Highly Time-Resolved Elemental Source Apportionment at a Prague Urban Traffic Site

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Abstract

Positive Matrix Factorization was used for source apportionment of 2-hour online metals (Xact625i; PM2.5) and elemental and organic carbon (EC/OC analyser; PM1) data at an urban traffic site in Prague, Czech Republic, for two month-long periods in spring (February – March) and winter (November – December), 2020 (400 samples). Five factors were obtained for spring: Local heating (39%), Soil/road dust (21%), Secondary inorganic aerosol (20%), Traffic (12%), and Road salt (9%), while four factors were resolved for winter: Local heating (20%), Soil/road dust (31%), Secondary inorganic aerosol (SIA: 36%), and Traffic (13%). Aside from SIA, the sources were of local origins. Air mass back trajectories calculated by HYSPLIT and concentration weighted trajectories (CWT) were used to identify long-distance sources. Greater PM concentrations during winter (~3 times greater) were attributed to both lower boundary layer heights and more abundant air masses from Central Europe, compared to more marine air masses during spring.

Keywords: PMF, online, Xact, EC/OC
1 INTRODUCTION

Atmospheric particulate matter (PM) has been widely studied due to the significant concerns over its effects on both human health and the environment. The World Health Organisation (WHO) estimated that up to 7 million premature deaths per year are caused by air pollution, of which PM$_{2.5}$ (particles with a 50% aerodynamic cut-off diameter of 2.5 μm) makes a significant contribution (WHO, 2022).

At urban sites, the chemical and physical properties of PM can be complex due to the wide variety of anthropogenic and non-anthropogenic sources. Traffic and domestic heating are often the main contributors to urban PM, but the long-range transport of industrial emissions, as well as those of natural origin (i.e., dust or wildfires), also have significant contributions (Lanz et al., 2010; Diapouli et al., 2017; Belis et al., 2019; Tessum et al., 2022). Although PM concentrations in European cities are relatively lower than those in countries such as China and India, air quality in numerous cities is still of concern. The European Environment Agency stated that in 2021, 97% of the EU’s urban population was exposed to unsafe levels of PM (EEA, 2023).

For simplicity, traffic emissions are often classified as either tailpipe or non-tailpipe in origin; tailpipe emissions are direct engine exhaust emissions, while non-tailpipe emissions include the abrasion of tyres, clutches, and brakes, as well as resuspended ‘road’ dust (Pant et al., 2013). They can be further categorised by their size distributions: road dust is mostly found in the coarse (PM$_{1.2.5-10}$) fraction (Harrison et al., 2021; Matthaios et al., 2022; Lin et al., 2022), tyres in both the fine (PM$_{0.1-1.2.5}$) and coarse fractions (US EPA, 2020; Lin et al., 2022), and brake wear particles being found in the fine, coarse, and ultrafine (PM$_{0.1}$) fractions (Iijima et al., 2007; Kukutschová et al. 2011; Kwak et al.,
2013), whereas tailpipe emissions are mainly found in the fine and ultrafine fractions (Kam et al., 2012; Vu et al., 2015).

With regards to public health, epidemiological studies have directly linked particulate vehicular emissions to numerous medical conditions such as cardiovascular morbidity, oxidative stress, and the exacerbation of pre-existing medical conditions (McCreanor et al., 2007; Grahame et al., 2010; Schraufnagel et al., 2019; Rich et al., 2019; Arias-Pérez et al., 2020), particularly from smaller particles (Li et al., 2010; Cheng et al., 2016), stemming from high trace metal and carbonaceous content in traffic-related PM.

The focus of recent regulations has been on reducing PM tailpipe emissions, such as the Euro 1- Euro 6 emission standards focusing on gaseous and particle number emissions, which have proven to be effective in reducing total vehicle tailpipe emissions (Barrett et al., 2015; Yu et al., 2019; Suarez-Bertoa et al., 2020; Giechaskiel et al., 2021; Sánchez-Ccoyllo et al., 2022). However, regulations targeting non-tailpipe emissions from passenger vehicles are still absent (OECD, 2020). These emissions remain a point of concern, as shown by the Euro 7 emission standard proposal that also requires reduction of non-tailpipe PM emissions. While the move towards an electric fleet will reduce tailpipe emissions, non-tailpipe emissions from these vehicles will continue to be significant (Timmers et al., 2016).

This work investigates the contribution of traffic to local PM$_{2.5}$ concentrations and identifies individual PM sources in Prague, the capital of the Czech Republic, through the elemental characterisation of aerosols at an urban traffic site. The project involves the use of source apportionment using two high-time resolution online instruments, the Xact625i ambient metals monitor and the Sunset Laboratory EC/OC analyser, the combination of which has only recently been
used for such modelling. While studies have involved the combination of online metal and carbonaceous analyses for source apportionment, the use of fraction-resolved EC (elemental carbon) and OC (organic carbon), to the best of our knowledge, has not yet been done (Ho et al., 2018; Wang et al., 2018; Liu et al., 2019; Jin et al., 2020; Wang et al., 2022; Zhang et al., 2022).

2 METHODS

2.1 Sampling Period and Location

Sampling took place at the Praha-Vršovice site (located in the garden of a kindergarten, 50°03′59″N, 14°26′46″E, 201 m ASL), an urban traffic site operated by the Czech Hydrometeorological Institute (CHMI). The site is located in the central European city of Prague (population ~1.2 million), at the bottom of an open and ventilated valley (Fig. 1). Local traffic emissions from the adjacent street consist of ~12,300 cars, ~400 heavy vehicles, and ~700 trams per day (TSK, 2022), with a tram stop positioned 100 m southeast. Further, there is a railroad station 200 m southwest (through which most trains entering Prague pass) and a highway 800 m away, which has a traffic volume of ~77,500 vehicles per day. While instances of excessive braking by cars are uncommon, braking by trams and trains using electrodynamic braking is common, with some instances of friction braking at low speeds. All trams and most trains are electric. Residential housing in the vicinity of the station is in the form of multi-story buildings.

Figure 1. Sampling site in Prague, Czech Republic.
Two campaigns were performed in 2020, February 27 to March 11 (spring) and November 28 to December 29 (winter), yielding a total of 400 samples (none of which overlapped with COVID-related lockdowns). The spring campaign provided 142 valid samples (from a total of 152); 1 sample lacked parallel measurements, while 9 were omitted from the analysis due to high scaled residuals (93.4% coverage). The winter campaign provided 258 valid samples (from a total of 267); 4 samples lacked parallel measurements, and 5 outliers with high residuals were omitted from the analysis (96.6% coverage).

2.2 EC/OC Concentrations

Concentrations of elemental carbon (EC) and organic carbon (OC) were determined using a field semi-continuous analyser (Sunset Laboratory Inc., USA; Bauer et al., 2009) connected to a PM$_1$ inlet at a flow rate of 8 lpm. Samples were collected on quartz fibre filter discs (2.01 cm$^2$) at two-hour intervals (set to UTC), including the thermal–optical analysis, which lasts approximately 15 minutes. The analysis was performed using the shortened EUSAAR2 protocol (Cavalli et al., 2010). Automatic optical corrections for charring were made during each measurement, and a split point between EC and OC was detected automatically (software: RTCalc726 with linear fit laser/temperature correction, Sunset Lab.). The instrument was equipped with a carbon parallel-plate denuder (Sunset Lab.) to remove volatile organic compounds so as to avoid a positive bias in the measured OC. Instrument blanks were measured once per day at midnight, representing a background instrument response without filter exposure. To include pyrolyzed OC (Pyr.OC), which overlaps the EC1, EC2, and sometimes EC3 fractions, the following fractions were used: OC1, OC2, OC3, OC4, Pyr.OC, EC(1-3)-Pyr.OC, and EC4 for subsequent analyses.
2.3 Elemental Composition

Elemental composition was analysed using an Xact625i Ambient Metals Monitor (Cooper Environmental Services, OR, USA), an online energy-dispersive X-ray fluorescence (ED-XRF) spectrometer at two-hour intervals (set to UTC). A general description of the Xact625 and its specifications can be found elsewhere (Park et al., 2014, Tremper et al., 2018). Three XRF excitation conditions were used, with different voltages, currents, and analysis times ranging from 25 to 48 kV and 1000 to 2000 mA. The Xact was fitted with a PM$_{2.5}$ head with a flow rate of 16.7 (+/- 1%) lpm at two-hour time resolution. Two daily quality assurance checks were performed at midnight (QA energy calibration and upscale tests) for a total of thirty minutes. Monthly maintenance checks included a flow rate check and leak test, while blank filter tests (used for blank corrections), flow calibrations, and elemental standard calibrations were performed tri-monthly. A total of 20 elements were available for the analysis of spring (S, Cl, K, Ca, Ti, Cr, Mn, Fe, Ni, Cu, Zn, As, Se, Br, Rb, Sr, Cd, Sn, Ba and Pb), and 17 for winter (lacking Rb, Cd, and Sn). Supplementary NO$_x$ and NO$_2$ measurements were recorded using a chemiluminescence analyser T200 (Teledyne API, CA, USA) at one-hour time resolution.

2.4 Positive Matrix Factorization Analysis

Positive Matrix Factorization (PMF) is a bilinear multivariate receptor model (Paatero and Tapper, 1994) often used to deconvolute mixtures of PM into their constituent source components. EPA (Environmental Protection Agency) PMF version 5.0.14.21735 was applied to the dataset. Missing values and those under detection limits (DLs) were replaced by (DL)/2, and their respective uncertainties by 5/6 × DL (Polissar et al., 1998). Species were set to ‘strong’, ‘weak’ (uncertainties
triplled) and ‘bad’ (omitted from analysis) based on their signal to noise ratio (Paatero and Hopke, 2003) as well as their Q/Qexp values (an indicator of goodness-of-fit).

Al and Si were added to the dataset using Al/Ca and Si/Ca ratios of 2.73 and 10.64 with respect to Western European upper continental crust data (Wedepohl, 1995). Since total PM$_{2.5}$ mass was not measured, the ‘total’ variable in PMF was calculated similar to Pokorná et al. (2013) using:

\[(NH_4)_2SO_4 = S \times 4.125\] (1)

\[NaCl = Cl \times \frac{25+35.7}{35.7}\] (2)

[Organic Matter (OM)] fractions = OC1 × 1.4, OC2 × 1.4, OC3 × 1.6, OC4 × 1.8, Pyr. OC × 1.8 (3)

[Soil] = 2.20 × Al + 2.49 × Si + 1.63 × Ca + 2.42 × Fe + 1.94 × Ti (Malm et al., 1994) (4)

Therefore, closing the gap towards total PM$_{2.5}$ mass, with the only variable now missing being nitrates.

Different numbers of factors were tested (2 – 8), with the final chosen solutions being 5 for spring and 4 for winter, provided in Tables S1 and S2 in the Supplemental Material. Three uncertainty estimation methods were used: displacement of factor elements (DISP), boot strapping (BS), and bootstrap enhanced by displacement of factor elements (BS-DISP). While BS estimates random errors, DISP analyses rotational ambiguity, with BS-DISP combining both aspects (Paatero et al., 2014). A full list of PMF parameter settings and error estimation results is seen in Table S3.

2.5 Air Mass Trajectories
Air mass back trajectories were calculated using the Hybrid Single Particle Lagrangian Integrated Trajectory (HYSPLIT) (Draxler, 2013; Stein et al., 2015) from the National Oceanic and Atmospheric Administration (NOAA) National Center for Environmental Prediction (NCEP) model. The model was started every three hours from the middle of the mixing layer, and the back trajectories were calculated for 72 hours. The calculations were based on the Global Forecast System (GFS) model with 0.25-degree spatial resolution downloaded as daily files from the official archive (NCEP, 2023) and performed in R (R Core Team, 2023) with the ‘openair’ package (Carslaw et al., 2012).

Local wind speed and direction were measured using an Ultrasonic wind sensor WMT 702 (VAISALA, Finland), with data only available averaged to 2 hours. To add the local wind speed and direction to the analyses, the conditional bivariate probability function method (CBPF) was employed (Uria-Tellaetxe et al., 2014). The median was used as the threshold to distinguish between high and low concentrations of the studied variable. To distinguish between large and moderate sources, a concentration weighted trajectory (CWT) was also calculated (Hsu et al., 2003).

### 2.6 Planetary Boundary Layer Height and Ventilation Index

Using the ERA5 hourly dataset (Hersbach, 2023) provided by the Copernicus Climate Change Service (ERA5, 2017), the planetary boundary layer height (PBLH) and u- and v-wind components were downloaded for the nearest grid point. The PBLH calculation in the model was based on the bulk Richardson number.

From the u- and v- wind components given by the reanalysis, wind speed and direction were calculated, to allow for 1-hour time resolution. The wind speed, together with PBLH, was used for the ventilation index (VI) calculation:
\[ VI = PBLH \times ws, \]  

where \( PBLH \) is in m and wind speed at 10 m in m s\(^{-1}\), resulting in VI in m\(^2\) s\(^{-1}\).

While the ERA5 wind speed data agreed with local data within 5% in both wind speed and wind direction, some scatter between the data was observed (\( r^2 = 0.60 \) for wind speed and \( r^2 = 0.65 \) for wind direction), and thus the ERA5 data was only used for the VI calculations and local wind speed and wind direction were used for all other analyses.

### 3. RESULTS AND DISCUSSION

#### 3.1 Descriptive Statistics

Elemental concentrations for both seasons are displayed in Fig. 2 (see Table S4 for a comprehensive summary). The average temperature and rainfall for spring and winter were 4.6\(^\circ\)C and 45 mm, and 2.5\(^\circ\)C and 17 mm, respectively (CHMI, 2020).

Figure 2. Elemental concentrations for both seasons with error bars representing analytical uncertainty.

#### 3.2 Comparison of Seasons

Total PM\(_{2.5}\) concentrations varied between the two seasons, as seen in Table S4 (total 5.9 and 15.8 \( \mu g \) m\(^{-3}\) for spring and winter, respectively), and were consistent throughout all variables aside from Br (due to the presence of salts). PM values were lower than those found in a Czech rural background station for spring and winter (14 and 22.5 \( \mu g \) m\(^{-3}\), respectively) (Schwarz et al., 2016), a suburban site in Prague during winter (27.6 \( \mu g \) m\(^{-3}\)) (Kozáková et al., 2018), two suburban sites in...
Prague for the whole year (24.4 and 25.1 µg m\(^{-3}\)) (Schwarz et al., 2019), and an urban site in Brno, an eastern Czech city, for spring and winter (16 and 30.4 µg m\(^{-3}\), respectively) (Cigánková et al., 2021). However, nitrate data were missing from the total PM\(_{2.5}\) concentrations, previously observed to make up 13% and 18% of PM\(_{2.5}\) mass in a regional background site of the Czech Republic for spring and winter, respectively (Schwarz et al., 2016). In addition, further controls have gone into effect to reduce airborne PM.

Figures 3a and b. Average air mass trajectories from cluster analyses and their percentage frequencies for spring (left) and winter (right). Each line covers a 36-hour period.

3.3 Air Mass Clustering

The difference in concentrations between seasons can be attributed in part to the distribution of air masses arrive at the sampling site (Figs. 3a and b). In total, 142 and 258 trajectories were clustered in spring and winter, respectively. Spring was associated with two groups of air masses with oceanic history in the last two days prior to reaching Prague, which had also passed over the British Isles (red) and France (blue). A smaller fraction of the air masses sampled (20%) can be described as a relatively stagnant situation over Central Europe, with air masses recirculating over the area (green). Meanwhile, elevated concentrations in winter were likely due to a period mainly associated with air masses passing over areas with substantial industrial activity and other anthropogenic emissions, especially characterised by higher emissions of S, Cr, K, As, and OC/EC fractions, such as those from South-East Europe (green), followed by air masses from the southwest (red). Sampled air masses also included those coming from Scandinavia (blue), as well as a small percentage of oceanic masses from
the west (turquoise). The abundance of marine air masses sampled in spring and the lack of Central European air masses likely explain the lower elemental concentrations.

The large effect of air mass influence (marine vs. continental) on aerosol concentrations in both the Czech Republic and Prague itself has been documented several times in our previous works, helping to explain the differences in total PM between the two datasets (Schwarz et al., 2016 and 2019). Boundary layer heights also differed greatly between the two seasons, with the resultant ventilation indexes (1747 and 736 m² s⁻¹ for spring and winter, respectively) further explaining the differences observed in elemental concentrations, especially in terms of locally sourced emissions (Pan et al., 2019).

3.4. Source Identification

Four common factors were identified for both seasons: i) Road dust/soil, characterised by crustal elements resuspended by traffic activity; ii) Local heating, a product of domestic heating using biomass and coal; iii) Secondary inorganic aerosol, representing the long-range transport of sulphur emissions, and iv) Traffic, related to both tailpipe and non-tailpipe emissions from traffic. An additional factor identified during spring was v) Road Salt, arising from the application of road salts during lower temperatures. PMF analysis results for both seasons are seen in Figs. 4 and 5.

3.4.1 Road Dust/Soil

The Road dust/soil factor contained markers of crustal dust, including Al, Si, Ca, Ti, and Mn (Taylor et al., 1995). Additionally, elements stemming from other urban sources including non-tailpipe emissions, such as Cu, Ba, Zn, and As, were observed. Road dust compositions are affected by the deposition of traffic-emitted particles (Tanner et al., 2008; Chang et al., 2009; Duong et al., 2011;
Vlasov et al., 2022). While spring was more associated with crustal elements, the winter factor contained elevated concentrations of Zn, As, Cr, and Ba, indicating greater mixing of crustal dust with urban dust and traffic emissions. This observation is also linked to higher wind speeds in spring (average 4 vs. 2.8 m s⁻¹), permitting more soil suspension and long-range transport of dust. Winter material was more local in nature, largely road dust emissions that are often dependent on meteorological conditions (Etyemezian et al., 2003), also given the placement of soil/gravel on roads during snow or ice conditions in Prague. The winter road dust/soil profile also included a significant fraction of the Cl explained variation suggesting the mixture of road salt into the road dust during this period. Road salts are placed on Prague streets in the form of crystalline NaCl above -5°, aqueous CaCl₂ below -15°, and a combination of the two at intermediate temperatures. The mass percentage contributions from Road dust/soil for spring and winter were 21% and 31%, respectively. Additionally, a clear diel pattern can be observed in spring, with total concentrations of the factor peaking during the morning and afternoon traffic peaks, while in winter, concentrations were elevated between the two traffic peak times without a drop in between (Fig. S3).

3.4.2 Local Heating
The *Local heating* factor represents domestic heating emissions in Prague and the surrounding areas. High emissions of K (63% and 35% total K emissions for spring and winter, respectively), Zn, As, and OC and Pyr.OC are observed, as well as emissions of EC(1-3)-Pyr.OC. Wood combustion is clearly seen through K, Pyr.OC, and OC and EC fractions (Samara, 2005; Li et al., 2018; Liang et al., 2021). Evidence of coal combustion is visible due to high As emissions in both seasons, a tracer of coal burning in Europe (Burmistrz et al., 2018), and the presence of Zn (Schleicher et al., 2023). The lack of S in the factor further points towards local or regional sources due to the lack of time required to oxidise SO₂ to SO₄²⁻ (Wang et al., 2020). Additionally, emissions of Br can be linked to biomass burning (Manö et al., 1994).

Coal and biomass burning are common means of domestic heating in Central Europe, specifically in the Czech Republic (Sillanpää et al., 2005; Rogula-Kozłowska et al., 2016; Kozáková et al., 2019), and the combustion of lignite (brown coal) and anthracite (black coal) in neighbouring countries (Pokorná et al., 2018). *Local heating* contributed 39% and 20% to total mass for spring and winter, respectively, representing the dominant factor for spring, commonly observed during heating seasons in the region (Pokorná et al., 2013). Since temperatures between the seasons were very similar, the higher relative contribution of *Local heating* in spring is mainly driven by the prevalence of cleaner air masses of marine origin; therefore, similar local heating emissions caused a higher share in the whole aerosol immission budget.

Low to medium wind speeds imply the factor was mainly of local nature (CBPF plots in Fig. S4). Diel patterns show changes in boundary layer height (BLH) from day-time to night-time, with night-day ventilation indexes ranging from 77-10351 m² s⁻¹ (spring) and 14-6550 m² s⁻¹ (winter), further justifying the case for the factor being locally sourced.
3.4.3 Secondary Inorganic Aerosol

High elemental S concentrations in the Secondary inorganic aerosol factor (~75% and 90% for spring and winter, respectively) point to aged, long-distance transported sulphate emissions (including coal-fired power plants) providing sufficient time for the oxidation of SO$_2$ from S(IV) to S(VI) (Wang et al., 2020). The factor was also associated with high OC and EC (Kang et al., 2011; Cesari et al., 2021), As (Guo et al., 2004), and Br (Burmistrz et al., 2016; Lee et al., 2018), all of which are representative of both brown coal and black coal often used in coal power plants in Central and Eastern Europe. Although Flue Gas Desulphurisation (FGD) systems are implemented throughout Europe, they are not typically 100% efficient (Córdoba, 2015).

Evidence of wood combustion during winter is seen in terms of the presence of K (Liang et al., 2021) pointing to domestic heating sources. The factor contributed highly to PM$_{2.5}$ for both spring and winter (20% and 36%), and while this was similarly observed in the Czech Republic (Pokorná et al., 2013; Kozáková et al., 2019), the contributions may be higher than expected.

HYPLIT back trajectories and CWT/CBPF plots aid in analysing the factor (Figs. 3 and S5). CWT and CBPF plots for this factor are in agreement with current coal-fired power plant locations in Europe (GEM, 2023). North-westerly winds were correlated with Secondary inorganic aerosol during spring, with emissions being transported from over Germany and Western Poland (Fig. 3b), which as of 2023, operated 58 and 42 coal plants, respectively. During winter, winds originated from the southeast over the Balkan region as well as part of the Mediterranean (Figs. 3 and S5). In particular, the CWT plot links emission sources in Romania and Hungary, which currently operate 8 and 3 coal plants, respectively. Further, the air masses passed over Eastern Czech Republic, an area with many
industry and metallurgy complexes (Pokorná et al., 2015; Pokorná et al., 2016; Leoni et al., 2018; Pokorná et al., 2018; Kozáková et al., 2019). Two peaks, from 07/03 to 08/03 and 10/12 to 11/12 are visible for this factor, correlated with Central Europe air masses (see Figs. S1 and S2).

3.4.4 Traffic

The Traffic factor, included emissions from road traffic, tram, and train activity. Traffic was associated with both tailpipe and non-tailpipe traffic emissions, despite being sampled at PM$_{2.5}$ (Pant et al., 2013). Zn was emitted from the combustion of lubricating oil, including zinc dithiophosphate (Chueinta et al., 2000; Andrade et al., 2012) and some tyre wear particles (Schleicher et al., 2023), due to zinc oxide being a component of tyres as a vulcanising agent (Jeong et al., 2022). Cu and Fe emissions found in the Traffic factor originate from brake wear particles (BWP) (Johansson et al., 2009; Gietl et al., 2010; Harrison et al., 2012).

High EC emissions were observed during both seasons, with 70% and 40% of EC4 emissions being found in the Traffic factor for spring and winter, respectively, indicating the activity of diesel vehicles; during winter, the contributions of OC are linked to gasoline vehicle activity (Li et al., 2014; Demir et al., 2022). Direct tailpipe emissions, usually found in PM$_{2.5}$, are dependent on vehicle fuel type (gasoline or diesel), age of the engine, and the condition of various internal components such as the catalytic converter, engine, and exhaust pipe, as well as the presence of diesel particulate filters (DPF) in diesel cars (Abu-Allaban et al., 2003; Tzamkiozis et al., 2010; Pant et al., 2013). EC emissions were likely emitted by diesel vehicles lacking DPF filters, which have been seen to dominate urban emissions in some cities (Shah et al., 2004; Kelly et al., 2012) and ten times higher PM emissions than gasoline vehicles (Alves et al., 2015). Emissions of Cr, Mn, and Ba seen in the factor can be attributed
to engine wear particles, lubricating oils, as well as other metals and alloys found in engine parts (Harrison et al., 2001; Thorpe et al., 2008; Nuchdang et al., 2023).

Additionally, clear diel trends for both spring and winter, representing morning and afternoon traffic peaks during the weekdays, are visible in Fig. S6. Supplementary NOx measurements showed high correlations between NOx and the Traffic factor ($r^2$ values of 0.76 and 0.54 for spring and winter, respectively; see Fig. S8), a strong marker of road traffic emissions (Beevers et al., 2012).

Traffic contributed 12% and 13% to total emissions for spring and winter, respectively. These values are far lower than those found in a multi-city study on curbside PM$_{2.5}$ analyses (40-60% traffic contribution) (Querol et al., 2004). However, these are similar to urban studies in Berlin (Pültz et al., 2023), Ostrava, a city in the Czech Republic (Pokorná et al., 2015), Athens (Diapouli et al., 2022), and a previous study near the current site in Prague, albeit at PM$_{10}$ (Schwarz et al., 2019). However, since traffic is considered to be the main polluter in Prague by the Czech Hydrometeorological Institute (CHMI, 2021), the contribution of traffic to total emissions is in the low range.

3.4.5 Road Salt

The Road salt factor identified during spring reflects road salting activities characterised by Cl, accounting for 9% of total mass during the season. While evidence of road salting in the winter Road dust/soil factor is seen by the presence of Cl, the combination of oceanic air masses during spring (Fig. 3a) and road salting allows the factor to emerge on its own. Diel patterns indicate the resuspension of road salt through traffic activity (Fig. S7).
4 CONCLUSIONS

This study used high-time resolution (2-h) online data of EC, OC, and ambient metals in the fine PM fraction for the spring and winter seasons of 2020 at an urban traffic site in Prague, Czech Republic, for source apportionment analyses. Backward air mass trajectories revealed air masses sampled originated from Central, Western, and South-Eastern Europe, as well as from the Atlantic Ocean. Four common factors were found between the seasons, namely: Traffic, Local heating, Secondary inorganic aerosol, and Road dust/soil, with a fifth Road salt factor being observed in spring.

- Source emissions were mostly local with some contribution from distant SIA.
- Higher concentrations of PM$_{2.5}$ were observed during winter, arising from both lower boundary layer height and anthropogenic-influenced air masses originating from South-East Europe.
- Both tailpipe and non-tailpipe emissions were evident, characterised by metals of non-tailpipe origin and organic and elemental carbon from tailpipe emissions.
- Local heating emissions were observed for both seasons, although dominant in spring, and were characterised by K, S, and OC/EC from biomass and coal burning.
- Long-range emissions from coal-fired power plants were observed in high concentrations, particularly during winter, when air masses originated from the southeast.
- Although direct traffic emissions were relatively low in terms of mass concentration, traffic had further involvement through the suspension of road dust, made up of both crustal and anthropogenic-sourced elements, and road salt.

CRediT Author Contributions

ACKNOWLEDGEMENTS

This work was supported by the by the Ministry of Education, Youth and Sports of the Czech Republic and the European Commission under the Horizon 2020 – Research and Innovation Framework Programme, (ACTRIS IMP - H2020-INFRADEV-2019-2), Grant Agreement no.: 871115 and LTAUSA19006. We are grateful for the support by the Czech Hydrometeorological Institute through collaboration and for ECM Eco Monitoring for providing us with their Xact625i used in the project.

DISCLAIMER

The authors declare no conflict of interest.

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Figure 1. Sampling site in Prague, Czech Republic.
Figure 2. Elemental concentrations for both seasons with error bars representing analytical uncertainty.

Figures 3a and b. Average air mass trajectories from cluster analyses and their percentage frequencies for spring (left) and winter (right). Each line covers a 36-hour period.
Figure 4. PMF source profiles for spring (left) and winter (right). Bars represent normalised mass concentrations (µg of constituent µg of PM mass), open circles represent mean DISP (displacement) values, error bars represent maximum and minimum DISP values, and black squares represent explained variation.
Figure 5. Factor percentage mass contributions to total PM2.5.