**Levels and drivers of urban black carbon and health risk assessment during pre- and COVID19 lockdown in Augsburg, Germany**

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**Abstract**

This study aimed to evaluate the levels and phenomenology of equivalent black carbon (eBC) at the city center of Augsburg, Germany (01/2018 to 12/2020). Furthermore, the potential health risk of eBC based on equivalent numbers of passively smoked cigarettes (PSC) was also evaluated, with special emphasis on the impact caused by the COVID19 lockdown restriction measures. As it could be expected, peak concentrations of eBC were commonly recorded in morning (06:00-8:00 LT) and night (19:00-22:00 LT) in all seasons, coinciding with traffic rush hours and atmospheric stagnation. The variability of eBC was highly influenced by diurnal variations in traffic and meteorology (air temperature (T), mixing-layer height (MLH), wind speed (WS)) across days and seasons. Furthermore, a marked “weekend effect” was evidenced, with an average eBC decrease of ~35% due to lower traffic flow. During the COVID19 lockdown period, an average ~60% reduction of the traffic flow resulted in ~30% eBC decrease, as it was the health risks of eBC exposure was markedly reduced during this period. The implementation of a multilinear regression analysis allowed to explain for 53% of the variability in measured eBC, indicating that the several factors (e.g., traffic and meteorology) may contribute simultaneously to this proportion. Overall, this study will provide valuable input to the policy makers to mitigate BC pollutant and its adverse effect on environment and human health.

**Key words:** black carbon, urban pollution, lockdown effect, health risk

**1. Introduction**

Carbon-containing particles make up a large portion of atmospheric aerosol (Zanatta *et al.*, 2016), and are mostly composed of organic matter (OM), mostly secondary in origin, and primary black carbon (BC) from incomplete combustion (Poschl, 2005). Global emissions of BC reached 8.6 Tg in 2017 (Xu *et al.*, 2021), and are likely to continue increasing. Recent estimations attributes to BC warming effect of +0.3 W/m2 (IPCC, 2021). Moreover, since 2012, the International Agency for Research on Cancer (IARC) has classified BC-containing diesel soot as group 1, carcinogenic to humans (IARC, 2012). European studies evidenced associations between BC concentrations and average life expectancy, demonstrating that reducing BC concentrations can extend the average life expectancy by 4 to 9 times higher compared to reducing PM2.5 concentrations (Grahame *et al.*, 2014). Official emission inventories for EU-27 reported that 26% of BC emissions are attributed to road transport, 39% to residential, commercial and institutional sources, 12% to waste management, and 9% to industrial sources (EEA, 2021).

The terms elemental carbon (EC) and BC are equivalent in terms of type of PM component. Usually, EC is used when the content of this carbonaceous component is determined using thermos-optical-transmission methods (Cavalli *et al.*, 2010). While BC is used when determined by optical (absorption) methods (Petzold *et al.*, 2013). However, to convert the BC absorption units into mass concentration factor (the mass absorption cross-section) has to be applied. This factor is determined by comparing EC determinations with absorption measurements. In this scenario, (Petzold et al., 2013) suggested using the term “equivalent” BC or eBC.

Notably, the health effects of eBC were reviewed elsewhere (van der Zee *et al*., 2016), and short-term epidemiological studies were found to provide sufficient evidence that daily changes in eBC were associated with short-term changes in health (all-cause and cardiovascular mortality, and and cardiopulmonary hospital admissions) (WHO, 2012). However, they also concluded that eBC or EC may not be a major directly toxic component of fine PM, but it may operate as a universal carrier of a wide variety of chemicals of varying toxicity to the lungs, the body’s major defense cells and possibly the systemic blood circulation.

In general, the following factors govern the eBC over a location: multiple emission sources (traffic, industry, domestic heating (Liu *et al.*, 2021a), transport processes (with a local origin and long-range transported) (Kant *et al.*, 2020), meteorology (air temperature (T) and relative humidity (RH) (Liu et al., 2021b), and dispersion conditions (wind speed (WS) and mixing layer height (MLH)) (Liakakou *et al.*, 2020). Long-term monitoring of eBC is necessary for estimating their climatic effects and forecasting future situations (Kiran *et al.*, 2018). Moreover, in the early 2020, coronavirus disease of 2019 (COVID19) invaded the majority of the countries, resulting in massive casualties and deaths (Hu *et al.*, 2021, Kontis *et al.*, 2020, Chakraborty and Maity, 2020). Under this scenario, several emergency measures, such as lockdown restriction measures, were implemented to prevent the COVID19 transmission. During the lockdown, urban air pollution markedly decreased in many regions (Sokhi *et al.*, 2021), e.g., India (Das *et al.*, 2021), China (Xu *et al.*, 2020), and Spain (Briz-Redón *et al.*, 2021, Querol *et al.*, 2021) among many others, but not many studies evaluated the impact of these lockdown on levels of eBC, especially in Europe.

Thus, to evaluate the level and phenomenology of urban eBC and to identify major controlling factors, integrated techniques characterizing the temporal variability of eBC in urban environments are relevant to assess urban air quality policies. Moreover, no such investigation has been conducted to Augsburg, Germany, even though comprehensive measurements are available here (Khedr *et al.*, 2022, Li *et al.*, 2018, Gu *et al.*, 2013, Gu *et al.*, 2011). This study aimed to assessing on the eBC phenomenology and major drivers of its concentrations at the city center of Augsburg from 01/2018 to 12/2020, with special attention to these patterns during the COVID19 lockdown and on major differences with the pre- and lockdown period. Finally, the potential health risks of BC exposure based on equivalent numbers of passively smoked cigarettes (PSC) was evaluated (van der Zee *et al.*, 2016). Overall, our findings prove and confirm that meteorology and traffic drive to variations in eBC, as well as lockdown affect.

**2. Material and methods**

**2.1 Measurement Locations**

The measurements were located at Augsburg (48.37N, 10.90E), the third largest city (area, 146.84 km2) in Bavaria, Germany. The city contains about 300,000 inhabitants and about 660,000 residents in its metropolitan area. The average annual temperature and precipitation are 13.2℃ and 767 mm, respectively.

Multi-wavelength measurements of aerosol light absorption were conducted for 3 years (01/2018 to 12/2020) at the aerosol characterization site at the University of Applied Sciences (UAS, 48.36N, 10.91E) in Augsburg, which has been operated jointly since 2004 by the Helmholtz Zentrum München (German Research Center for Environmental Health, Munich) and the Environmental Science Center, Augsburg University (Pitz *et al.*, 2008). It is located about 1 km south-east of the city center and is considered as an urban background site (Cyrys *et al.*, 2008). The site is surrounded by campus buildings, a tram depot, and a small company within a radius of 100 m. The nearest main road is in the north-east at a distance of around 120 m. At 200 m, this site is almost surrounded by residential areas and small parks (Pitz et al., 2008).

**2.2 Measurement Set-Up and Instrumentation**

Aerosol light absorption was continuously measured using an Aethalometer (Magee Scientific, Model AE33, Slovenia), equipped with PM10 inlet impactor. It measures and records the light attenuation by particulate matter continuously deposited on a Teflon-coated glass fiber filter tape at seven wavelengths, ranging from Ultraviolet (UV) to Infrared (IR), with a temporal resolution of 1 min and a sampling flow rate of 5.0 l/min. The Aethalometer converts filter-deposit attenuation by an internal algorithm to equivalent mass concentration (Drinovec *et al.*, 2015). The measured attenuation at 880 nm is interpreted as eBC (Olson *et al.*, 2015). Aethalomter data were recorded with 1 h resolution.

**2.3 Data processing**

The data sets on eBC and other parameters (e.g. meteorology and traffic data) covered from 01/2018 to 12/2020 (36 months). Briefly, the time variation (diurnal, weekly, and monthly) with the average calculation was processed and defined as normal days (ND, COVID19 free period). For 2020, it was defined as the COVID19 period. The annual cycle has been described as four meteorological seasons: winter (December to February), spring (March to May), summer (June to August), and autumn (September to November). The day and night times were set using local time (LT) according to winter time (WT, the last Sunday of October every year) and daylight saving time (DST, the last Sunday of March every year). Generally, the sunrise and sunset times in Augsburg are approximately 07:00 LT in winter, 04:00 in summer, and 17:00 in winter, 20:00 in summer, respectively. The index data were collected and analyzed based on hourly resolution and therefore the clocks were synchronized by hour once the data has been collected.

**2.3.1 Meteorological data and mixing layer height monitoring**

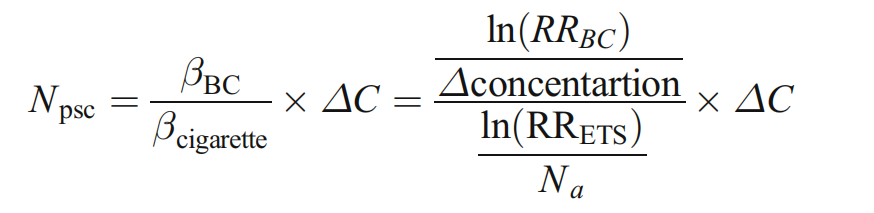
The meteorological parameters, including air temperature (T) and wind speed (WS), were available online from the German Weather Service (Deutscher Wetterdienst, https://www.dwd.de), and located 3 km south to the BC monitoring site (UAS) (48.33N,10.90E), and mixing layer height (MLH) was measured at the UAS site. Generally, the MLH was one of the most important factors for the characteristics of the mixed layer of the atmosphere (Yh and Sumi, 1995). It refers to the near-ground height where the energy and momentum of the material in the atmospheric mixing layer can be fully mixed within a certain period of time due to thermal convection or mechanical force turbulence. It also reflected important parameters for the dispersion of pollutants in the vertical direction (Lv *et al.*, 2020). In this study, an enhanced single-lens ceilometer (CL31, Vaisala, Finland) was used to monitor the MLH (MüNkEL *et al.*, 2012).

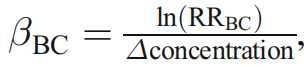
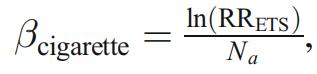
**2.3.2 Traffic density determination**

Traffic volumes were accessed from data recorded by the Augsburg city authorities. All traffic light induction loops were laid out along the entire city's main roads (Fig. S1). During the survey, traffic volumes were counted in each direction. Vehicle counts were summed over 15-minute intervals. In this study, the cumulative number of traffic flows was calculated as the hourly vehicle count for the city. In order to represent the current traffic situation throughout Augsburg, we included all loops in the analysis of this study.

**2.4 Health risk assessment of BC**

A generally method for assessing the health risks of eBC exposure has not yet been established. However, the higher exposure to eBC may cause serious health impacts for humans, such as cardiovascular, cerebrovascular, respiratory diseases, and lung cancer diseases in humans (Ali *et al.*, 2021, Ambade *et al.*, 2021, Gong *et al.*, 2019). Recently, the health risks of passive smoking, environmental tobacco smoke (ETS), were estimated to be equal to the health risk of eBC due to their similar characteristics (Ali et al., 2021, van der Zee et al., 2016, Pani *et al.*, 2020). For example, (1) both have similar health effects, (2) both are mostly unavoidable and involuntar, and (3) both have similar exposure routes, i.e., inhalation (van der Zee et al., 2016). Therefore, this approach has been applied to assess the health risks of eBC using the equivalent numbers of passively smoked cigarettes (PSC) (van der Zee et al., 2016) to establish health risk estimates. Briefly, four health endpoints, including low birth weight (LBW) implying a birth weight < 2.5 kg after 37-week gestation, percentage lung function decrement of school aged children (PLFD) for children, lung cancer (LC), and cardiovascular mortality (CM) for adults, were selected to evaluate the health risks (van der Zee et al., 2016). The equivalent amounts of PSC (Npsc) can be calculated as follows:

(1)

where  is the regression coefficient per 1 μg/m3 of eBC;  is the regression coefficient per cigarette; Δconcentration = 1 μg/m3 of eBC; ΔC refers to the difference between the monitored and background concentrations of aerosol eBC; RRBC is relative risk for the association between eBC and the health endpoint; RRETS means relative risk for ETS exposure; *N*a represents the assumed number of cigarettes, which is 9 for children exposed to parental smoking in relation to the risk of PLFD and 7 for adults in relation to the risk of LBW for infants of non-smoking mothers, CM, and LC (van der Zee et al., 2016).

**2.5 Statistical analysis**

The eBC data are reported as averaged concentrations ± standard deviation (SD). The correlations between hourly averages of meteorological factors and eBC for the whole period and each season in Augsburg were analyzed by the Spearman correlation coefficient. The analysis of significant differences of eBC, meteorological factors, and health risks posed by eBC aerosol at the sampling site in different periods was carried out by one-way analysis of variance (ANOVA) employing Duncan’s multiple range test at *p* < 0.05 and *p* < 0.01, which was carried out using SPSS Software (IBM SPSS Statistics 25, Chicago, IL, USA) and the Data Processing System 9.5 (DPS) (Tang Q Y, 2013). In addition, the multivariate relationships between eBC and the influencing factors (e.g., meteorological factors and traffic) were examined by stepwise multiple linear regression analyses using SPSS Software. The graphs in this study were processed using OriginPro 2021b (9.85, OriginLab Corporation, Northampton, MA, USA.)

**3 Result and discussion**

**3.1 Overview of eBC aerosol in Augsburg**

The year 2020 was special because of two lockdowns due to the COVID19 pandemic. Therefore, the following sections discuss the situation in 2018 and 2019. The year 2020 and the influences of the lockdowns on the eBC percentages will be discussed separately. The average eBC during the 01/2018 to 12/2019 was 1.13±0.97 μg/m3 (Table 1), which was comparable with the previous observations in Barcelona (Spain), Lugano (Switzerland), and Paris (France) (Reche *et al.*, 2011, Healy *et al.*, 2012) and German cities (Table 1) (Sun *et al.*, 2019). However, slightly lower concentrations were found in comparison to Augsburg data for 2009-2014 (with an eBC average of 1.40±1.50 μg/m3).

The average eBC in the four seasons were significantly different (*p* < 0.05). Generally, the eBC showed higher concentrations in autumn (1.23±1.14 μg/m3) and winter (1.49±1.27 μg/m3) than that in summer (0.94±0.44 μg/m3)and spring (1.08±0.85 μg/m3). The main reasons most probably are increased wood burning for residential heating in downtown of Augsburg and differences in dispersion conditions in these seasons (Brandt *et al.*, 2011, Liu, X., *et al.,* 2022). Moreover, the short term variability of eBC in 2018 was comparable with 2019, indicating that eBC pollution incidents have similar trends during both years (Fig. S2). However, the overall eBC was significantly higher (*p* < 0.05) in 2018 than in 2019. Briefly, the significant level of eBC was found in the three seasons (spring, summer and autumn), with the exception of winter. By comparing the main influencing factors in both years, it was found that the MLH was significantly lower (*p* < 0.05) in 2018 than that in 2019 analyzed by ANOVA, which may decrease the dispersion of eBC in 2018. Moreover, the traffic was significantly higher in 2018 than in 2019, further increasing eBC in 2018.

Table 1 Comparison of eBC measured in this study and with that of values reported in the literature.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Site description | City, Country | Observation time | | eBC (μg/m3) | Reference |
| Urban background | Barcelona, Spain | 2009 | | 1.70±0.6 | (Reche et al., 2011) |
| Urban background | Lugano, Switzerland | 2009 | | 1.80±0.9 | (Reche et al., 2011) |
| Urban background | Paris, France | 01-02/2010 | | 1.83±0.18 | (Healy et al., 2012) |
| Urban background | Baden-Württemberg, Germany | 03-05/2009 | | 1.87±0.86 | (Kutzner *et al.*, 2018) |
| Urban background | Leipzig, Germany | 2009-2014 | | 0.80±1.3 | (Sun et al., 2019) |
| Urban background | Dresden, Germany | 2009-2014 | | 0.90±1.1 | (Sun et al., 2019) |
| Urban background | Augsburg, Germany | 2009-2014 | | 1.40±1.5 | (Sun et al., 2019) |
| Urban backg1round | Annaberg-Buchholz, Germany | 2009-2014 | | 1.10±1.8 | (Sun et al., 2019) |
| Urban Industrial | Marylebone, England | 2009 | | 0.70±0.4 | (Reche et al., 2011) |
| Urban Industrial | Baden-Württemberg, Germany | 03-05/2009 | | 1.98±0.87 | (Kutzner et al., 2018) |
| Urban traffic | Baden-Württemberg, Germany | 03-05/2009 | | 5.01±2.30 | (Kutzner et al., 2018) |
| Urban traffic | Leipzig, Germany | 2009-2014 | | 2.30±2.0 | (Sun et al., 2019) |
| Urban background | Augsburg, Germany | 01/2018-12/2019 | Spring | 1.08±0.85 | This study |
| Summer | 0.94±0.44 |
| Autumn | 1.23±1.14 |
| Winter | 1.49±1.27 |
| Annual | 1.13±0.97 |

**3.2 Diurnal variations of eBC aerosol in different periods**

Diurnal variation of pollutants can effectively reflect the role of local anthropogenic activities (Reddy *et al.*, 2012, Zhou *et al.*, 2018). In this study, our results showed that the trends of eBC in 2018 and 2019 were closely similar, thus the data from both years were averaged in this section when discussing the diurnal variation of eBC. Hence, 24 h eBC from 01/2018 to 12/2019 were separated by month and the variability is shown in Fig. 1. A comparably morning peak in eBC occurred ~06:00-08:00 LT about an hour after the local sunrise, which may be attributed to the increasing vehicular emissions. While after morning peak, the eBC was decreased due to increase of the MLH. Lowest concentrations usually were detected in the afternoon (12:00-16:00 LT) in all four seasons due to a combination of increased atmospheric ventilation during the afternoon, increased of the MLH (Fig. 1). It is important to note that during the afternoons, conditions are favorable enough to facilitate the dispersion of road traffic and industrial emissions due to high MLH and WS, hence, the near surface eBC do not build up. While a broad evening peak regularly was observed between ~19:00-22:00 LT. The eBC peak during these hours might be linked to increased vehicular emissions associated with evening rush hour traffic, household heating, and open burning of wood biomass (particularly during winter), as well as unfavorable meteorological conditions (e.g., low MLH) for dispersion, due to decreased convective activity and vertical momentum transport. As a result, the eBC near the surface were enhanced. The weak decreasing trend during midnight to early morning (00:00 to 03:00 LT) can be ascribed to the fact that as the night progresses, anthropogenic emissions reduce and some dry deposition of the near surface particles occurs at low WS resulting in a dip in concentrations compared to the previous evening peak.

In urban areas, eBC are dominated by vehicular emissions, and changes in traffic patterns will impact to eBC trends. Therefore, to further illustrate the diurnal trend of eBC, the weekly diurnal variation of eBC were analyzed. There was significant variation in the mean concentrations of eBC between weekdays and weekends (Fig. S3). During weekends mornings, there were no/weak peaks in eBC (Fig. S3) relative to weekdays because the low traffic density and people are lately asynchronously out in the morning during weekends. However, only a later peak was found on Saturday evening compared to the corresponding time on weekdays.

Interestingly, the diurnal variability of eBC showed somewhat different trends in different months within the same season. For example, in autumn, the morning peaks of eBC occurred around 6:00 LT in Sep. and Oct., while around 8:00 in Nov. Moreover, the nighttime peak was also earlier (18:00-19:00 LT) in Nov. than in Sep./Oct. (19:00-20:00 LT). This phenomenon was due to Nov. is transition time from autumn to winter (Chen *et al.*, 2013, Zeka *et al.*, 2006), which gradually change sunrise-sunset times, and subsequently meteorological indicators (e.g., WS, T, and MLH), thus affecting eBC. This further supported the high influence of meteorological factors on eBC, which gradually change sunrise-sunset times, and subsequently meteorological indicators (e.g., WS, T, and MLH), thus affecting eBC.

Fig.1

Fig. 1. Diurnal variation of average eBC in different seasons during 01/2018-12/2019 (18+19) in Augsburg.

**3.3 Impact of influencing factors to eBC**

3.3.1 Meteorological factors

(Järvi *et al.*, 2008)observed that WS and MLH have strong effect to the eBC. In this study, 24 h eBC and the meteorological trends in different season during the observed period were evaluated (Fig. S4). The results showed that the meteorological indicators exhibited significant seasonal variation, thus affecting the variability of eBC in different season. In general, T and WS strongly influenced the eBC. These revealed extremely similar trends across all time periods, rising gradually in the morning as the sun rises, peaking between 12:00-15:00 LT , and then lowering to a minimum during the night (Fig. S4). During daytime, with increasing solar radiation, T and WS (and MLH a little later) usually increase during the day, which could decrease the eBC. While in the evening, low WS and T inversion led to the accumulation of eBC in the lower atmospheric layer. Increased T and WS resulted in MLH delayed by approximately 1.5 h in the morning and 1-3 h in the evening (Fig. S4).

To further investigate the relationship between meteorological factors and eBC, the correlation between them during 24 h were analyzed (Fig. S5). A negative correlation between T and eBC was significantly observed, which was similar with our previous study in Augsburg (Liu et al., 2021a). In fact, T has an indirect influences on the eBC. For example, the increased contribution of the building heating due to lower T (Liu et al., 2020, Liakakou et al., 2020), and also increased emissions from the traffic due to the cold start and driving conditions at low T (Louis *et al.*, 2016).

In addition, the influence of MLH and WS on eBC is shown in Fig. 2, emphasizing that MLH and WS were associated with the accumulation and dispersion of eBC levels in all seasons (Fig. 2). Briefly, in this study, the lowest hourly average MLH was 390 m in winter, proving that the lower MLH during the heating season was not conducive to the dispersion of eBC, thus leading to accumulation of eBC. Previous studies showed that atmospheric MLH can explain more than 50% of the variation in near-surface pollutant concentrations mainly during winter (Schäfer *et al.*, 2006, SCHäFER *et al.*, 2016). A similar phenomenon was also found in some periods of the autumn (Fig. 2). In contrast, the higher average of MLH was found in spring (~680 m) and summer (~670 m), promoting the dispersion of eBC. Meanwhile, the WS was also negatively related to the eBC, which also evidenced in Fig. S5. This result showed the wind dependence of eBC in entire day, which was similar with our previous studies in small towns (Liu et al., 2020), the city center (Liu et al., 2021a), and the whole Germany cities (Liu *et al.*, 2021b). Accordingly, the higher the WS, the cleaner air involved in pollutants per unit time. Moreover, under the control of the strong wind generated by the cold air transit, the eBC can hardly accumulate rapidly, hence, the eBC can be maintained at a relatively low level for a long time. It is worth noting that at calm conditions (WS < 1.0 m/s), comparably high eBC were still found even with MLH up to about 900 m (e.g., spring and autumn), indicating WS may have a higher effect than MLH in some extent. In short, lower MLH ( < 400 m) and WS ( < 1.0 m/s) resulted in higher eBC in winter, followed by autumn and spring, possibly due to unfavorable vertical dispersion.

Fig2

Fig. 2. Contour plot of the eBC (shaded) averaged for 100 m grids of the mixing layer height (MLH) and 0.5 m/s grids for the wind speed (WS) in different seasons (winter, spring, summer, and autumn) during 01/2018 to 12/2019. The data analysis was performed on the hourly basis.

3.3.2 Traffic volume

According to the previous studies, one of the main sources of eBC in urban areas is traffic emission (Winiger *et al.*, 2017, Şahin *et al.*, 2020), emphasizing the importance of eBC as a contributor to traffic-related health effects by (WHO, 2012). To evaluate the hypothesis of traffic-driven variations in eBC measured over Augsburg, the diurnal and weekly variation of eBC and the traffic volume were separately analyzed in different season (Fig. S6). The result showed that the traffic volume influenced the variability of eBC. For example, the eBC increased gradually with the rise of traffic volume in morning rush hour (~6:00-8:00 LT). Then it gradually decreased after 8:00 until reached its lowest value at ~14:00, although traffic volumes were still increasing. This suggested that eBC, emitted by traffic, was rapidly dispersed during this period due to higher WS and MLH. In the evening rush hour (16:00-17:00), traffic volume reached the highest peak, while the reasonable mixing due to the nocturnal boundary layer not having been well established, thus emissions from vehicular traffic are not sufficiently diluted.

At weekends (Saturday and Sunday), the traffic flows were significantly lower than weekdays in all seasons, missing morning rush hour and maxima in traffic density between 12:00 to 17:00 LT. The phenomenon suggested the “weekend effect”, which contributed to the lower eBC during weekends (Fig. S6). Moreover, this was comparable with the previous studies that the weekend eBC were significantly lower than weekday eBC by 31% in Tirupati of Andhra Pradesh, India (Hussain *et al.*, 2018), and the decrease of eBC in Mexico city, Mexico due to fewer large transport vehicles that are fueled by diesel (Retama *et al.*, 2015).As a result, the eBC did not show any high peaks in the daytime during weekends, but they gradually increased around 14:00 LT until they reached a peak concentration at 21:00-22:00 LT, causing by the accumulation effect on eBC from traffic and heating emissions (Liu et al., 2021a).

3.3.3 The situation in 2020 - lockdown effect on eBC

Due to the impact of COVID19, people's travel activities were restricted in 2020, meanwhile, since 01/01/2020, Augsburg is the first city in Germany with free public transport in a narrow area of the city center, so called city-zone (<https://www.avv-augsburg.de/fahrtauskunft/tickets-tarife/city-zone>). As a result, the traffic flow in the Augsburg city center was significantly reduced compared to the ND (01/2018-12/2019), thus affecting the variation of eBC over the four seasons in 2020 (Fig. S2). Briefly, in 2020, winter had the highest eBC (0.89±0.85 μg/m3), followed by autumn (0.78±0.65 μg/m3), spring (0.61±0.51 μg/m3) and summer (0.40±0.28 μg/m3). Meanwhile, the traffic volume was reduced by about 50% in 2020 compared to the ND period.

In Germany, the first national wide lockdown started from 16/03/2020 to 30/04/2020 (1st lockdown). While the second one started from 02/10/2020 to the early 2021 (in our study we selected 02/11 to 31/12/2020 as 2nd lockdown). To evaluate the hypothesis of lockdown-driven variations in measured eBC, therefore, we explored the variation of eBC during two lockdown periods and their corresponding time during ND in Augsburg. During the 1st lockdown, the diurnal eBC and traffic volumes were strongly significantly decreased (*p* < 0.01) compared to the corresponding ND (Fig. 3a,b). However, the meteorological factors, including T, WS and MLH, were not significantly different during this period (*p* > 0.05) (Fig. 3a,b), indicating that the lower eBC was mainly due to the reduction of traffic volume. During the 2nd lockdown, even though the WS and MLH were slightly lower than the corresponding ND (Fig. 3c,d), however, the diurnal eBC and traffic volumes were still strongly significantly decreased (*p* < 0.01) compared to the corresponding ND (Fig. 3c,d), further indicating the lockdown effect on eBC. Moreover, a prohibition on the sale of fireworks, a ban on fireworks in public places, and a nighttime curfew, may lead to a significant reduction in the number of fireworks discharged in the city, especially between Christmas and New Year eve (24/12-31/12/2020) which may further reduce the sources of pollution and the eBC during the 2nd lockdown (Khedr et al., 2022).

In short, during the lockdown periods, we found an average ~60% reduction of traffic volume that caused a reduction of eBC in Augsburg of ~30%, indicating lockdown played an important role in the reduction of eBC. This finding is close to the one by (Evangeliou *et al.*, 2021) who found a decrease of 40% of eBC in Germany during 2020 lockdown, compared to the same period in the previous 5 years (2015-2019).

Fig3

Fig. 3. The diurnal variations of eBC, meteorological factors (temperature (T), mixing layer height (MLH), wind speed (WS)) and traffic number in two lockdown period and their comparable periods in normal days (a, 1st normal days: 16/03-30/04/2018 and 2019, b, 1st lockdown: 16/03-30/04/2020; c, 2nd normal days: 02/11-31/12/2018 and 2019; d, 2nd lockdown: 02/11-31/12/2020).

3.3.4 Mixed effect and modeling

Based on the effect of meteorological factors (T, WS, and MLH) and traffic volume on the diurnal variations of eBC, the impact on urban eBC is not influenced by a single effect, but rather by multiple factors. For example, in winter, eBC were starting to increase from 5:00 to 8:00 LT due to increased traffic volume, while meteorological factors, such as WS and MLH, were relatively low, accelerating the accumulation of eBC (Fig. S4). After 8:00 LT, the effect of meteorological factors (WS and MLH) on the dispersion of eBC gradually increased and the eBC gradually decreased, even though the traffic volume increased. At around 14:00 LT, the T, WS, and MLH reached their highest values and the eBC reached its lowest peak. The meteorological factors then gradually decreased, while the traffic volume was still increasing, and under this double effect, the eBC gradually increased, indicating that the dispersion effect gradually decreased due to the increasing value of WS and MLH. After the evening rush hour (~16:00-17:00 LT), the traffic volume started to drop, while the accumulation effect was gradually increased due to the decreasing value of the WS and MLH, further contributing to the accumulation of eBC (Fig. S4). In particular, in the winter, the evening eBC was also influenced by eBC emissions from heating activities (Gu et al., 2011, LfU, 2009, Brandt et al., 2011).

Combined with the diurnal correlations, the scatter plot between eBC, and each influencing factors in different season, a certain linear relationship between eBC and each influencing factor was observed (Fig. S5 & S7). Therefore, the stepwise regression method, which can eliminate the variables with large covariance among the independent variables selected into the model, was used to construct a multiple linear regression equation between eBC and influencing factors. The results showed that the independent variables, including T, WS, MLH, and traffic volume, could predict eBC well (R2 = 0.53, Table 2), proposing the following formula.

eBC=1.76-0.03T-2.21\*10-4MLH-0.15WS+3.66\*10-7Traffic.

In addition, the equations for hourly eBC for Augsburg in different seasons were also provided in Table S1. Overall, the R2 for each seasonal model was greater than 0.45, indicating a clear relationship between eBC and the influencing factors.

Table 2. Statistical equations for eBC in Augsburg during the observed period (n = 672, T, air temperature; MLH, mixed layer height; WS, wind speed).

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| Dependent | Independent | Value | Standard Error | *p* | Adj. R-Square |
| eBC | Intercept | 1.76 | 4.08E-02 | <0.01 | 0.53 |
| T | -0.03 | 2.24E-03 |
| MLH | -2.21E-04 | 4.55E-05 |
| WS | -0.15 | 1.62E-02 |
| Traffic | 3.66E-07 | 5.59E-08 |

**3.4 Health risk of eBC**

3.4.1 Health risk of eBC during normal days and lockdown period

eBC is closely associated with several human cardiovascular and respiratory diseases to pose potential risks to the human health (Wu *et al.*, 2018). Health risk assessment model was adopted by previous study to use four health endpoints, including LBW, PLFD, LC, and CM for assessing potential health risks posed by eBC (van der Zee et al., 2016). Therefore, in this study, the health risks exposure to eBC were also evaluated by LBW, PLFD, LC, and CM, and expressed into equivalent numbers of PSC. Health risks posed by eBC at the sampling site were significantly different in different seasons (*p* <0.01).

In general, during the normal days (01/2018 to 12/2019), the average health risks values of LBW, PLFD, LC and CM reached 2.77, 7.86, 1.54, and 3.07 equivalent numbers of PSC in Augsburg (average 3.81 PSC), respectively. Those values were lower than that in Ontario, Canada (Healy *et al.*, 2017), Granada, Spain (Lyamani *et al.*, 2011), Milan, Italy (Invernizzi *et al.*, 2011), and Stockholm, Sweden (Krecl *et al.*, 2017), but still higher than those in Ny-Ålesund, Norway (Markowicz *et al.*, 2017), and Los Angeles, USA (Krasowsky *et al.*, 2016). (van der Zee et al., 2016) reported the health risks exposure to 1 µg/m3 of eBC, being equivalent to 4 PSC per day across the four health outcomes, which higher than the overall estimated health risks of eBC in Augsburg.

The results also evidenced that the potential health risks of eBC during the 1st lockdown were 1.54 for LBW, 0.86 for LC, 1.71 for CM, and 4.38 for PLFD, which were significantly lower (average 2.12 PSC) than the potential risk of eBC in the ND period (2018 and 2019, Fig. S8). The lower potential health risks of eBC were also found in the 2nd lockdown (average 2.88 PSC). This further suggested that the risk exposure to eBC was further mitigated by the lockdown measures to some extent .

3.4.2 Limitations

Although this study quantified the potential risk of eBC based on the equivalent numbers of PSC, the model still has some limitations. For example, there was no literature to quantify the number of cigarettes smoked indoors, and the assumed number of cigarettes (*Na*) in the model, which was based only on assumptions about the time smokers spend at home and their smoking habits from the literature, did not adequately reflect the average number of children and adults who smoke passively at home. In addition, there were still differences between epidemiological studies of ETS and air pollution. For example, health effect estimates for adult ETS also applied to people who did not actively smoke. However, in the epidemiology of air pollution, the entire population should be included. Finally, the future research was still required to evaluate the health risk of eBC assessment for local policy makers and the public in the setting of practical environmental health problems.

**4 Conclusions**

Exposure to eBC, an optically absorbing substance, has been one of the major environmental problems in metropolitan areas. This study analyzed the long-term temporal characteristics of eBC in downtown Augsburg during 2018-2020. Meanwhile, its associations with meteorology, traffic volume, mixing layer height, lockdown effect and the health risk assessment posed by eBC was also evaluated. The seasonal, weekly and diurnal variability in eBC showed significant different due to seasonal changes in the emission rates, traffic volume, atmospheric mixing and dispersion conditions, and boundary-layer dynamics. The average eBC during the winter months were over two folds higher than the summer months, which was attributed to local sources (e.g., traffic emissions) and meteorological factors (e.g., T, WS, and MLH). In addition, an established eBC model in this study with its influencing factors could explain a proportion (53%) of the variability in measured eBC, proving that meteorology and traffic drove to variations in eBC. Overall, our study showed that lockdown effect reduced the eBC and further mitigated the health risks of eBC. It is expected that our findings will help to motivate policy makers, government officials, and the public to increase the number of outdoor monitoring stations. To the best of our knowledge, this is the first report on the health risks posed by eBC in the Germany urban city, which can be used for future monitoring programs and regulations for air pollution control.

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