



Effect of Fireworks, Chinese New Year and the COVID-19 Lockdown on Air Pollution and Public Attitudes

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ABSTRACT

Concentrations of primary air pollutants are driven by emissions and weather patterns, which control their production and dispersion. The early months of the year see the celebratory use of fireworks, a week-long public holiday in China, but in 2020 overlapped in Hubei Province with lockdowns, some of > 70 days duration. The urban lockdowns enforced to mitigate the COVID-19 pandemic give a chance to explore the effect of rapid changes in societal activities on air pollution, with a public willing to leave views on social media and show a continuing concern about the return of pollution problems after COVID-19 restrictions are lifted. Fireworks typically give rise to sharp peaks in PM_{2.5} concentrations, though the magnitude of these peaks in both Wuhan and Beijing has decreased under tighter regulation in recent years, along with general reductions in pollutant emissions. Firework smoke is now most evident in smaller outlying cities and towns. The *holiday effect*, a reduction in pollutant concentrations when normal work activities are curtailed, is only apparent for NO₂ in the holiday week in Wuhan (2015–2020), but not Beijing. Lockdown in Wuhan was characterised by decreases in NO₂, along with more subtle changes in particulate matter and other pollutants. Ozone noticeably increases as there is less NO available for titration, but such change may not be widespread geographically. Beijing, where restrictions were less stringent, showed some improvement in air quality, though this is more difficult to detect, even within the 5th Ring Road.

Keywords: Beijing; Holiday effect; Social media; Weekend effect; Wuhan.

INTRODUCTION

Shifts in the pattern of human behaviour can dramatically change local air quality. Daily or weekly cycles are the most obvious, but the effects of holidays are also well known. Conveniently, social and work patterns differ country to country. Large events such as the Olympic Games or World Cup can change local policies and effectively alter environmental quality (Preuss, 2013). Significant events such as APEC meetings (Wang *et al.*, 2018; Brimblecombe and Zong, 2019) or public celebrations such as the Victory Parade (Wang *et al.*, 2017) may also have a short term impact on air pollution by reducing urban emissions, while political demonstrations and protests can block traffic and also affect air pollution at a local level (Brimblecombe and Ning, 2015; Brimblecombe, 2020a) and may see themselves linked to issues of emissions e.g., *gilets jaunes* (Kinniburgh, 2019).

The *weekend effect*, arising because differing pollutant emissions between weekdays and weekends, reflects a

regular pattern in human activity. This is relatively easy to identify, so received comment in Victorian cities where pollution was argued to derive from cooking Sunday roast dinners (Brimblecombe, 1987). Today, pollutants such as CO, NO_x, NO₂, SO₂ and PM_{2.5,10} are typically found to be higher on weekdays, because of rush hour traffic, work patterns and industrial activities, but also influenced by population and urbanization (Qin *et al.*, 2004; Riga Karandinos, 2005; Pudasainee *et al.*, 2010; Sadanaga *et al.*, 2012; Wolf *et al.*, 2013; Cerro *et al.*, 2014; Porter *et al.*, 2014; Henschel *et al.*, 2015; Malik *et al.*, 2015; Martins *et al.*, 2015). Ozone as a secondary photochemical oxidant, shows more complex behavior: lower on weekdays than on weekends (Tang *et al.*, 2008), although sometimes it can also be higher during the week (Huryn *et al.*, 2014; Xie *et al.*, 2016).

The *holiday effect* refers to changed air pollutant concentrations that arise at vacation times or holidays. This is illustrated by the dramatic changes that come about from the decrease of primary pollutant emissions during the *Day of Atonement* (Levy, 2013) or the celebration of *Mid-Autumn Festival* in Taiwan where families cook on barbecues (Tsai *et al.*, 2015). The holiday effect can be studied over extended periods, often extending beyond a weekend, e.g., New Year or *Spring Festival*, as celebrated in Asia, perhaps the most

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important festivals of the Chinese lunar calendar. Tan *et al.* (2009) revealed the concentrations of NO_x, CO, non-methane hydrocarbons, SO₂ and PM₁₀ in Taiwan from 1994 to 2006 were lower across the New Year holiday than the non-holiday times, although the concentration of ozone was higher, i.e., NO_x titration absent. Chen *et al.* (2014) showed significant reductions of PM_{2.5} and PM₁₀ in 31 Chinese cities, while Gong *et al.* (2014) reported decreased concentrations of key air pollutants.

The New Year is welcomed with large quantities of fireworks, so there is also a significant potential for increased pollution. There have been many attempts by the Chinese government to reduce fireworks since the 1980s, though not codified until 1992 (Zhou and Zhou, 2018). The early morning hours of the New Year can experience some of the highest particulate loadings of the year, and concentrations at some locations can exceed 1000 µg m⁻³ (Huang *et al.*, 2012; Zhao *et al.*, 2014; Kong *et al.*, 2015; Ye *et al.*, 2016; Lai and Brimblecombe, 2017; Ji *et al.*, 2018; Yuan *et al.*, 2020). The firework-derived pollutants are not spatially homogeneous, rather these are dependent on the level of regulatory enforcement and shifts towards public firework displays (Lai and Brimblecombe, 2017; Yao *et al.*, 2019; Lai and Brimblecombe, 2020).

The impact of the global spread of COVID-19 has led to global restrictions on population movement. Although such restrictions are not historically unique (Brimblecombe, 2020b) in the 20th century these have led to emissions reduction (e.g., Safarian *et al.*, 2020; Sharma *et al.*, 2020) from factories and transport systems and a changed electricity demand (Narajewski and Ziel, 2020). Yue *et al.* (2020) suggest carbon emissions decreased by 9.8% in 2020 compared to February–March/2019. Reduced emissions have allowed numerous observations of reduced air pollutants from satellite (Bauwens *et al.*, 2020) and within China (Bao and Zhang, 2020; Cole *et al.*, 2020; Dutheil *et al.*, 2020; Liu *et al.*, 2020; Muhammad *et al.*, 2020; Wang and Su, 2020; Xu *et al.*, 2020; Yue *et al.*, 2020), India (Sharma *et al.*, 2020), South East Asia (Abdullah *et al.*, 2020; Mohd Nadzir *et al.*, 2020), Europe (Lewis, 2020; Tobías *et al.*, 2020), US (Muhammad, *et al.*, 2020) and South America (Dantas *et al.*, 2020; Nakada and Urban, 2020). The work has often been at an early stage and publications difficult to assess (London *et al.*, 2020; Servick and Enserink, 2020). Many are simple comparisons with the preceding months or the same season of 2019, with Cole *et al.* (2020) rightly suggesting a nuance to air quality during lockdown that reporting has failed to acknowledge. Some have argued that peak air pollution concentrations remain (Wang *et al.*, 2020), while Shi and Brasseur (2020) suggest health risks from the increased ozone (Huang *et al.*, 2020), along with sensitization of populations that has been imposed by prior exposure to air pollutants (Wu *et al.*, 2020).

Here we link changes in the pattern of human activity in the early months of 2020, as reflected in air pollution in Beijing and Wuhan. In particular, the study examines the differing effects of holidays, lockdown, and celebratory fireworks on air quality as touched upon by Yang *et al.* (2020). We wish to emphasise a subtlety to changes in air quality imposed by lockdown that are often lost in more sweeping generalisations

in the media (Cole *et al.*, 2020). The work makes use of comments on social media that hint at societal viewpoints, an important consideration in policy making. It establishes differences imposed by recent events, but also considers potential for changes in exposure and reflects on what COVID-19 might mean for longer term policy.

METHODS

This study focuses on Beijing, China's capital (Fig. 1) with a population of more than 21 million in the urban area covering some 4,144 km² and Wuhan (Pop. 11 million; 2018) one of the nine National Central Cities of China (Fig. 1), along with three nearby a prefecture-level cities of Hubei Province: Huanggang (7.4 million; 2017), Ezhou (1.05 million; 2010) and Huangshi (2.4 million; 2010). The air pollution data comes from official records (<http://aqicn.org/city/>) in and around Beijing (12 *urban* within the 5th Ring Road and 11 *suburban sites* outside this Road), from ten sites in Wuhan and three from Huanggang and Ezhou and five from Huangshi. The records are available at hourly intervals for six pollutants: NO₂, PM_{2.5}, PM₁₀, CO, SO₂ and O₃. They typically become available from 2014, although some limited earlier records can be found. The contribution of fireworks to airborne particles requires a correction to be made for other sources. In this study, this was made using CO concentration as fireworks release relatively small amounts (Li *et al.*, 2006). The correction factor was established from the ratio of average PM_{2.5} and CO concentrations, which can be well correlated during non-firework periods (as in Wang, 2015); the non-firework period selected here were the 14 days before the Chinese New Year. The contribution of fireworks to the particulate load was calculated by the subtraction.

The Chinese New Year is set by the lunar calendar as: 10 Feb 2013, 31 Jan 2014, 19 Feb 2015, 8 Feb 2016, 28 Jan 2017, 16 Feb 2018, 5 Feb 2019 and 25 Jan 2020. However, the statutory holiday period in Mainland China starts on the Eve of Chinese New Year and covers seven days. In 2020, this was extended by two further days because of the coronavirus epidemic. Wuhan was at the centre of the pandemic and as it became necessary to contain the disease in Hubei province, the central government suspended all public transport in Wuhan from the morning of 23/January/2020; the first time in history a megacity had entered lockdown. Within hours, travel restrictions were also imposed on nearby Huanggang and Ezhou and in Huangshi the next day. The streets of Wuhan became silent and the once-bustling city a ghost town. Although grocery stores and some shops remained open, most people stayed at home only making the most essential of trips outside. Lockdown ended in Huangshi on the 13/March and in Huanggang and Ezhou on 25/March. Wuhan remained under lockdown until 8/April.

In Beijing, there was no strict lockdown policy, but the government suggested people remain indoors, and while outside adopt *social distancing*. Beijing authorities cancelled events and closed some major tourist attractions, including the Forbidden City and the National Maritime Museum (> 24/January) and the Palace Museum (> 25/January) through to the end of April.

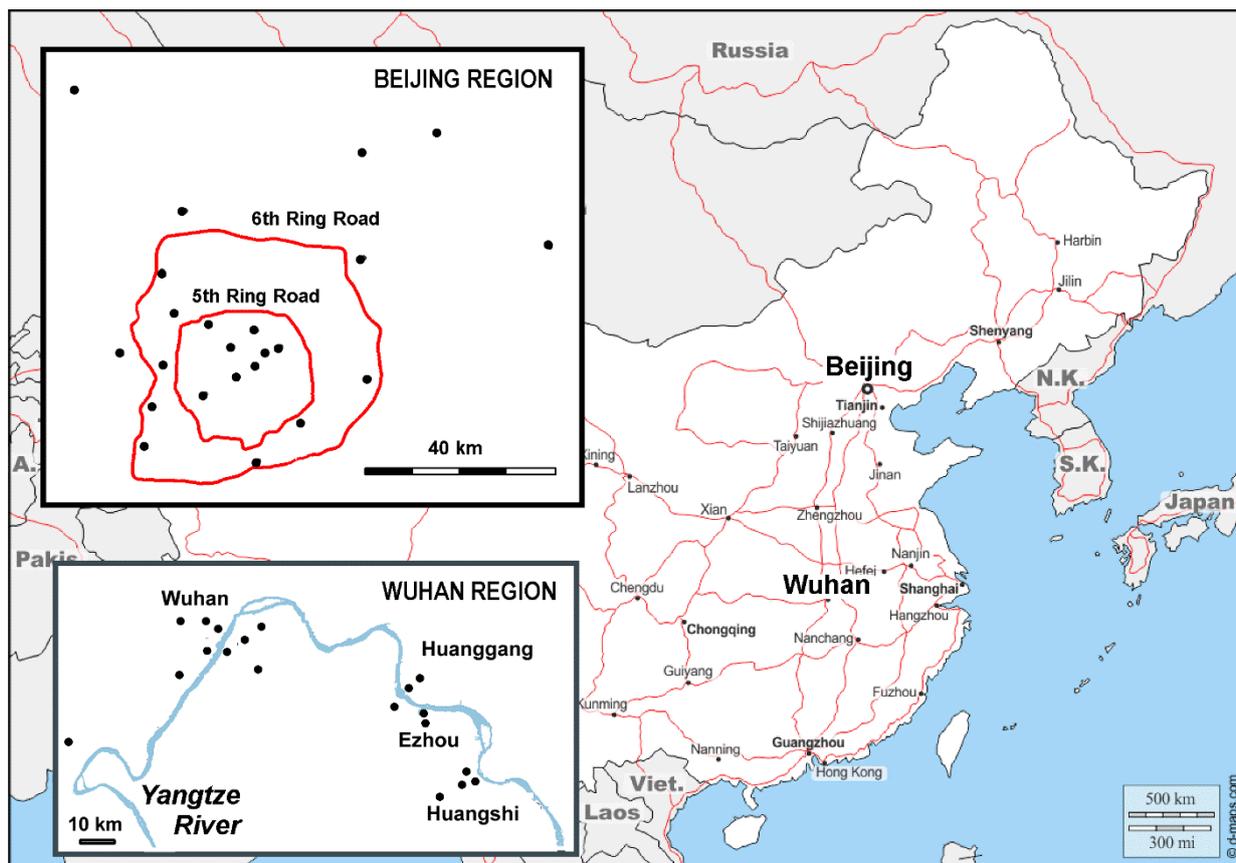


Fig. 1. Map showing the monitoring sites in Wuhan and Beijing.

Hardly surprising, the lockdown and closures led to considerable comment on social media, which provides useful insights on public attitudes (Brimblecombe and Zong, 2019) that can be useful in formulating environmental policies (Deguen *et al.*, 2012). This is especially true with commentary on fireworks as these are difficult to purchase and their use often restricted, so information can be easiest to follow on social media (Lai and Brimblecombe, 2020). We obtained social media content of postings to *SinaWeibo* (<https://www.weibo.com>) using the combined advanced search and the characters “Wuhan fireworks” from 1/January to 30/April, in both 2019 and 2020 and to a lesser extent Beijing. Additionally, attitudes of Wuhan residents to changes in air pollution during lockdown were explored by gathering data with the search terms “Wuhan lockdown”. The data from the two years were compared using the *log-likelihood and effect size calculator* available at (<http://ucrel.lancs.ac.uk/llwizard.html>) from Lancaster University, which uses log-likelihood (LL) to assess where it is probable that the null hypothesis can be rejected and allow us to assess differences in the frequency of occurrence. The more familiar chi-squared test could have been used, but it is difficult to apply where there are few instances of particular concepts found in the social media posts.

Statistical calculations have benefited from *Vassarstat* (<http://vassarstats.net/>). The *t*-test was used in Student’s and Welch’s form, the latter appropriate for unequal variance. The ANOVA calculator (for either independent or correlated

samples) to compare sites for different periods of the year (i.e., before, during and after Chinese New Year) and occasionally, its non-parametric equivalent, the Friedman test (test statistic Q , which approximates to χ^2 the common probability distribution used in inferential statistics), with Tukey’s HSD test used to assess comparisons.

RESULTS AND DISCUSSION

Seasonal and Secular Change in Air Pollutants

Fig. 2 shows the secular change of pollutants in Beijing and Wuhan from 2013 for (a) NO_2 , (b) $\text{PM}_{2.5}$, (c) $\text{PM}_{2.5}$, (d) CO , (e) SO_2 , and (f) O_3 , and also gives hints of the seasonal change. It shows that primary pollutants have typically been in decline in Beijing and Wuhan as they have for much of China (Cheng *et al.*, 2018). These secular improvements in Wuhan are also nicely seen in the plots 2013–2020 from Cole *et al.* (2020), which remove weather effects and for Beijing in Vu *et al.* (2019). Fig. 2 also suggests that in general the early part of the year sees improving air quality, although ozone typically rises through the year and shows summer maxima.

Contribution from Fireworks

The early months of the year are traditionally celebrated with fireworks as part of Spring Festival, but there have been persistent attempts by the government to reduce this pollution source (Zhou and Zhou, 2018). Although these have not

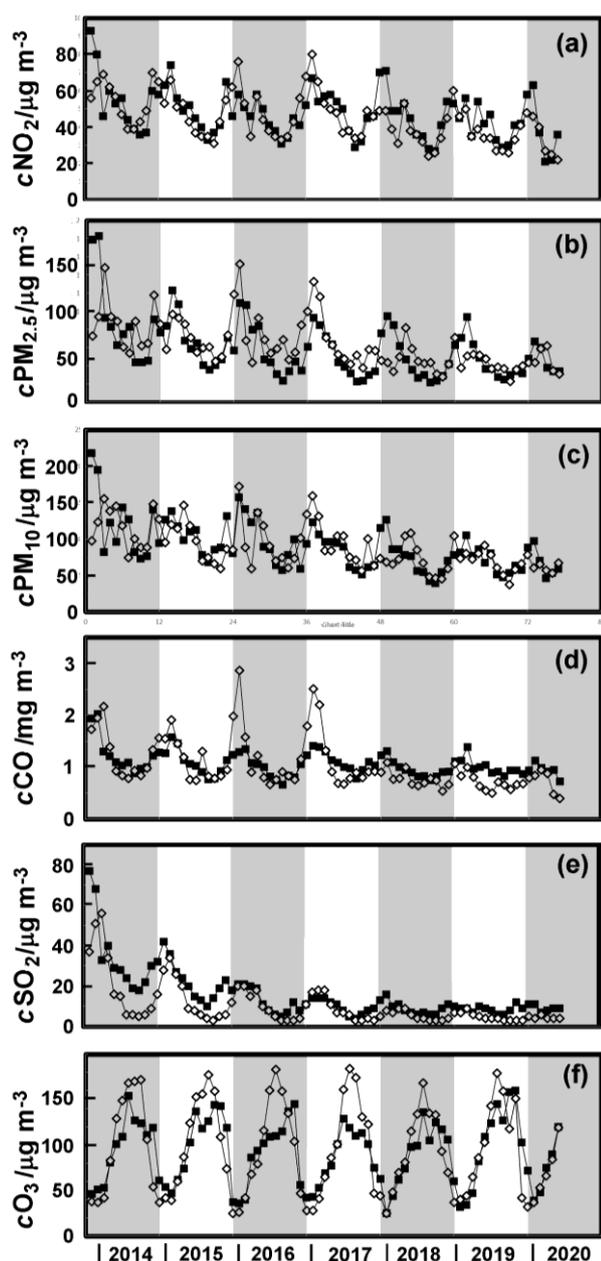


Fig. 2. Monthly average concentrations (a) NO_2 , (b) $\text{PM}_{2.5}$, (c) PM_{10} , (d) CO , (e) SO_2 and (f) O_3 across all sites in Wuhan (squares) and Beijing (open diamonds) from December 2013 to April 2020. Shading denotes 12-month periods.

always been effective, efforts have become more concerted in the present decade, which has seen reductions in emissions even though fireworks remain popular (Yao *et al.*, 2019; Lai and Brimblecombe, 2020). The particulate matter once characteristic of the celebrations in Beijing (Li *et al.*, 2006; Wang *et al.*, 2016; Sun, 2017; Hu *et al.*, 2019) has been reduced in recent years and firework smoke is less evident within the 5th Ring Road (Lai and Brimblecombe, 2020). Fig. 3(a) shows that the sharp peaks in $\text{PM}_{2.5}$ from fireworks typical of the early hours of New Year are no longer apparent in the capital. In 2020 (bold dark curve), there was no evidence of a peak characteristic of earlier years but given

the concerns over public gatherings at the time it is hardly surprising. However, as is typical this better air quality at New Year did not extend to more rural areas (Fig. 3(b)). Peaks in particulate matter can still be seen at rural locations (Cui *et al.*, 2020), with concentrations at almost $900 \mu\text{g m}^{-3}$ (~02:00 hrs at Liulihe). Some posts to Weibo suggested firework displays, although many residents within the 5th Ring Road did not see or hear any, so activities were restricted to more remote areas. There were comments that local government was messaging residents not to use fireworks, rather than giving hints on limiting COVID-19.

In Wuhan, fireworks have also been a source of pollutants (Han *et al.*, 2014; Liu *et al.*, 2017), so subject to official control. In theory, the early morning peak of $\text{PM}_{2.5}$ evident in 2013 should not recur (Fig. 3(c)), yet there was a noticeable peak in 2016 across much of the city. The increase begins just after midnight and is not associated with a parallel increase in CO , so suggests it likely derives from fireworks (Lai and Brimblecombe, 2017, 2020) and supported by some Weibo posts about the illegal fireworks. High particulate concentrations were not evident at Yuanshun Road (site 1334A), near the important Chenhu wetland, the site of the Late Imperial story *A Corpse's Transmutation* by Pu Songling. In the neighboring city of Huangshi (see map Fig. 1) restrictions are in place, but in Huanggang to the southeast of Wuhan fireworks are less restricted, so there are noticeable peaks in $\text{PM}_{2.5}$ at New Year.

The New Year of 2020 was naturally a more restrained event. Although there was a noticeable peak in $\text{PM}_{2.5}$ concentration about 05:00 on 25/January at sites in Wuhan (bold line Fig. 3(c)), it is unlikely to derive from fireworks, because there is a parallel peak in CO , so may be the early morning transfer of supplies and human assistance to the beleaguered city. In 2020, there are small, but noticeable peaks in $\text{PM}_{2.5}$ late on the Eve of the New Year in Huangshi, Ezhou and Huanggang (Fig. 3(d)), but these are smaller than peaks ascribed earlier to non-firework activities, in Wuhan.

Some posts to Weibo from Ezhou said that the sound of fireworks could be heard from rural areas and suggested their decrease in use or the resultant quiet and tranquility; comments frequently found in 2020. While the year of lockdown saw fewer posts about fireworks, many expressed a desire for displays even in such tragic times (Table 1), so the percentage enthusiastic about fireworks rose more than ten-fold (1.4–16.7%). Nevertheless, there were still a few posts, in both 2020 and 2019, mentioning bans on the use of firework, but little comment on illegal sales, which have been a problem in the past (Lai and Brimblecombe, 2020). There were some posts reflecting concern over safety issues or accidents with fireworks and the resultant smoke haze, though not significantly different ($p > 0.05$) between the two years. Although there was little change in the frequency of comment on the cultural relevance of fireworks, social media revealed a significant increase ($p < .0001$) in comments on Nian traditions. It appeared that residents during the pandemic remained attached to ancient notions that burning fireworks would drive away the epidemic, which could be seen as an evil spirit.

There is evidence that cities have much reduced the use

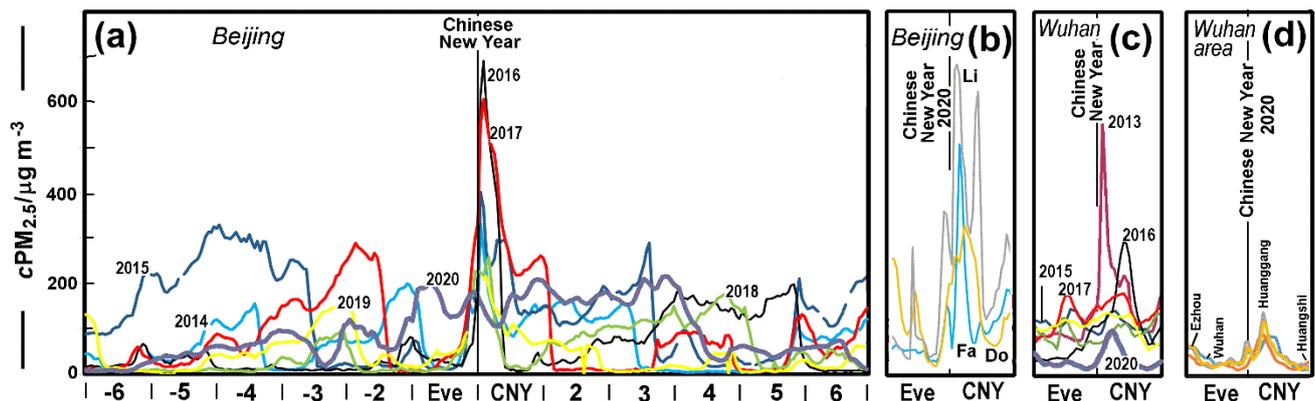


Fig. 3. PM_{2.5} concentrations (a) Averaged for urban and suburban stations in Beijing for six days and six days after Chinese New Year. (b) Particulate matter on the eve and first day of the New Year of 2020 less central sites of Fangshan (Fa- blue line) Donggaun (Do- yellow line) and Liulihe (Li- gray line). Averaged for urban and suburban stations in Wuhan. (c) Averaged PM_{2.5} for sites in Huangshi, Ezhou and Huanggang for the eve and first day of the New Year of 2020. Note: Scale of (b) is slightly different and spans 1000 $\mu\text{g m}^{-3}$.

Table 1. Key concepts identified within raw counts and frequency as a percentage from posts on social media from 1/January–30/April, in 2019 and 2020. Note the log-likelihood value (LL) 5% critical value at LL = 3.84 ($p < 0.05$; denoted *) and 0.01% critical value LL = 15.13 ($p < 0.000$; denoted **).

Concept	2019	%	2020	%	LL
<i>Corpus size: posts</i>	2387		785		
Heard/saw fireworks	41	1.7	145	18.5	223**
Decreasing use (description)	1	0.04	10	1.3	22**
Quiet	9	0.4	18	2.3	21**
Enthusiastic about fireworks	33	1.4	131	16.7	220**
Against fireworks	4	0.2	14	1.8	22**
Ban (description)	67	2.8	37	4.7	6*
Illegal sales (description)	21	0.9	6	0.8	0.6
Concerns safety-accidents	49	2.1	12	1.5	0.9
Firework smoke (description)	12	0.5	5	0.6	0.2
Cultural importance	8	0.3	5	0.6	1.2
Loss of Nian culture	22	0.9	28	3.6	22**

of fireworks, but emissions are still reflected in high particulate concentrations at more rural locations and smaller towns and cities, where the bans have not been set or not enforced. Such regulatory heterogeneity can be found elsewhere, so while sharp peaks in PM_{2.5} are no longer so common in Chinese cities, they remain a characteristic of the less populated areas that surround these cities (Lai and Brimblecombe, 2017, 2019).

As many on Weibo had expressed the desire to see fireworks: “although Wuhan banned the fireworks in urban areas this year, I really hope the government will allow residents to have fireworks in this special year. It represents a victory on this White War”. There was evidence of such celebrations after lockdown ended in early April. Social media suggests that some Wuhan residents went to Ezhou and there are hints of a distinct peak of PM_{2.5} between 19:00 and 21:00 on 8/April (Fig. 4(a)). This is likely to derive from fireworks as tuning the data with CO concentrations fails to remove the peak (Fig. 4(b)), as described in the method. A few posts to Weibo mentioned that they observed some small firecrackers at the end of the lockdown in outlying

suburban areas. Most of the celebrations of the end of lockdown took place in Wuhan a few days later, with a broad peak of pollutants (Figs. 4(c) and 4(d)) in the early morning hours of the 10 April (Good Friday, though Easter is not celebrated in China); this supports claims of fireworks found on Weibo, which even extended to the rural site in the Chenhu wetlands (black line Fig. 4(c)).

Holiday Effect on Air Pollutants

The data were split into three periods (i) the two weeks prior to New Year’s Eve, (ii) the seven days across the holiday and (iii) the two weeks following. The general expectation would be that the holiday week would have lower emissions, which is essentially at the heart of the holiday effect. However, seasonal change occurs across this period so the meteorology is not constant, and the picture can readily become more confused as illustrated in Fig. 5.

In the case of NO₂, the Chinese New Year in Wuhan shows the lowest average concentrations (Fig. 5(a)), although unsurprisingly 2020, with its lockdown after the New Year meant concentrations were low beyond the formal holiday

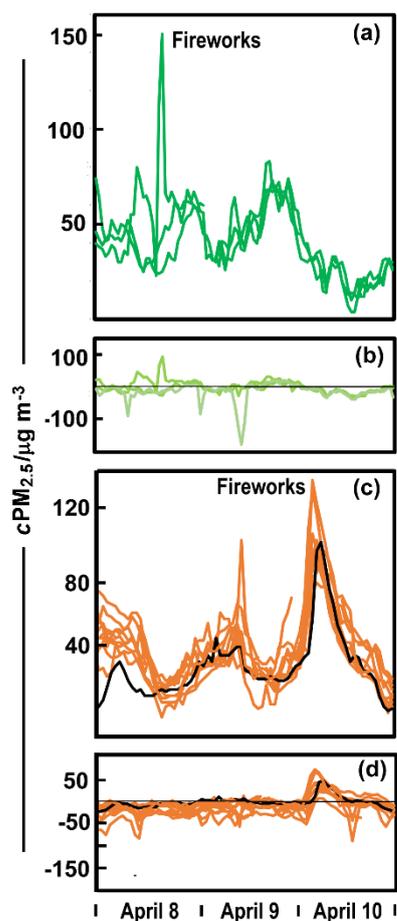


Fig. 4. Particulate matter concentrations on the days after the lockdown ended in Wuhan. (a) $PM_{2.5}$ from the three sites in Ezhou. (b) $PM_{2.5}$ concentrations in Ezhou corrected for non-firework sources using CO concentrations. (c) $PM_{2.5}$ from the nine sites in Wuhan with the remote in Chenhu wetlands as a dark line. (d) $PM_{2.5}$ concentrations in Wuhan corrected for non-firework sources using CO concentrations.

period. The average concentrations from all the sites available for the six years are as means for the three periods: (i) 55.6, (ii) 26.3 and (iii) 40.8 $\mu\text{g m}^{-3}$. An ANOVA test (for correlated samples) suggests the difference is significant $p < .0001$ and the Tukey HSD test shows significant differences ($p < .01$) for all three combinations of pairs. Although ANOVA is fairly robust against deviations from normality the three datasets were checked against the Friedman test (a non-parametric equivalent of the correlated ANOVA), which gave $Q = 41.3$ and ($p < .01$; $n = 55$) and mean ranks for the three periods: 2.8, 1 and 2.2, with the holiday clearly lowest. As 2020 was so unusual in Wuhan the three periods were compared at the nine sites revealing once again the holiday period was the lowest: means (i) 39.7, (ii) 17.8 and (iii) 21.2 $\mu\text{g m}^{-3}$. In this case the Tukey HSD test revealed these differences of lower significance: (i):(ii) $p < .01$ (i):(iii) $p < .01$ and (ii):(iii) $p < .05$. The lower significance of the difference during the New Year and the following lockdown is visually apparent from Fig. 5(a), but the differences are supported by the Friedman test ($Q = 16.2$; $p < .0003$; $n = 9$ giving mean ranks

for the periods as: 3, 1.1 and 1.9.

The picture for particulate matter is less clear (Figs. 5(b) and 5(c)). The $PM_{2.5}$ concentrations over the New Year holiday are higher than the period that follows for all years, but inconsistent when compared with the period before; a situation somewhat similar for PM_{10} . Despite the lack of visual clarity in Fig. 5, the ANOVA suggests the holiday period is typically in the middle: 85, 76 and 63 $\mu\text{g m}^{-3}$; a significant difference $p < .0001$ with the Tukey HSD differences showing significant differences ($p < .01$) for all three combinations of pairs. The statistics for other pollutants are listed in the top rows of Table 2. In Wuhan seasonal change imposes a general decline in pollutants except for ozone which increases (in line with Fig. 2). However, as shown in Figs. 5(d) and 5(f), the holiday effect is rather weak for the other pollutants in Wuhan. A confused picture is also found for Beijing in NO_2 concentrations (Fig. 5(g)), where the holiday is not always a period of lower pollution. In the case of $PM_{2.5}$, the holidays tend to be rather higher, these higher concentrations, perhaps resulting from fireworks. The statistical analysis for the pollutants in Beijing is given in the lower part of Table 2. Overall, the holiday effect is not especially distinctive and all what we can conclude is that Wuhan has lower NO_2 in the holidays and Beijing higher particulate concentrations.

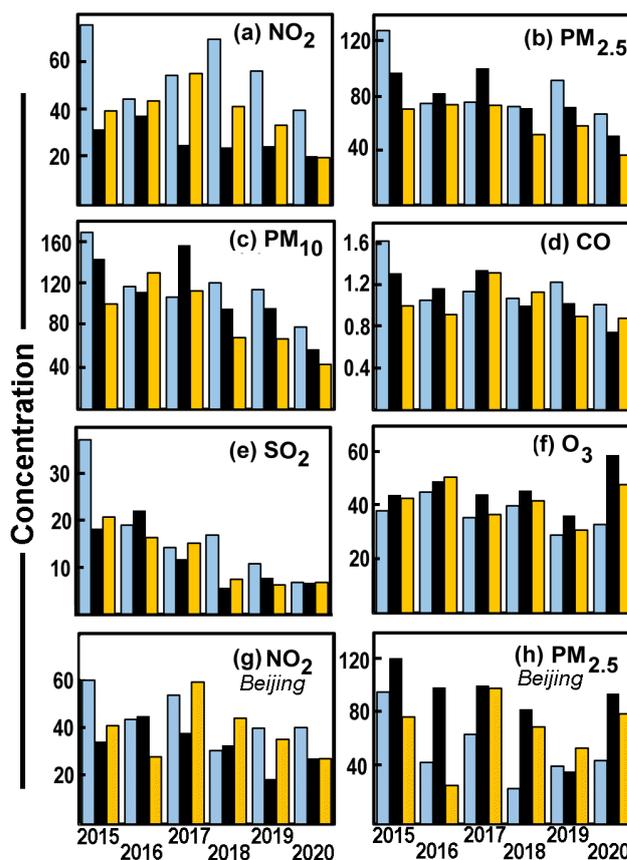


Fig. 5. Average pollutant concentrations before, during (black bar) and after Chinese New Year 2015–2020 for (a) NO_2 , (b) $PM_{2.5}$, (c) PM_{10} , (d) CO, (e) SO_2 and (f) O_3 in Wuhan and (g) NO_2 and (h) $PM_{2.5}$ in Beijing. Note: concentrations as $\mu\text{g m}^{-3}$, except CO as mg m^{-3} .

Table 2. Average pollutant concentrations before, during and after Chinese New Year (2015–2020) used in the ANOVA, on correlated samples, with the Tukey HSD test indicating the significance (p) of the differences between the New Year, and before and after. Note: ns = not significant concentrations $\mu\text{g m}^{-3}$, except CO as mg m^{-3} , Beijing is taken from the 23 urban and suburban sites.

Pollutant	Before	p	During	p	After
<i>Wuhan</i>					
NO ₂	56	< .01	26	< .01	41
PM _{2.5}	85	< .01	76	< .01	63
PM ₁₀	119	< .01	103	< .05	92
CO	1.19	ns	1.15	< .01	1.05
SO ₂	18	< .01	11	ns	13
O ₃	38	< .01	44	ns	43
<i>Beijing</i>					
NO ₂	38	< .01	47	ns	48
PM _{2.5}	54	< .01	92	< .01	69
PM ₁₀	69	< .01	125	< .01	71
CO	1.13	ns	1.19	< .01	1.08
SO ₂	16	< .01	19	< .01	13
O ₃	38	< .01	47	ns	48

Effect Lockdown on Air Pollutants/Quality

Fig. 6 shows the concentration changes in a number of air pollutants in Wuhan and Beijing over the first four months of 2019 and 2020. Figs. 6(a) and 6(b) suggest a big change in NO₂ averaged from all Wuhan sites after lockdown, compared with 2019, in line with observation of other research groups (Cole *et al.*, 2020; Dutheil *et al.*, 2020; Muhammad, *et al.*, 2020; Wang and Su, 2020). It is also possible to notice an apparent reduction in variance. However, the average concentrations are different, so this is not a fair test, hence the coefficient of variation (c_v i.e., standard deviation/mean) for the individual sites were compared. The average values of c_v for NO₂ across Wuhan in 2019 and 2020 were 0.425 and 0.364, although a paired t -test suggests this difference is not especially significant ($p_1 = 0.05$).

However, the rural site at Yuanshan Road is odd, with c_v in 2019 and 2020: 0.48 and 0.56, but concentrations were low there. If this site is eliminated the average c_v for 2019 and 2020 become significant at the $p_1 = 0.022$. It hints that emissions may show less variation under lockdown, perhaps because larger more constant sources, (power stations) dominate over fluctuating sources, such as small factories or traffic in the absence of congestion etc. An analysis of the posts to Weibo under lockdown reveal that of almost 30000 posts, only 13 mentioned the air pollution (three mention the situation beyond China). Some posts showed an awareness of the NASA satellite observations that NO₂ had decreased during lockdown. One post said that "Blue sky and white clouds? The lockdown had been enforced for more than one month, but I still cannot see any blue. so serious is pollution for my city... " and another post: "Well, Wuhan is to be relieved of the lockdown policy tomorrow, but I still can feel the air pollution." This hints at a familiar concern over the temporary nature of these improvements (Brimblecombe and Zong, 2019). After the lockdown period only 9 of some 8000 posts, mentioned air pollution, showing interest in improved air quality world-wide. One post said: "When people stayed at home avoiding COVID-19, air quality in New

Delhi improved, reflecting that air pollution is associated with production. While the air pollution in Wuhan did not vanish during lockdown, so ...it is not just related to industry." Even though official reports said air pollution declined significantly, Wuhan commentators felt it was still there.

Changes in measured PM_{2.5} are not so apparent from Figs. 6(c) and 6(d), and those for O₃ (Figs. 6(e) and 6(f)) not visually impressive. The concentrations of NO₂, PM_{2.5} and O₃ averaged from the urban and suburban sites in Beijing across similar periods are shown in Figs. 6(g)–6(l). Once again, the changes between the years are hardly striking. Reductions in road use in Beijing were substantial. Beijing Municipal Bureau Statistics showed that road movement of people dropping from 52 million in February 2019 to only 8.2 million in 2020 and rose from 10.3 in 2019 to 7.3 million tons in 2020 (<http://tjj.beijing.gov.cn/>). Nevertheless, large such reductions in traffic flow were likely more apparent in urbanised areas.

Average pollutant concentrations over the lockdown period 23/January-8/April/2020 are compared with 2019 (with one less day duration because 2020 was a leap year) for Wuhan and Beijing in Table 3. The table was assembled from daily averages, calculated from at least 20 hourly measurements at individual sites. Welch's t -test (i.e., unequal variances) was used to determine the significance of the difference between the years. The average NO₂ concentrations are overall much lower for 2020 (black) and we additionally note that differences in NO₂ between the two years were significant ($p_1 < .0001$) across all the nine individual sites (note: individual site data is not displayed in the table). Similarly, the PM_{2.5} concentrations show a striking difference between the two years with the t -test indicating $p_1 < .0001$. The PM₁₀ data is incomplete, but the raw data hints at decreases. However, we should be cautious in ascribing these differences to the lockdown as it might simply be that weather favoured more dispersion in 2020. Nevertheless, de-weathered data (Vu *et al.*, 2019; Cole *et al.*, 2020), suggest there were subtle though real changes as a result of lockdown. There is a possibility

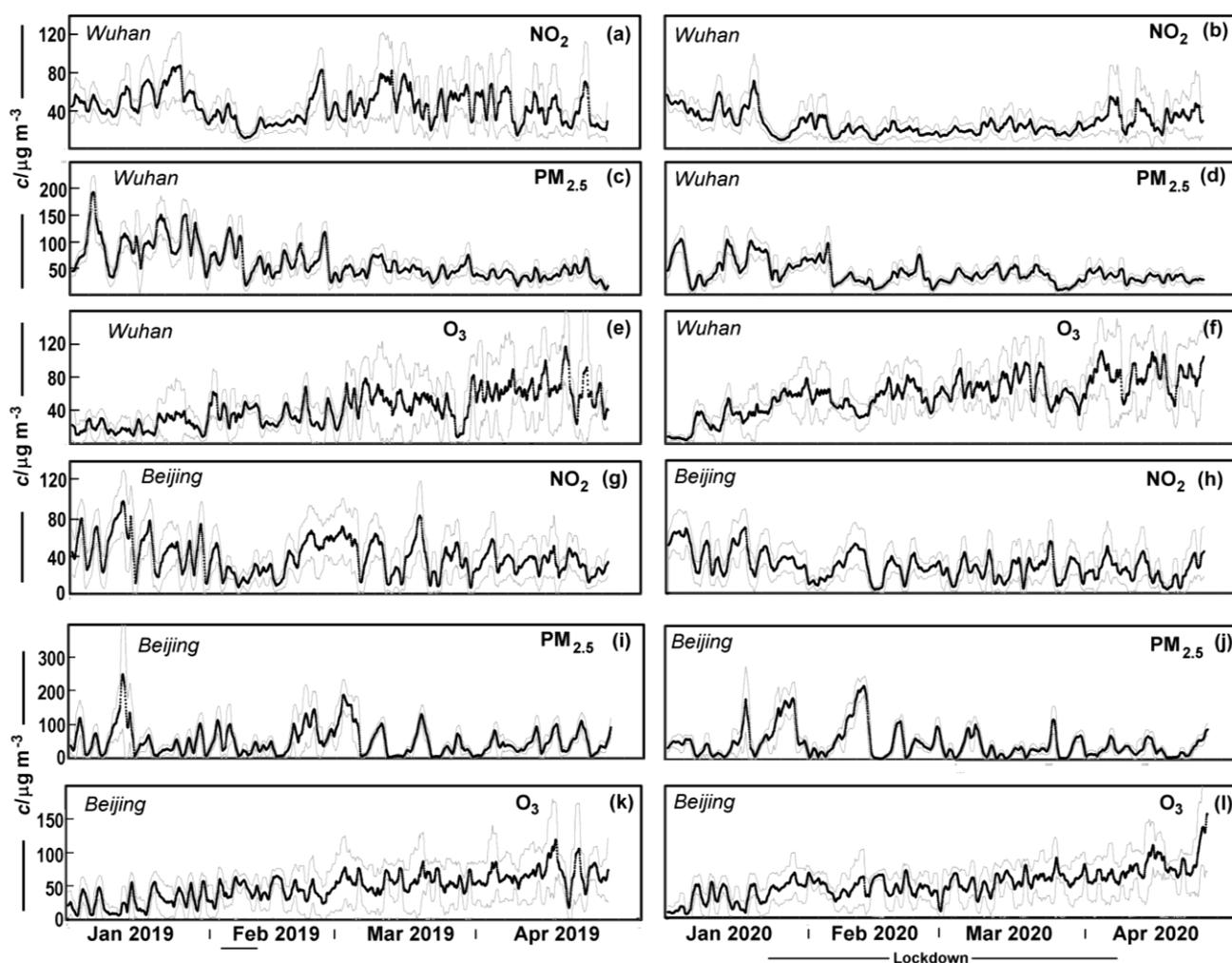


Fig. 6. Running means (25 point i.e., ~24 hours, to be representative of daily values) of hourly pollutant concentrations: January to April over all sites in Wuhan and urban and suburban sites in Beijing. The grey lines define the upper and lower bounds of the standard deviation. (a) NO_2 in Wuhan 2019, (b) NO_2 in Wuhan 2020, (c) $\text{PM}_{2.5}$ in Wuhan 2019, (d) $\text{PM}_{2.5}$ in Wuhan 2020, (e) O_3 in Wuhan 2019, (f) O_3 in Wuhan 2020, (g) NO_2 in Beijing 2019, (h) NO_2 in Beijing 2020, (i) $\text{PM}_{2.5}$ in Beijing 2019, (j) $\text{PM}_{2.5}$ in Beijing 2020, (k) O_3 in Beijing 2019 and (l) O_3 in Beijing 2020. Note: Chinese New Year of 2019 and the Wuhan lockdown of 2020 are marked as bars below the x -axis.

of increased secondary aerosol formation at night under lockdown (Huang *et al.*, 2020) and after sunrise secondary aerosol production has been observed after firework displays (Retama *et al.*, 2019). There also look to be significant decreases in CO (although these were not apparent across all the individual sites) and SO_2 . We noticed that the rural site near Chenhu Wetland shows higher concentrations in 2020 for both CO and SO_2 , although these are not really significant. Ozone as a secondary pollutant, that is often titrated out by NO in urban areas, might well show higher concentrations under lockdown when emissions are lower. This proves to be the case, with average O_3 concentrations higher at all sites in 2020 ($p_1 < .0001$), with the exception of the wetland site ($p_1 = 0.03$). Here there is little change in O_3 , as at this rural site, NO_x is likely to be low every year.

The picture is less clear for Beijing (Table 3), but the decreases in NO_2 and SO_2 are significant with a slight increase in O_3 of modest significance. The decrease in average NO_2

concentrations was more apparent at urban sites (within the 5th Ring Road); understandable as these were more likely to show an effect from COVID-19 restrictions. The average $\text{PM}_{2.5}$ concentrations at the urban sites and suburban sites (beyond the 5th Ring Road) are not especially distinctive; indeed at many sites the concentrations are slightly higher under the restrictions, although the individual differences were significant at two of the 23 sites ($p_1 \sim 0.05$ confidence level as from Welch's t-Test comparing average daily values between the two years). The concentrations of CO show some sites higher and some to be lower during the restrictions, but none with significant differences ($p_1 > .1$). Ozone concentrations appear higher at all but one of the urban site during the COVID-19 restrictions, although this was only significant ($p_1 < .1$) at only six of these sites.

In general, concentrations for primary pollutants tended to be higher in 2019, although the long secular decrease in emissions (and meteorological differences) might account

Table 3. Average pollutant concentrations (c) and standard deviations (σ) for 2019 and 2020 over the lockdown period extracted from daily average values across the cities. The p value refers to the significance of the difference using Welch's t -test. Note: - = record incomplete. $\mu\text{g m}^{-3}$, except CO as mg m^{-3} , Beijing is taken from the 23 urban and suburban sites.

Pollutant	2019		2020		t -test
	c	σ	c	σ	p
<i>Wuhan</i>					
NO ₂	45.2	17.8	21.2	6.8	< .0001
PM _{2.5}	58.5	26.5	38.0	16.8	< .0001
PM ₁₀	66.4	33.8	46.2	19.7	< .0002
CO	1.01	0.25	0.89	0.19	< .0005
SO ₂	8.8	3.0	7.9	2.5	< 0.03
O ₃	42.1	16.9	62.6	15.3	< .0001
<i>Beijing</i>					
NO ₂	37.2	17.6	26.3	11.7	< .001
PM _{2.5}	50.0	40.2	53.8	50.0	0.30
PM ₁₀	80.8	51.6	-	-	
CO	0.71	0.34	0.74	0.51	0.32
SO ₂	5.8	2.8	4.2	2.4	< .001
O ₃	50.0	14.5	54.2	12.9	0.03

for some of this. However, as previously noted the well-recognised change in NO₂ is clear, while the change in particulate concentrations is less significant. The changes in surface O₃ are probably regionally variant with an increase in concentrations found across China, especially in Northern regions (Huang *et al.*, 2020; Shi and Brasseur, 2020), along with striking increases in Beijing and Wuhan, but only modest changes in Shanghai and Guangzhou (Zhao *et al.*, 2020).

Weekly cycles can reveal the variations in social behaviour and are a good indicator of human effects on air pollutants. These are likely to be less sensitive to changes in climate between years. Diurnal cycles in Wuhan for 2019 and 2020 over the days of lockdown are seen in Fig. 7, which shows hourly averages (across all sites) separating weekdays and weekends (i.e., non-work days which include Saturdays, Sundays and public holidays). The NO₂ concentrations are obviously much lower in 2020 (Fig. 7(b)), and although the diurnal pattern is similar for each year, i.e., minimum in the early afternoon and maximum in the early evening, there are subtle differences. Weekday/weekends cycles are similar for 2019 with weekday and weekend means of 45.4 and 44.5 $\mu\text{g m}^{-3}$, a difference that is barely significant from a t -test ($p_1 = 0.04$) paired across sites. Under lockdown these are 21.2 and 19.2 $\mu\text{g m}^{-3}$, a significant difference ($p_1 < .0001$) that arises from higher values at night over the weekends. In 2019, weekends revealed slightly higher concentrations of PM_{2.5} throughout the day (56 and 64 $\mu\text{g m}^{-3}$; $p_1 < .0001$); perhaps people enjoying an evening out, while under lockdown these concentrations were typically lower during the weekends (39.0 and 34.5 $\mu\text{g m}^{-3}$; $p_1 < .0001$). The changes likely arose from the restriction of outdoor weekend activities likely to generate PM_{2.5}. Weekends under lockdown shows PM_{2.5} concentrations that are almost invariant through the day. The t -test, paired across sites, showed that the weekday and weekend concentrations of CO in 2019 were almost identical, with no significant differences in concentrations (1.011 and 1.014 mg m^{-3} ; $p_1 = 0.27$). Under lockdown in 2020, as with particles there is a slight, though significant decrease in the

concentrations during weekends (0.899 and 0.857 mg m^{-3} ; $p_1 < .0001$), which show less of a daily cycle.

Discussion

The effect of celebratory fireworks, the Chinese New Year holiday and the lockdown restrictions, as a response to the COVID-19 pandemic, all left signatures on the atmospheric environment. The effect of fireworks is evident in elevated particulate concentrations during brief periods of celebration, though increasing regulation has come to limit the effect and seems to have gradually pushed any peaks to the outer margins of Chinese cities (Lai and Brimblecombe, 2017, 2020). The holiday effect, while much discussed in the past, seems limited to the nitrogen oxides, and difficult to detect in Wuhan and Beijing. Contemporary social behaviour may allow emissions to remain relatively high even during the holidays and result in a decline in the magnitude of the holiday effect, which aligns with trends in the holiday effect observed elsewhere in the region (Kim *et al.*, 2020).

Bao and Zhang (2020) show broad improvements in SO₂, PM_{2.5}, PM₁₀, NO₂, and CO across 44 Chinese cities of the Jingjinji megalopolis, 1/January–21/March/2020. Although lockdown had an impact on air quality, it is necessary to recognise the subtlety of the changes. It has been popular in publications to assert a reduction of air pollutants in lockdown by comparing the period immediately before and after or alternatively comparing 2020 with 2019; indeed this paper adopts such methods, but the changing seasons and weather can make comparisons difficult. As Cole *et al.*, (2020) rightly argue, even after de-weathering the data: “Our results indicate that the impact of the lockdown varied by pollutant, a nuance that newspaper reports of cleaner post-lockdown air have generally failed to acknowledge.” They note that while PM₁₀ concentrations declined, this was for a shorter period. Our work agrees with this and suggests the case for improved CO and SO₂ is even less convincing. The effects in Beijing are necessarily subtle given the restrained response to COVID-19 in the capital, though these are

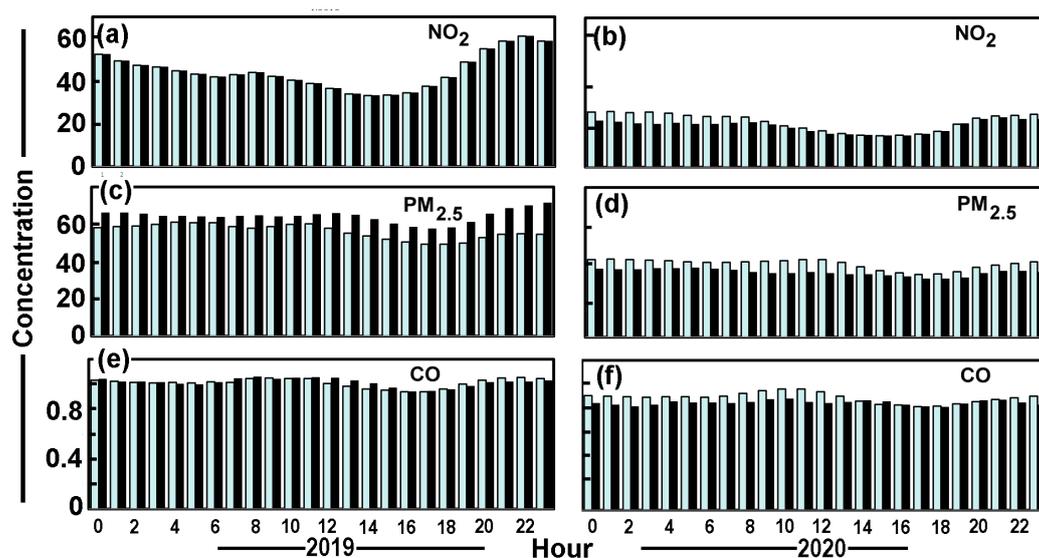


Fig. 7. Diurnal plot as hourly average concentrations from all sites in Wuhan for weekdays (light bars) and weekends (dark bars) across the period 23/Jan to 7/April for 2019 and 2020 (representing lockdown for the latter year) for (a) NO₂ in 2019, (b) NO₂ in 2020, (c) PM_{2.5} in 2019, (d) PM_{2.5} in 2020, (e) CO in 2019 and (f) CO in 2020. Note: concentrations as $\mu\text{g m}^{-3}$, for NO₂ in 2020 and PM_{2.5} while CO is as mg m^{-3} .

clearest for urban NO₂. Increases in ozone through a lack of titration by NO are apparent, but are clearer in urban Wuhan, while less significant in the surrounding countryside, where NO_x is inevitably lower. Such issues may well limit the spread of ozone increases more regionally (Xu *et al.*, 2020).

CONCLUSIONS

While peaks in PM_{2.5} in the early morning of New Year were once characteristic of Chinese cities, regulations have reduced their occurrence such that they only persist in less populated areas. The holiday effect is now rather indistinct, given the variation of activities even during holidays under COVID-19 period, while the fireworks effects in some places (i.e., rural areas) still can be observed although the firework peak at midnight on the eve Chinese New Year is smaller than in the past because of increasing regulatory restriction. During lockdown, Weibo users showed a concern that the air pollution, perhaps “temporarily lost, would be found again once it ended”, with social media thus revealing citizen concern about long term improvement to air quality. Views, expressed across much of the world, suggest we should use the pandemic as a catalyst for broad environmental improvement, much as John Evelyn saw London's Great Fire of 1666 as a chance to replan the city (<http://digitalarchive.mcmaster.ca/islandora/object/macrepo%3A31438>), remove pollutant sources within London, and place them downwind of the metropolis. Arguments that pollution exposure has been reduced by COVID-19 lockdown and actually saved lives may be more difficult to establish, because the improvements in air quality can be subtle, spatially limited and not always long-lived. As most people are indoors during the lockdown it is also necessary to consider indoor exposure in estimating health outcomes. This means that although ozone concentrations were enhanced in urban agglomerations under

lockdown (less apparent in the countryside), it is possible that ozone will not be strongly transferred indoors. Research has considered the primary rather than secondary aerosol, so this may represent a further area for study. While the first wave of the COVID-19 pandemic may have been short, it has given us much to think about the relation between human activity and our environment.

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