PM_{2.5}$ were not available). In addition, to compare particle results with the background concentrations, Sile air quality monitoring station, which is in the northern part of the city (provided by the Ministry of Environment and Urbanization), is used. Traffic density data is provided by the traffic department of Istanbul Metropolitan Municipality (Table 1). Mean meteorological data (temperature, wind speed, rain, and humidity) were obtained from the nearest weather station (Florya; 40.9758 N, 28.7865 E). During the sampling campaigns, weather was temperate (15–23 °C), humidity was moderate (75–83%), wind speed was moderate (<6 m s^{-1}), and there was nearly no rain. Average meteorological conditions of C1, C2, and C3 given in Table 1 have similar patterns, making it easier to assess the results of the study.

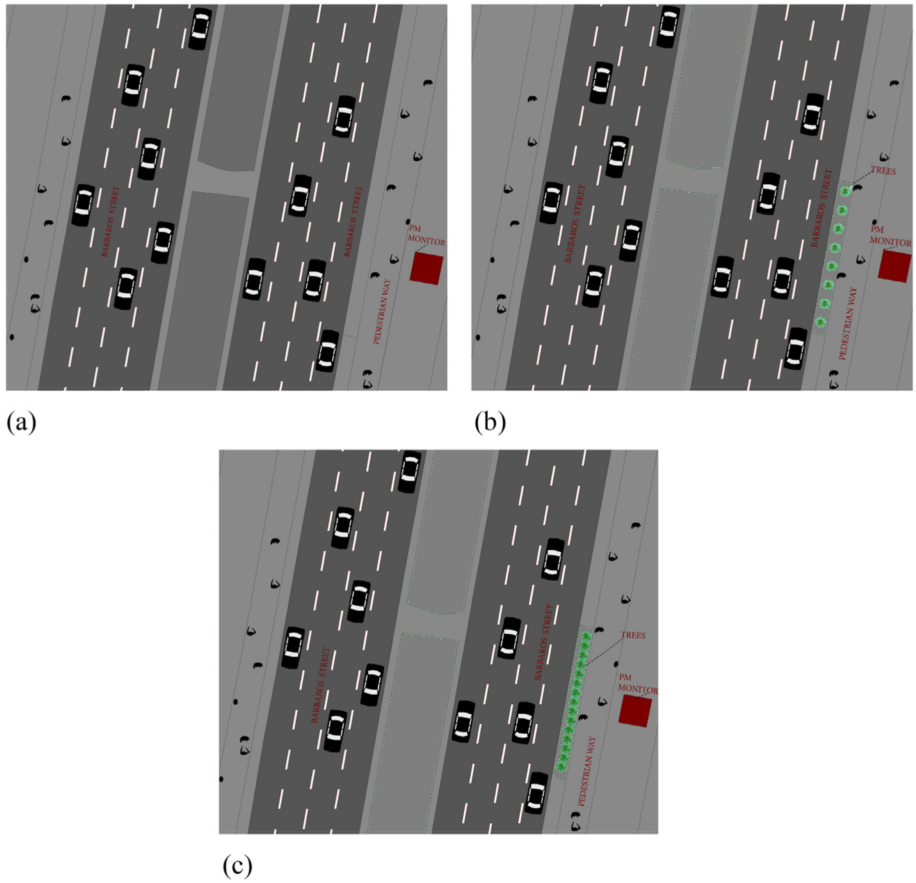

Fig. 2. Schematic representation of the sampling site with three different cases (not to scale); (a) no trees case-C1, (b) trees with gaps case-C2, (c) thick trees with no gaps case-C3.

2.2. Experimental methods

The effect of installed roadside trees was examined by field sampling and instrumental analysis. Firstly, filter and leaf samples were collected, then the instrumental analysis was performed for the $PM_{2.5}$ and elemental concentrations. $PM_{2.5}$ samples were collected by a medium volume sampler "Zambelli ISO PLUS 6000" located at a distance of 9.8 m from the road and 2 m above the ground. The sampler worked at a flow rate of 1 $m^3\ h^{-1}$ for one day with teflon filters (Pall-PTFE, 2 μm pore size and 47 mm diameter). Filters were conditioned in a desiccator at room temperature in open petri dishes for 24 h. Filter samples were transported to the sampling site from the laboratory in petri dishes sealed with aluminum foil every morning at 9 a.m. manually (number of samples for each case is given in Table 1). $PM_{2.5}$ mass concentrations were determined gravimetrically by subtracting the initial mass of the filter sample from the final mass and then divided by the volume of the air passed through it. Filters were weighted using an analytical balance (Ohaus Adventurer Pro-AV264).

Five leaves were collected from each tree and then they were divided into two groups. Half of them were washed with deionized distilled water to remove dust particles in a standardized procedure and remaining samples were analyzed unwashed. Removal rates by the leaves were calculated by comparing unwashed and washed leaves. Leaves were sampled on both sides of the trees (i.e., those growing on roadside and pedestrian side) and from different heights. Leaves were isolated, oven-dried at 80 °C for 48 h, then 0.2 g was taken and transferred into Teflon vessels with adding 8 ml of 65% HNO_3 (Merck). Air filter samples were isolated in sterile Petri dishes and transferred into Teflon vessels. Then, 5 ml 98% H_2SO_4 (Merck) and 3 ml 72% $HClO_4$ (Merck) were added. Samples were mineralized in a microwave oven (Berghof-MWS2); 5 min at 145 °C, 5 min at 165 °C, and 20 min at 175 °C. After drawing calibration curves, element measurements (Al, Cd, Cr, Cu, Fe, Mn, Ni, Pb, V, and Zn) were conducted by ICP-OES (Perkin Elmer - Optima 7000DV).

Maintenance and calibration of the analytical devices used in the study were done before the measurements. Blank samples were used during the field and laboratory measurements to produce reliable and accurate results. Blank filters used for $PM_{2.5}$ mass concentration measurements exposed to the same sampling and weighing steps, but air passed through the blank filters for only 1 min (two blank filters were used for each case). Then, filters were removed from the sampler and treated like a real sample. Blank filter values (mean ± standard deviation $= 0.9 \pm 0.1\ \mu g\ m^{-3}$) used for correcting the $PM_{2.5}$ readings from the analytical balance. Quality control procedures were also performed for the elemental analysis of the air samples. Blank samples used for HM measurements were found to have non-detectable levels of the HMs.

2.3. Estimation of $PM_{2.5}$ and HM exposure

Epidemiological evidence suggests a causal relationship between particle exposure and cardiovascular morbidity and mortality (Brook et al., 2010). Ambient air is the potential source of exposure to particulate matter, and the primary route of this exposure is inhalation. Proximity to contaminant sources is an important factor that affects personal exposures. Pedestrians are exposed to particulate emissions from vehicles by direct inhalation while walking along the roadside. Increased physical effort leads to higher breathing rates, thus elevated levels of inhaled doses and subsequent lung deposition of air pollutants per unit time (Int Panis et al., 2010). The monitoring site is on a very common walking route of the city (Barbaros Street). Here, pedestrians receive a high dose of air pollutants due to the heavy traffic including diesel buses. In this study, exposures to nine common HMs are investigated. Inhaled contaminant dose depends on the ventilation rate (VR) and the pollutant concentration. VR, also known as breathing or inhalation rates is the volume of air that is inhaled by an individual in a specified time period (U.S. EPA, 2008). The product of VR and the measured concentrations of the pollutants give the inhaled amount (Al-Dabbous and Kumar, 2014). In this study, VR values recommended by U.S. EPA (2011) were used for light-intensity activities (defined as walking at speed 2.4–4.8 km h^{-1}). Average of VR for ages between 1 and 71 years old was calculated and used for the assessment of inhaled contaminant doses (Table 4).

3. Results and discussion

3.1. Variations of $PM_{2.5}$ concentrations

The descriptive statistics of daily $PM_{2.5}$ concentrations are presented in Table 2 for each case study. The air quality guideline for daily mean $PM_{2.5}$ concentration target is 25 μg m^{-3} (WHO, 2005). Results show that the average $PM_{2.5}$ level is higher than the World Health Organization (WHO) 2005-guideline limits. To reduce the particle pollution, C2 and C3 tests were applied with temporary installing tree lines. C3 decreased the mean concentration of $PM_{2.5}$ compared with C1 and C2. Mean $PM_{2.5}$ concentration increased in C2 compared with the no-tree-case, indicating the negative effect of gaps between the trees.

$PM_{2.5}$ concentration at C2 is slightly increased (2.4%) compared to C1. The main reason is the increase of the vehicle number. During C1 average vehicle count was 37,653 vehicles day^{-1}, but increased to 44,399 vehicles day^{-1} at C2 (Table 1). Increase in vehicle number (17.9%) and also the tree configuration with gaps at C2 increased the $PM_{2.5}$ concentration. The increase at C2 is followed by 20.83% decrease at C3. Notice that although the vehicle number increased to the maximum value of 49,877 vehicles day^{-1}, $PM_{2.5}$ concentration has decreased to 28.81 μg m^{-3} at C3, showing that removing the gaps between the trees decreased the $PM_{2.5}$ concentrations. Moreover, heavy metal concentrations found on the filter samples showed a decreasing trend at both C2 and C3.

Additional statistical analysis was performed to understand the relations between the variation of the $PM_{2.5}$ concentrations with the meteorological and traffic data. Two non-parametric (Kendall and Spearman) and one parametric (Pearson) correlation tests were used to derive the correlations. Although correlation coefficient (R) values were lower (<0.8), p-values showed that there is significant correlation between the $PM_{2.5}$ with precipitation (Pearson's test, $p < 0.05$; Kendall's and Spearman's tests, $p < 0.01$), and with relative humidity (Kendall's and Spearman's tests, $p < 0.05$). No significant correlations are observed with the temperature and wind speed ($0.2 < R$). Spearman's test indicated a significant correlation ($p < 0.05$) between $PM_{2.5}$ and traffic counts, but correlation coefficient values are low ($0.3 < R$). The reason for the low R-values is because of the resolution of the PM data which are daily averages, so are not sufficient to catch the trend of the hourly traffic values.

The previous study investigated the $PM_{2.5}$ and black carbon concentrations at different sampling sites in Istanbul (Ozdemir et al., 2014). Results showed that $PM_{2.5}$ concentrations were between 30 and 41 μg m^{-3} at traffic-affected sites, and lower concentration of 27 μg m^{-3} was measured at the urban background site. Highest $PM_{2.5}$ deposition by the trees was found in C3 (Table 2), which is nearly the urban background level of Istanbul. Thus, comparison of the results suggests that the design of the trees has a significant effect on the reduction of fine particles. Besides, standard deviations for the daily $PM_{2.5}$ concentrations in Table 2 are large because of the varying traffic volume.

Table 2
Statistical summaries of daily $PM_{2.5}$ concentrations (μg m^{-3}).

	Mean	Standard deviation	Median	Minimum	Maximum
C1	35.54	27.81	28.28	12.12	131.67
C2	36.39	16.91	33.41	12.11	80.80
C3	28.81	10.75	28.29	8.12	60.63

Fig. 3 shows the results of the $PM_{2.5}$ concentrations for the three cases with box and whisker plot. As can be seen in the figure, C2 was not successful for the reduction of fine particles. Mean and median concentrations increased compared with C1, while only the maximum value is decreased. For C3, we can see the impact of trees with the mean value decreased below the levels in C1 and C2.

Fine particles are sensitive to even minor air turbulence, and are difficult to deposit (Chen et al., 2016). The gaps between trees in C2 allow particles to pass through the vegetation and making it more difficult to remove of $PM_{2.5}$. Therefore, C2 was not effective in removing fine particles, but $PM_{2.5}$ was captured more in C3 with the less ventilation between the trees. Greater density or wider buffer configurations are encouraged to increase the mitigation potential (Hagler et al., 2012). This is comparable with the results reported in the previous studies. Mitchell and Maher (2009) observed a 15% decrease, Bealey et al. (2007) reported up to 20% reduction in particle concentrations with the presence of trees.

$PM_{2.5}$ values reached very high levels (132 μg m^{-3}) in C1, highlighting the importance of mitigation measures in the city. The reason for the very high fine particle concentration is the dense and also stop-and-go traffic. It is known that idling vehicles emit higher pollution concentrations than moving vehicles (Zhu et al., 2002). Epidemiological evidence also supports the importance of stop and go traffic to health effects. For instance, Ryan et al. (2005) found that infants living very near (<100 m) to stop-and-go traffic had a significantly increased prevalence of wheezing when compared with those that were not exposed.

The air quality monitoring station of the Turkish Ministry of Environment and Urbanization close to the study site provided concurrent PM_{10} concentrations ($PM_{2.5}$ was not available). The monitoring station is around 1 km far from the sampling site and has no vegetation barrier. Average PM_{10} concentrations for the time periods of C1, C2, and C3 were: 37 μg m^{-3}, 42 μg m^{-3}, and 65 μg m^{-3}, respectively. PM_{10} increase is parallel to the increase of vehicle number (Table 1) over this three-month study. To compare particle results with the background concentrations, Sile air quality monitoring station is used which is in the northern part of the city (provided by the Ministry of Environment and Urbanization). Average background PM_{10} concentrations for C1, C2, and C3 time periods were: 23 μg m^{-3}, 20 μg m^{-3}, and 26 μg m^{-3}, respectively, which is showing a significant difference between the urban concentrations.

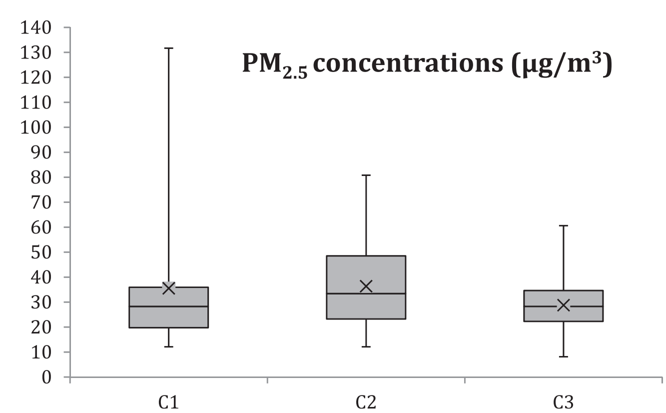

Fig. 3. Box and whisker plot of $PM_{2.5}$ concentrations for the three cases; mean (cross sign), median (solid line in box), 25th and 75th percentiles (upper and lower ends of boxes), 10th and 90th percentiles (upper and lower whiskers).

3.2. Variations of HM concentrations

Metals are often present in the environment naturally at low levels, but exposure to larger amounts can be dangerous. Pedestrians are exposed to HM contaminated air during walking along the roadside. In order to understand the effect of trees to the HM exposure, we have analyzed filter and leaf samples for the HMs: aluminum (Al), cadmium (Cd), chromium (Cr), copper (Cu), iron (Fe), manganese (Mn), nickel (Ni), lead (Pb), and zinc (Zn). The results of the HM measurements from washed and unwashed leaves of Mediterranean Cypress for C1, C2, and C3 are given in Table 3. All calculations were based on the dry weight of washed and unwashed leaves. Data analysis for the differences in HM results among three different cases was performed by using multivariate analysis of variance (MANOVA) with Tukey's Post-Hoc HSD test (level of statistical significance is expressed as $^{**}p < 0.01$). The C1 represents the values of washed leaves of the day till the start of C2. Case 2 also shows the values of washed and unwashed leaves of the first day of C3. Finally, C3 shows the values of the samples of washed and unwashed leaves of the last day of the experiment. Table 3 clearly shows an increase in the amounts of heavy metals of the samples of washed and unwashed leaves throughout the observed period. For the leaf samples, Al and Fe were found to show the highest roadside concentrations and Cd, Cr, and Ni in the lowest levels. Results show that particles are rich with Al and Fe where iron is released from high-temperature combustion processes (Petrovsky and Ellwood, 1999), which indicate vehicle-related air pollution.

HM removal rates of leaves for C2 and C3 are shown in Fig. 4. Removal rates for Ni, Pb, Cu, Zn, and Mn show the highest removal in C3 with 42.56%, 26.60%, 19.61%, 19.12%, and 19.07%, respectively. These metals are used as markers for vehicle emissions (Pant and Harrison, 2013), which indicate vehicle fuel combustion as the major source for these HMs. However, particles arise not only from the exhaust emission, also from vehicle wear such as tire and brake components as well as from resuspension of roadside dust (Pant and Harrison, 2013). There are key tracers used for these non-exhaust emissions. For instance, tire wear has been reported to be a significant source of Zn. Fe and Cu are tracers for brake wear (Adachi and Tainosho, 2004). It can be seen from Fig. 4 that most of the HMs removed during the C3 in higher removal rates compared to C2. Maximum removal difference is observed for Ni (12.7% in C2 and 29.9% in C3), and a recent study reported that low birth weight risk is associated with the exposure of nickel (Sun et al., 2016).

Table 3
Concentrations of HMs (mg/kg dw) in washed leaf (WL) and unwashed leaf (UWL) samples collected on the first day of C2, last day of C2 and last day of C3 (mean difference is significant at 0.01 (**) level by the multivariate analysis of variance-MANOVA); the "±" sign shows the standard deviation values.

		First day of C2	Last day of C2	Last day of C3
Al	WL	268.86 ± 1.32	273.31 ± 1.35**	277.61 ± 1.38**
	UWL	–	284.25 ± 1.41**	301.90 ± 1.49**
Cd	WL	0.36 ± 0.002	0.37 ± 0.002**	0.38 ± 0.002**
	UWL	–	0.38 ± 0.002**	0.39 ± 0.002**
Cr	WL	0.56 ± 0.003	0.60 ± 0.003**	0.66 ± 0.003**
	UWL	–	0.63 ± 0.003**	0.69 ± 0.003**
Cu	WL	7.21 ± 0.04	7.47 ± 0.04**	8.33 ± 0.04**
	UWL	–	8.14 ± 0.04**	9.40 ± 0.05**
Fe	WL	422.07 ± 2.08	427.28 ± 2.11**	441.67 ± 2.19**
	UWL	–	447.57 ± 2.23**	470.33 ± 2.33**
Mn	WL	37.18 ± 0.18	39.52 ± 0.20**	45.88 ± 0.23**
	UWL	–	43.43 ± 0.22**	51.01 ± 0.25**
Ni	WL	0.88 ± 0.004	0.95 ± 0.01**	1.04 ± 0.01**
	UWL	–	1.09 ± 0.01**	1.48 ± 0.01**
Pb	WL	2.62 ± 0.01	3.11 ± 0.02**	4.05 ± 0.02**
	UWL	–	3.50 ± 0.02**	4.79 ± 0.02**
Zn	WL	15.89 ± 0.08	16.70 ± 0.08**	18.23 ± 0.09**
	UWL	–	18.22 ± 0.09**	20.43 ± 0.10**

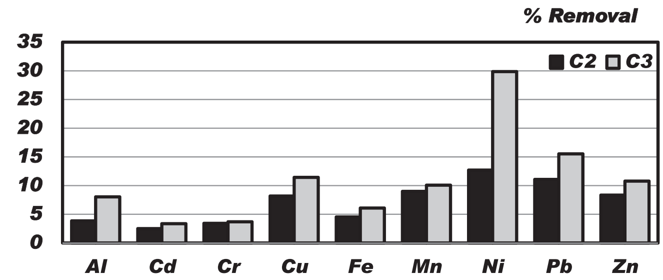

Fig. 4. HM removal percentage of the tree leaves for C2 and C3.

Road traffic was found to be a key contributor to fine, ultrafine, and nanoparticles such as Zn, and Pb (Lin et al., 2005). Particularly, Cd and Zn have been reported to be strongly associated with diesel fuel, whereas Cu and Mn have been found to be associated with gasoline and liquefied petroleum gas (LPG) engines (Lin et al., 2005; Cheng et al., 2010). Zn is also a possible source of vehicle brake and tire wear and Mn is used as an anti-knock agent in the fuel (Huhn et al., 1995). Al, Fe, and Mn have a significant crustal origin; Cd and Cu from brake wear; Pb and V from petrol (Hassanien, 2011).

A similar dependence on tree leaves is observed for HMs in $PM_{2.5}$ samples. Fig. 5 shows the reduction of HM concentrations from C1 to C3 (69%-Ni, 67%-V, 61%-Pb, 55%-Cd, 47%-Zn, 33%-Al and Cr, 30%-Mn, 18%-Fe, and 5%-Cu reduction). C2 also displayed a considerable amount of reduction compared to C1 for Cd, Al, and Fe with 27%, 22%, and 13%, respectively. HM concentrations of $PM_{2.5}$ for the no-tree case (C1) is comparable with the results of a previous study conducted at the same location (Demir et al., 2010).

For the HMs investigated in filter samples, the highest roadside concentrations were found for Fe (726 ng m^{-3} in C3). High iron content of $PM_{2.5}$ might represent a particular potential hazard to health (Maher et al., 2013), such as causing oxidative stress in lung and brain cells (Borm et al., 2007; Allsop et al., 2008). Oxidative brain damage is a characteristic of Alzheimer's and Parkinson's disease (Block et al., 2007; Allsop et al., 2008). In C3, iron concentration is reduced to 595 ng m^{-3}. WHO air quality guideline limit for cadmium is 5 ng m^{-3} (WHO, 2000), and Cd concentration is reduced below this level in C3.

Lead is a highly toxic metal and as a result of related health concerns, its use in gasoline has been stopped since 2004 in Turkey. However, the lead-containing vehicle parts (fuel tanks and hoses, piston coatings, etc.) can contribute to the non-fuel sources of Pb (Maher et al., 2008). The concentrations of the lead reduced by 61% in C3 by 20 ng m^{-3}. This is below 0.5 μg m^3, which is the E.U. Commission (1999) ambient air quality guideline limit for lead. In Europe, the background concentration of lead in air was mainly within the 10–30 ng m^{-3} range (Aas and Breivik, 2007). The lead concentration at the roadside has reduced to nearly the background values.

Nickel compounds are human carcinogens, and emissions of nickel occur from the combustion of fossil fuel (WHO, 2000). Results show that nickel concentrations have been reduced by 69% to 8 ng m^{-3} in C3. This is below the E.U. (2012) ambient air quality guideline for nickel of 20 ng m^{-3}. Manganese is reduced from 10 to 7 ng m^{-3}, and below 0.15 μg m^{-3} environmental exposure to manganese is not likely to have adverse effects (WHO, 2000). Vanadium is a potent respiratory irritant, and it is believed that below 1 μg m^{-3} environmental exposure to vanadium is not likely to have adverse effects on health (WHO, 2000). Fig. 5 shows that vanadium is also reduced in C2 and C3, below to a safe level. For chromium and its compounds, there is sufficient evidence of carcinogenicity. Air quality guideline for Cr is 2.5 ng m^{-3} (WHO, 2000), however, chromium is decreased to 4 ng m^{-3} still higher than the limit. It can be concluded that roadside trees have different removal efficiency on HMs, which can result due to a range of factors (e.g., wind speed, particle diameter, magnetic properties, etc.). Slight differences might result from the limited number of samples and short overall sampling time.

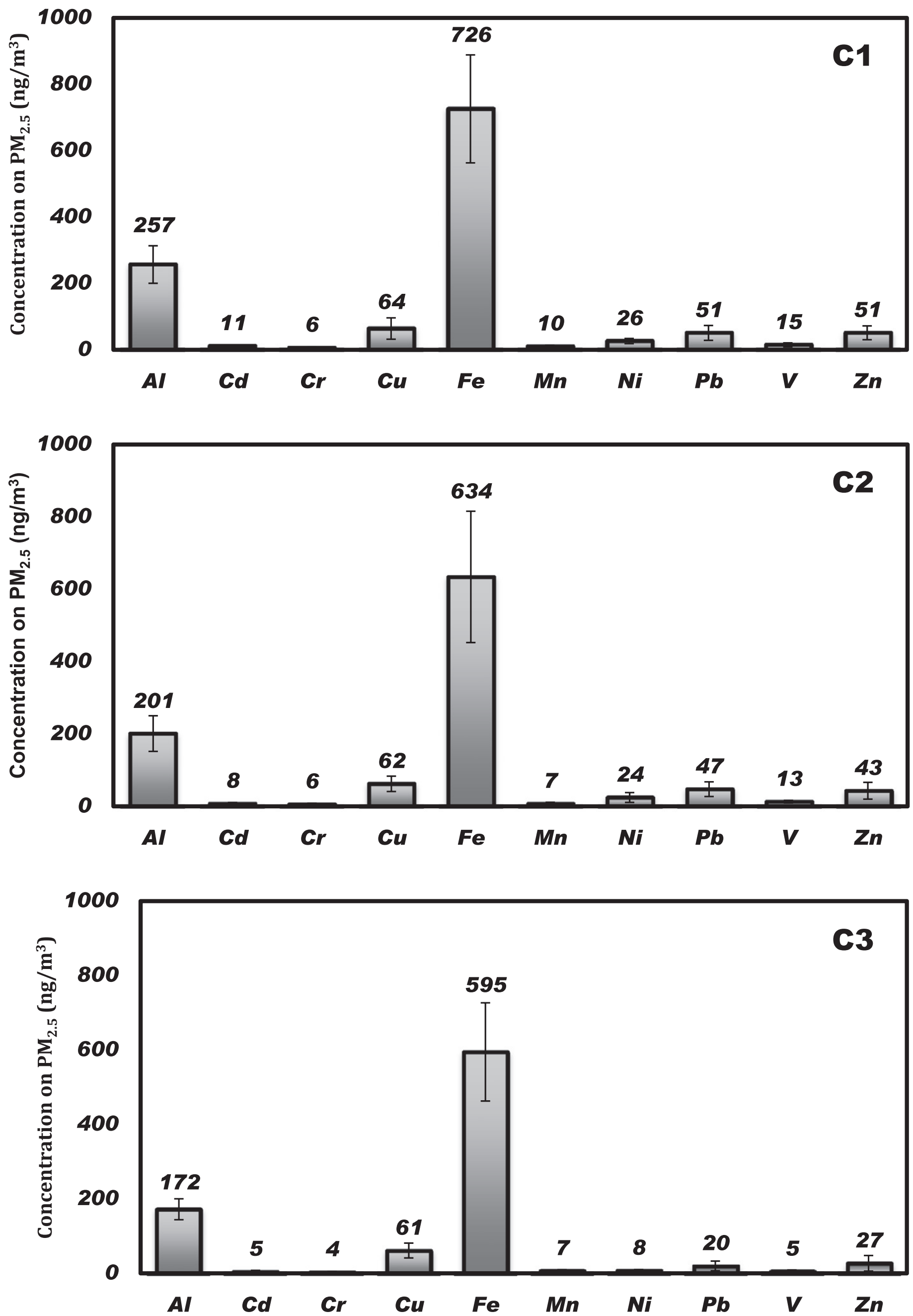

Fig. 5. Concentrations of HMs (ng m^{-3}) in $PM_{2.5}$ (values above the bars show the mean values).

An additional elemental analysis is carried out for the background HMs with the same methodology. Buyukada Island is selected as a background location, which is in the Marmara Sea close to the southeast coast of Istanbul. The island does not have vehicle traffic. According to the results of this background study, difference between HM concentrations of urban and background tree leaves are; 21, 259, 169, 197, 74, 120, 62, 275, and 236% for Al, Cd, Cr, Cu, Fe, Mn, Ni, Pb, and Zn, respectively. The difference between urban and background concentrations shows the obvious traffic effect.

3.3. Exposure analysis

In air pollution, the primary route of exposure is inhalation. Direct inhalation of particulate emissions from vehicles is a potential pathway of exposure for the pedestrians. Trees can reduce the pedestrian's exposure to traffic-induced particles through the interception on its leaf surface (Al-Dabbous and Kumar, 2014; Brantley et al., 2014). Measured data is assessed for the exposure analysis in order to see the difference between the three cases. Table 4 presents the calculated $PM_{2.5}$ and HM exposures for each case. The age groups for the exposure calculations are; 1–11, 11–21, 21–31, 31–41, 41–51, 51–61, and 61–71. The average ventilation rates (VR) for these age groups are; 1.16×10^{-2} m^3 min^{-1}, 1.33×10^{-2} m^3 min^{-1}, 1.3×10^{-2} m^3 min^{-1}, 1.36×10^{-2} m^3 min^{-1}, 1.44×10^{-2} m^3 min^{-1}, 1.46×10^{-2} m^3 min^{-1}, and 1.41×10^{-2} m^3 min^{-1}, respectively (U.S. EPA, 2011). These values are recommended for light-intensity physical activities including walking. Exposures shown in Table 4 were calculated by multiplying VR values for each age group with the measured average pollutant concentrations.

Influence of presence (C3) and absence (C1) of vegetation on the exposure can be seen in Table 4. Exposure to all of the contaminants reduced with planting the adjacent trees in C3. Time spent walking in such a heavily trafficked environment will affect the exposure amount

Table 4
$PM_{2.5}$ (μg min^{-1}) and HM (ng min^{-1}) exposure according to the age groups (lowest values are marked in bold and results presented here are for male subjects with light-intensity activities).

	Age group	$PM_{2.5}$	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb	V	Zn
C1	1–11	0,41	2,97	0,127	0,069	0,74	8,40	0,116	0,30	0,59	0,174	0,59
	11–21	0,47	3,42	0,146	0,080	0,85	9,66	0,133	0,35	0,68	0,200	0,68
	21–31	0,46	3,34	0,143	0,078	0,83	9,44	0,130	0,34	0,66	0,195	0,66
	31–41	0,48	3,49	0,150	0,082	0,87	9,87	0,136	0,35	0,69	0,204	0,69
	41–51	0,51	3,70	0,158	0,086	0,92	10,45	0,144	0,37	0,73	0,216	0,73
	51–61	0,52	3,75	0,161	0,088	0,93	10,60	0,146	0,38	0,74	0,219	0,74
	61–71	0,50	3,62	0,155	0,085	0,90	10,24	0,141	0,37	0,72	0,212	0,72
C2	1–11	0,42	2,33	0,093	0,069	0,72	7,31	**0,081**	0,28	0,54	0,150	0,50
	11–21	0,48	2,67	0,106	0,080	0,82	8,41	0,093	0,32	0,62	0,173	0,57
	21–31	0,47	2,61	0,104	0,078	0,81	8,22	0,091	0,31	0,61	0,169	0,56
	31–41	0,49	2,73	0,109	0,082	0,84	8,59	0,095	0,33	0,64	0,177	0,58
	41–51	0,52	2,89	0,115	0,086	0,89	9,10	0,101	0,35	0,68	0,187	0,62
	51–61	0,53	2,93	0,117	0,088	0,90	9,23	0,102	0,35	0,69	0,190	0,63
	61–71	0,51	2,83	0,113	0,085	0,87	8,91	0,099	0,34	0,66	0,183	0,61
C3	1–11	**0,33**	**1,99**	**0,058**	**0,046**	**0,71**	**6,89**	**0,081**	**0,09**	**0,23**	**0,058**	**0,31**
	11–21	0,38	2,29	0,067	0,053	0,81	7,91	0,093	0,11	0,27	0,067	0,36
	21–31	0,37	2,24	0,065	0,052	0,79	7,73	0,091	0,10	0,26	0,065	0,35
	31–41	0,39	2,34	0,068	0,054	0,83	8,09	0,095	0,11	0,27	0,068	0,37
	41–51	0,41	2,48	0,072	0,058	0,88	8,57	0,101	0,11	0,29	0,072	0,39
	51–61	0,42	2,51	0,073	0,058	0,89	8,69	0,102	0,12	0,29	0,073	0,39
	61–71	0,41	2,42	0,071	0,056	0,86	8,39	0,099	0,11	0,28	0,071	0,38

and large peaks of exposure may be experienced in rush hours of the traffic. Especially, for sensitive individuals (e.g. asthmatics), these may imply significantly increased acute health risks (Svartengren et al., 2000). Cadmium and lead are very toxic metals and are known human carcinogens (Martin and Griswold, 2009). As can be seen in Table 4, Cd and Pb exposures are decreased significantly in C3 (>50%). Allergic skin reactions are the most common health effects of nickel. Manganese is characterized by neurotoxic effects and vanadium is a possible respiratory irritant. Human exposure to chromium compounds causes the occurrence of cancer, particularly lung cancer (WHO, 2000). According to the results presented in Table 4, exposures to these heavy metals are also reduced in different amounts in C3.

4. Conclusions

Roadside air quality is an increasing public health concern, especially in the city centers, where the traffic congestion generally occurs. Vehicle traffic in Europe contributed over 60% of the total particle number emissions in 2010 and this can be up to 90% along the roadsides in polluted urban environments (Kumar et al., 2010). This study investigates the mitigation effect of roadside-tree-planting on the traffic-derived $PM_{2.5}$ and its HM constituents. The study is conducted in three different cases at the roadside of a dense traffic street in Istanbul. Tree leaves and $PM_{2.5}$ samples were collected and analyzed in each case.

Air filter and leaf samples have both demonstrated decreasing $PM_{2.5}$ and HM concentrations with the planting of the roadside trees. The reduced pollutant concentrations reflect the filtering effect of the adjacent trees, especially in C3. High $PM_{2.5}$ concentrations reaching 132 μg m^{-3} in C1 and increasing number of hospital admissions due to the respiratory problems are indicating the necessity of a mitigation measure. Gaps between tree stands in C2 allow the transport of traffic-related air pollution to the pedestrian area, pointing out that denser plantation increases the efficiency for the particle reduction capacity. The single line of young trees as planted in C3, $PM_{2.5}$ concentrations can be reduced with 17%, which is the urban background level in Istanbul. Normally, particle concentrations decrease by 10–20% in the first 200 m from busy roads, with no further decrease at greater distances (Brauer et al., 2012). In this study, the sampling station is 9.8 m from the road at the study site, therefore reduction is due to the planted trees and dilution will play a minor role for the decrease of the concentrations.

To reduce the exposure risk to vulnerable populations, locating schools, childcare facilities, hospitals, and residences a distance of 150 m from busy roads is advised (Brauer et al., 2012). However, implementing this for the already established cities is impossible. Results of this research showed that roadside Mediterranean Cypress tree plantation like in C3 design has a significant impact on the reduction of traffic-related fine particle pollution. This might be a solution to mitigate the increasing air pollution exposure problem in Istanbul for the pedestrians and most importantly for vulnerable populations. The design and type of tree are also important when using vegetation for the air quality improvements. In order to reduce the increasing health problems associated with air pollution in the city, the easy, cheap and sustainable method proposed in this study can be used by the municipalities.

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