AIR QUALITY EXPERT GROUP

Annex to the AQEG report - submissions cited in the published work: Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK.

Rapid evidence review – June 2020.

Prepared for:

Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government;
and Department of Agriculture, Environment and Rural Affairs in Northern Ireland

MAYOR OF LONDON

Estimation of changes in air pollution in London during the COVID-19 outbreak

Response to the UK Government's Air Quality Expert Group call for evidence

April 2020

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Introduction

On 4 April 2020 the UK Government's Air Quality Expert Group (AQEG), acting on a request from the Department of Environment, Food and Rural Affairs, called for evidence to address a set of urgent short-term questions related to recent and ongoing changes in UK air quality. For more information please see: https://uk-air.defra.gov.uk/news?view=259.

The following document is London's response to this call for evidence. It will aim to address two of the key questions identified AQEG:

- What sectors or areas of socioeconomic activity do you anticipate will show a decrease in air pollution emissions, and by how much? Are there any emissions sources or sectors which might be anticipated to lead to an increase in emissions in the next three months?
- Can you provide estimates for how emissions and ambient concentrations of NOx, NO_2, PM, O_3, VOC, NH_3 etc may have changed since the COVID outbreak? Where possible please provide data sets to support your response.

This report covers the period to from 1 January 20 April 2020. As requested by AQEG the evidence has been kept brief, with additional context and data provided in the Appendix.

This evidence has been published alongside the Central London ULEZ Ten Month Report which outlines the improvements in air pollution in London in the period preceding the COVID-19 outbreak.

It is important that the change in air pollution concentrations as a result of COVID-19 measures are framed in the context of London's normal seasonal pattern for pollutants and the substantial improvements in London's air quality in recent years, in particular in central London where the Ultra Low Emission Zone has already significantly reduced concentrations of pollutants, as demonstrated in Figure 1.

Figure 1 shows the change in hourly average NO_2 at all sites in central London, from the period January – April. The red line shows the hourly trends in NO_2 in central London from 1 January 2017 – 20 April 2017 (before changes associated with the central ULEZ took full effect). The green line shows the hourly trends in NO_2 in central London from 1 January – 15 March 2020, with the ULEZ in place. The blue line shows the hourly trends in NO_2 in central London from 16 March – 20 April 2020, with COVID-19 measures in place.

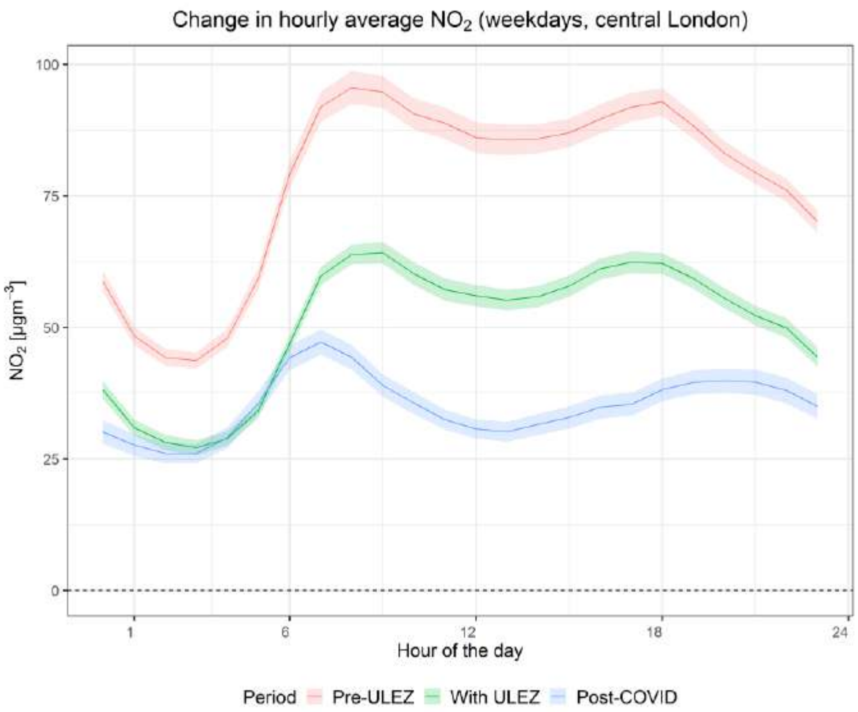

Figure 1. Change in hourly average NO_2 in central London

In 2020, before measures to address the COVID outbreak were introduced, hourly average NO_2 at all sites in central London had already reduced by over one third (35 per cent) compared to the same period in 2017. Since 16 March 2020 there has been an additional reduction of 26 per cent. As will be shown later in this report the reduction is even higher at roadside sites.

In recent years policies and measures have been introduced in London (including Low Emission Bus Zones, the ULEZ and changes to the taxi fleet) that have resulted in significant improvements in air quality. Other studies have compared air quality in the post-COVID period to the same period for previous years. Whilst this may be appropriate for other locations, it is not appropriate for London due to the significant recent improvements which pre-date the COVID outbreak. This analysis instead compares the periods 1 January - 15 March 2020 and 16 March 2020 - 20 April 2020.

Changes in emissions

COVID-19 is likely to impact the majority of emissions sources in London including road transport, aviation, construction, domestic and commercial heating and commercial cooking. A breakdown of emissions sources in London is provided in Figure A 1 - Figure A 6 in the Appendix. Please note, these only account for emission sources within London. As evidenced in the next chapter, transboundary sources (over which London has no control) appear to have been less impacted by stricter COVID-19 measures. This includes emissions from agriculture. Particulates derived from ammonia are the single largest contribution to imported background pollution in London. Agriculture is the dominant source of ammonia emissions in the UK, accounting for around 87 per cent of all emissions. Unlike most other air pollutants, emissions of ammonia have been rising since 2013.

Data is not yet available for many sectors, with the exception of transport, for which Transport for London has good data. Other major emissions sources which are likely be significantly reduced are construction, commercial cooking and commercial heating.

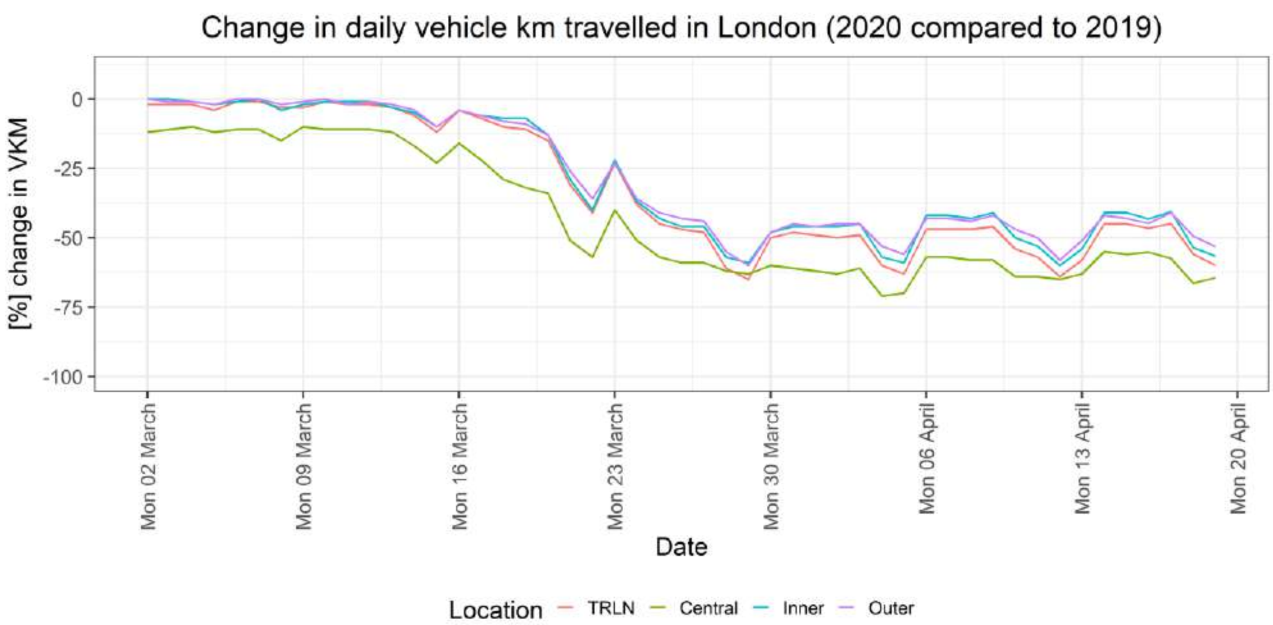

Figure 2. Change in daily vehicle km travelled in London (TfL, 2020)

Road transport accounts for around half of London's NOx emissions and a third of PM emissions. Since the beginning of March road traffic in London has reduced by around 50 per cent Londonwide. Figure 2 shows the percentage reduction in vehicle kilometres travelled in 2020 compared to the comparable day in 2019. Please note, the central London ULEZ, which was introduced in April 2019, had already reduced traffic in the central zone by approximately 10 per cent.

Departure from usual travel behaviour began around Monday 16 March, when the UK Government strongly recommended social distancing and home working where possible.

Changes in concentrations

The following analysis uses data from London's automatic air quality monitoring stations (which are also used for statutory reporting) to assess the changes in concentrations of nitrogen dioxide (NO_2), NOx, ozone (O_3) and particulate matter ($PM_{2.5}$, PM_{10}) since Monday 16 March 2020. At this early stage the changes in air pollution (both positive and negative) reported here cannot be attributed solely to the COVID pandemic because the period before and after were subject to different meteorological conditions, which have not been corrected for in this analysis.

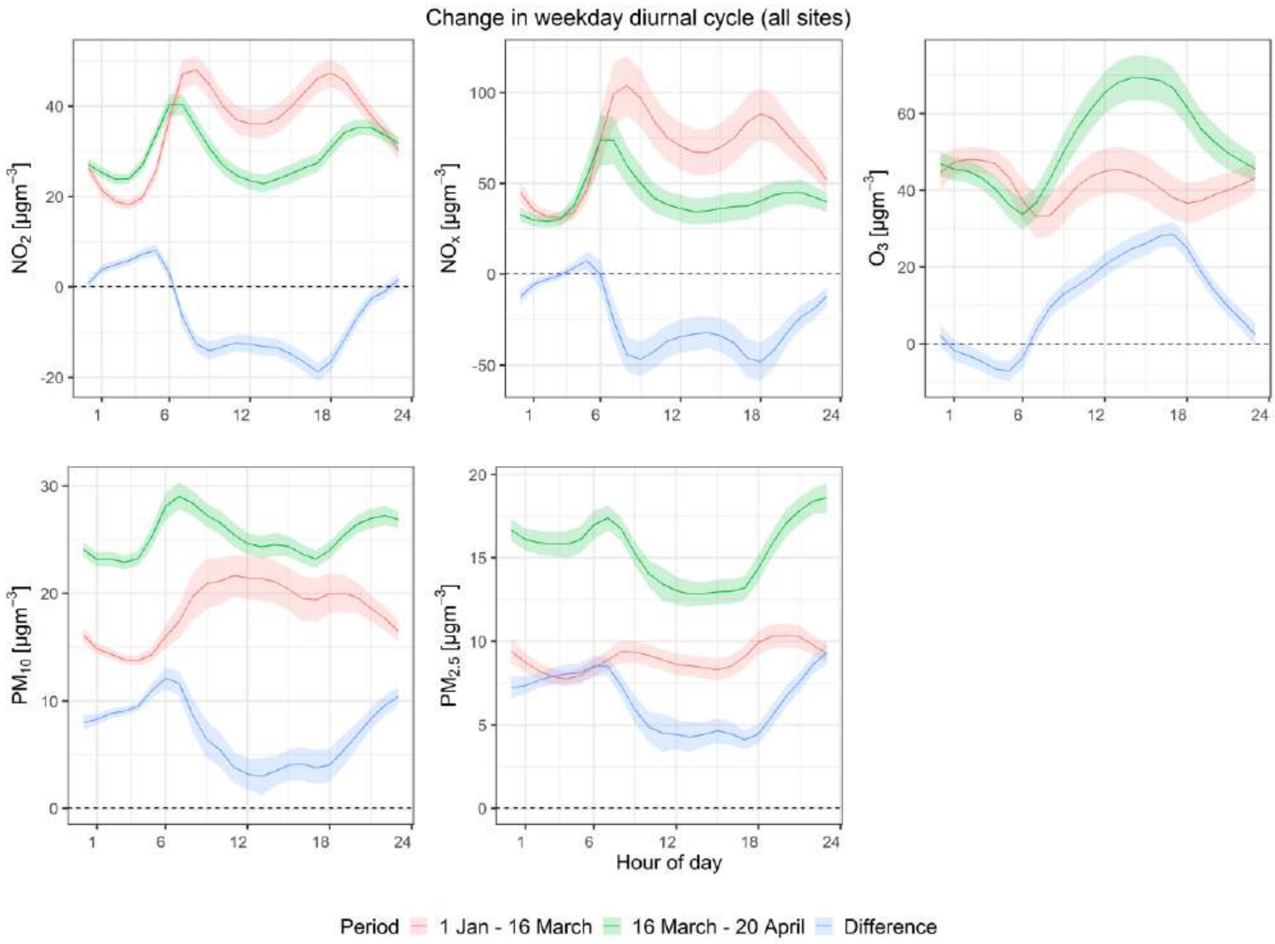

Figure 3. Change in diurnal cycle of pollutants since 16 March 2020

Figure 3 shows the change in hourly average NO_2, NO_x, O_3, PM_{10} and $PM_{2.5}$ in the period since 16 March, compared to the period 1 January to 16 March. The key findings are there have been overall reductions in NO_2 and NO_x, and increases in O_3, PM_{10} and $PM_{2.5}$. In addition, London has had a number of particulate pollution episodes since 16 March. This highlights that London's poor air quality is not solely related to road transport. To improve London's air quality further action is required on other sources, including domestic burning and agricultural emissions. The following sections provide more detailed analysis by pollutant.

Nitrogen dioxide (NO_2)

There has been a significant reduction in NO_2 since 16 March, these changes are in addition to the reductions already delivered by the central London ULEZ and other policies. The greatest reductions have been measured at kerbside and roadside sites in central and inner London. Daily average NO_2 has reduced by around 40 per cent at roadside sites in central London, and 20 per cent elsewhere. This is despite a slight increase in NO_2 measured at regional background sites outside of London. NO_2 has significantly reduced at some of London's busiest locations. At Oxford Street daily average NO_2 has reduced by 23 ugm^{-3}, a reduction of 47 per cent. Similarly, Marylebone Road has reported a reduction of 26 ugm^{-3}, a reduction of 48 per cent.

The reduction in NO_2 has not been uniform throughout the day. Figure 3 shows changes in daily, hourly and monthly NO_2 at roadside sites in inner London before and after the 16 March. Since 16 March there has been an increase in hourly average NO_2 between the hours of around 00:00 – 05:00, followed by a decrease during the day.

Statistical analysis can be used to identify the proportion of NO_2 at roadside sites which is directly attributable to traffic, removing the impact of changes in background concentrations. This is known as the roadside increment. The roadside increment of NO_2 has reduced by around 52 per cent in central London and 25 per cent in the rest of London.

For more data on changes in concentrations of NO_2 please see Appendix 3.

NOx

There have been even larger reductions in NOx concentrations. As with NO_2 the greatest reduction has been at kerbside and roadside sites. Kerbside sites in inner London have measured a 71 µgm^3 reduction in daily average NOx, a reduction of 47 per cent. Roadside sites in central London have measured a reduction of 62 µgm^{3}, a reduction of 56 per cent.

For more data on changes in concentrations of NO_x please see Appendix 4.

Ozone (O_3)

Daily average O_3 has increased at all 23 sites included in this analysis. In the period since 16 March there has been a 6 µgm^3 (11 per cent) increase in daily average O_3. This is not unusual for this time of year (see Appendix 5). However, the increase at many London sites far exceeds the increase in regional background. For example, at Marylebone Road (the only O_3 kerbside monitoring station included in this analysis) daily average O_3 increased by 24 µgm^3 (119 per cent). Other roadside sites and background sites in inner and central London measured increased in daily average O_3 of between 30 – 50 per cent. This indicates the increase in O_3 may also be being driven by the reduction in NOx emissions.

The World Health Organization guideline limit for O_3 is an 8-hour mean of 100 µgm^3. Since 16 March 9 sites in London have recorded an 8-hour mean over the WHO recommended limit. The EU legal air quality limit value for ozone is 120 µgm^3 over an 8-hour mean and no site in London exceeded this during the period.

For more data on changes in concentrations of O_3 please see Appendix 5.

Fine particulate matter ($PM_{2.5}$)

In the period since 16 March there have been a number of moderate particulate matter episodes, resulting in a 69 per cent (4.6 µgm^{-3}) increase in daily average $PM_{2.5}$ at regional background sites outside of London. This is not unusual for this time of year (see Appendix 6). Spring time is often the worst time of the year for particulate pollution in London, spring time episodes are associated with agriculture emissions which can travel long distances.

All site types within London measured an increase in daily average $PM_{2.5}$ since the 16 March of between 1 – 3 µgm^3 (14 – 43 per cent). However, the relative increase at sites in London are significantly less than for the regional background sites. This indicates there has been a reduction in the London local contribution to $PM_{2.5}$, and this is countering some of the regional increase. The reduction in local contribution is likely to be a result of a decrease in local emissions from transport, construction and (in central London) commercial cooking. However, King’s College London have stated concentrations may have been influenced by an increase in domestic garden and wood burning within London during the lockdown period.

It is possible to estimate the reduction in London's local contribution to $PM_{2.5}$ by assuming changes measured at the regional background sites represent the true change in background (4.6 μgm^{-3}). The difference between the change measured at the London sites and the change at a regional level provides an estimate for the reduction in London local contribution. The reduction in daily average local contribution varied by site type and locations, with an average of 2 μgm^{-3} across all sites which would represent an approximate 10 per cent reduction. The estimated reduction in the local contribution at some roadside sites, for example Euston Road, was over 3 μgm^{-3}, equating to a reduction of over 20 per cent.

The World Health Organization guideline limit for $PM_{2.5}$ is a 24-hour mean of 25 μgm^{3}. Since 16 March nearly all sites in London (and regional background sites) have recorded a daily mean over the WHO recommended limit.

For more data on changes in concentrations of $PM_{2.5}$ please see Appendix 6.

Particulate matter (PM_{10})

Similarly, there was a 74 per cent (8 μgm^{-3}) increase in daily average PM_{10} at regional background sites outside of London in the period since 16 March. Again, the daily average increase at sites within London was significantly lower than this, central London roadside and kerbside sites reported no change in daily average PM_{10} and industrial sites in inner London reported a small decrease. This indicates that there has been a significant reduction in London sources of PM_{10}. As is the case for $PM_{2.5}$ the reduction in local contribution is likely to be a result of a decrease in local emissions from transport, construction and (in central London) commercial cooking.

For more data on changes in concentrations of PM_{10} please see Appendix 7.

Appendix 1. Methodology

All air quality data analysis was performed using the open source statistical software R.

The period of comparison in this analysis is 1 January 2020 to 15 March 2020 and 16 March 2020 to 19 April 2020. The 16 March has been chosen as the split because this is when the UK Government recommended social distancing and working from home where possible and also when Transport for London report a departure from usual travel behaviour (see Figure 2).

London has an established weekly pattern for pollutants. Therefore, reductions in this analysis are calculated using comparable weekdays only. For example, the average reduction on Friday 20 March was calculated by averaging all Fridays between Monday 1 January and Monday 16 March and then subtracting the daily average for Friday 20 March.

Comparison to regional background

Both the period before, and the period after COVID-19 measures were introduced are subject to natural variability, complicated by the fact the spring is often the worst time of the year for many pollutants in London. Changes at rural (regional background) sites outside of London have been used to apportion between natural variability and impact of COVID-19 measures. The regional sites used for this are:

- Lullington Heath (LH), AURN
- Rochester Stoke (ROCH), AURN
- Chilbolton Observatory (CHBO), AURN

Appendix 2. Emissions sources in London

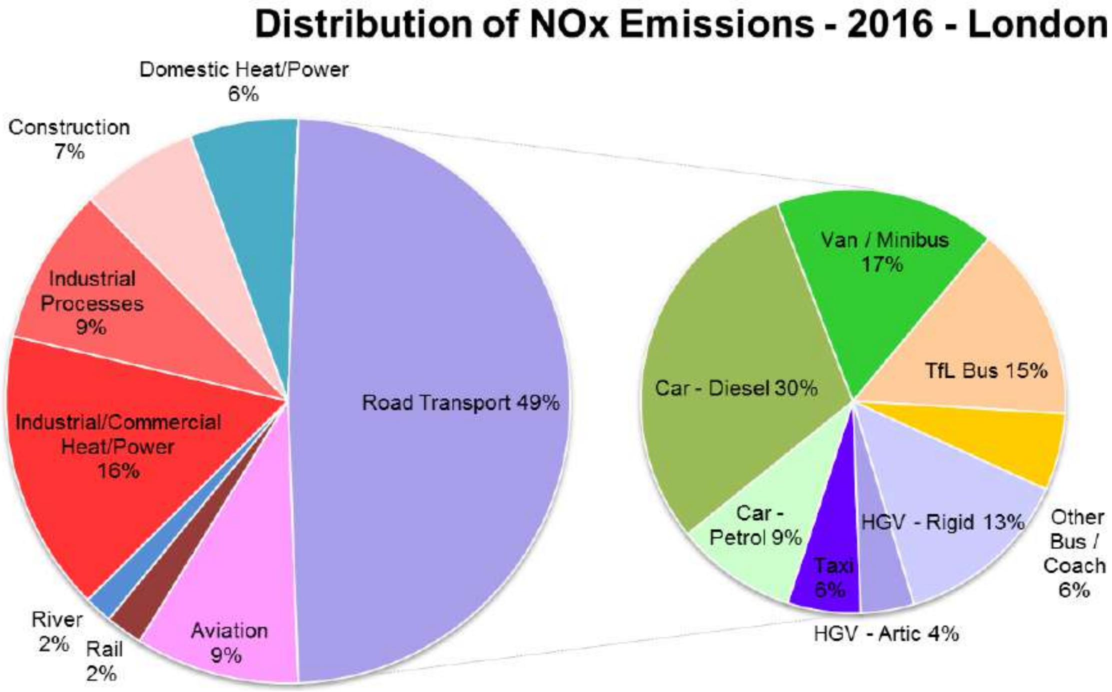

Figure A 1. Source apportionment of NOx emissions in London (LAEI 2016)

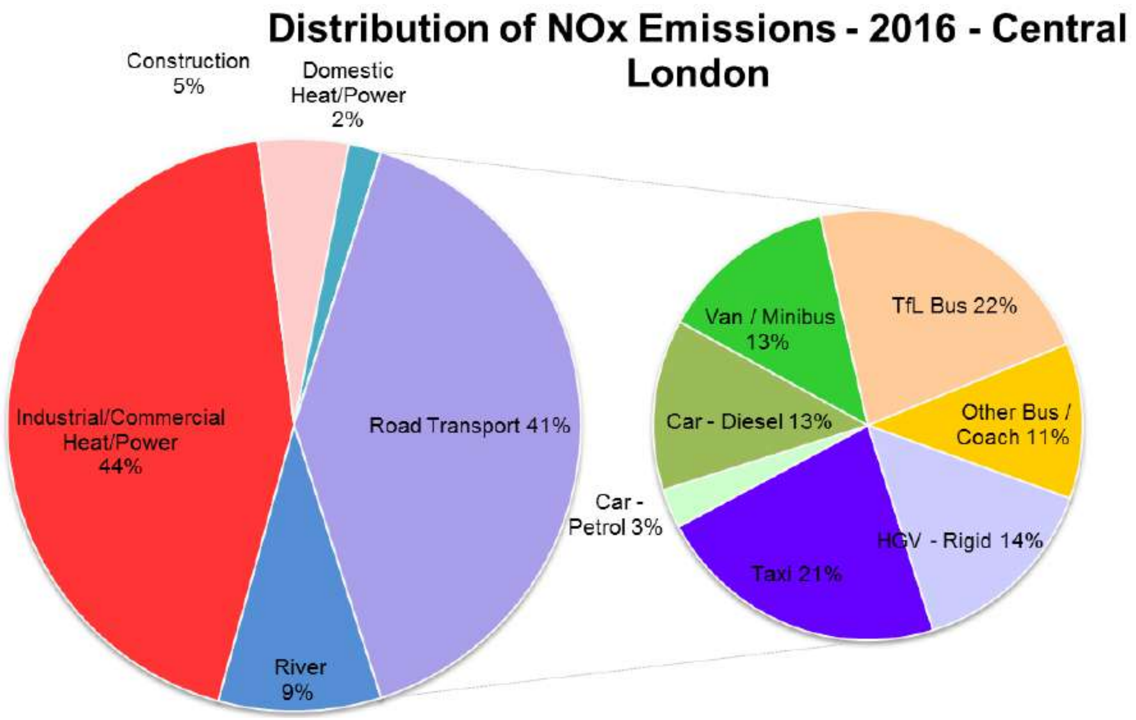

Figure A 2. Source apportionment of NOx emissions in Central London (LAEI 2016)

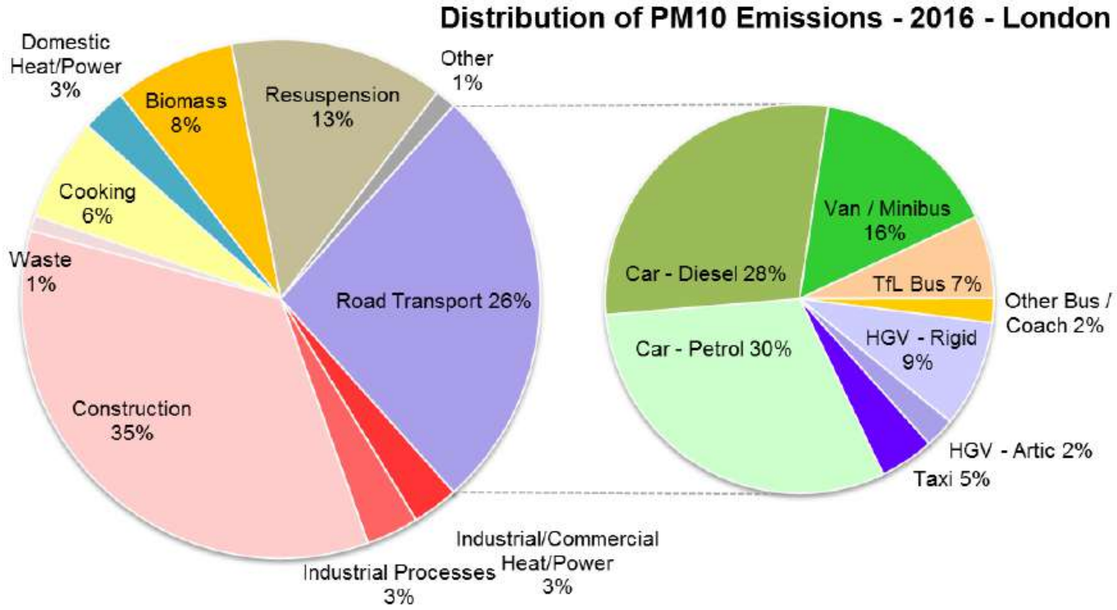

Figure A 3. Source apportionment of PM_{10} emissions in London (LAEI 2016)

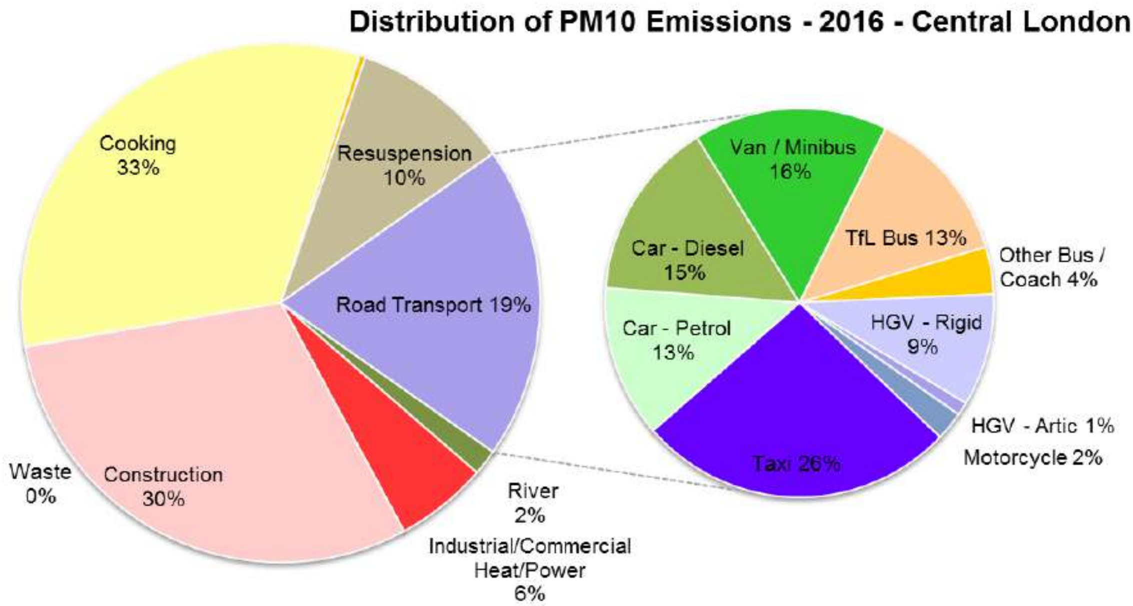

Figure A 4. Source apportionment of PM_{10} emissions in central London (LAEI 2016)

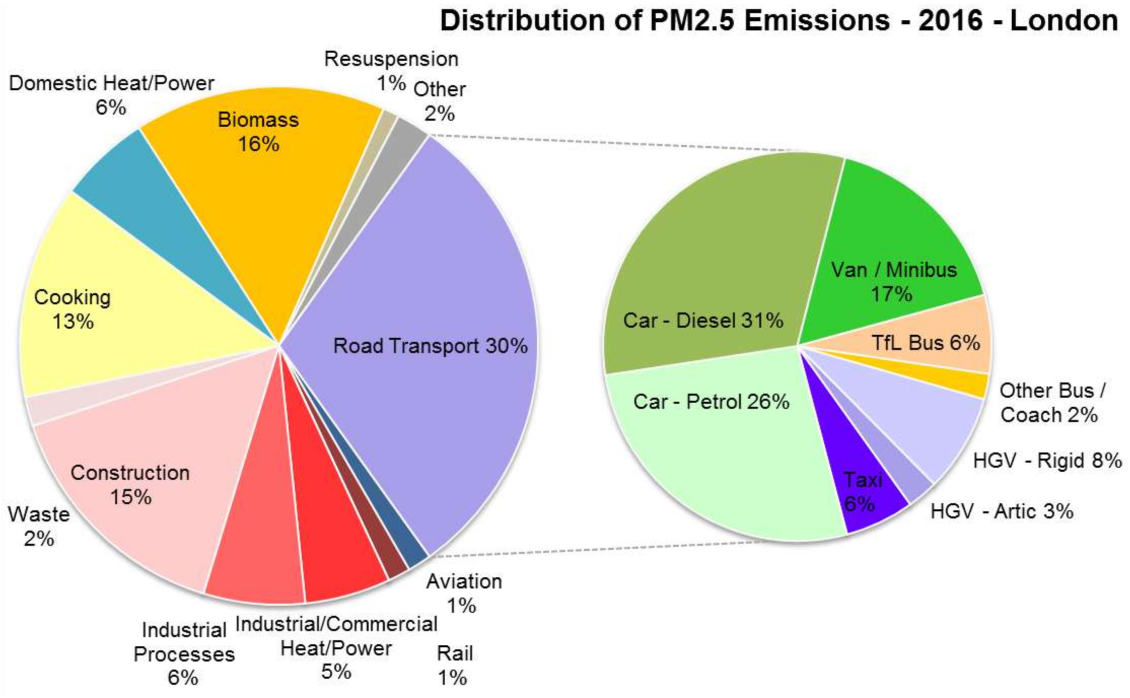

Figure A 5. Source apportionment of $PM_{2.5}$ emission in London (LAEI 2016)

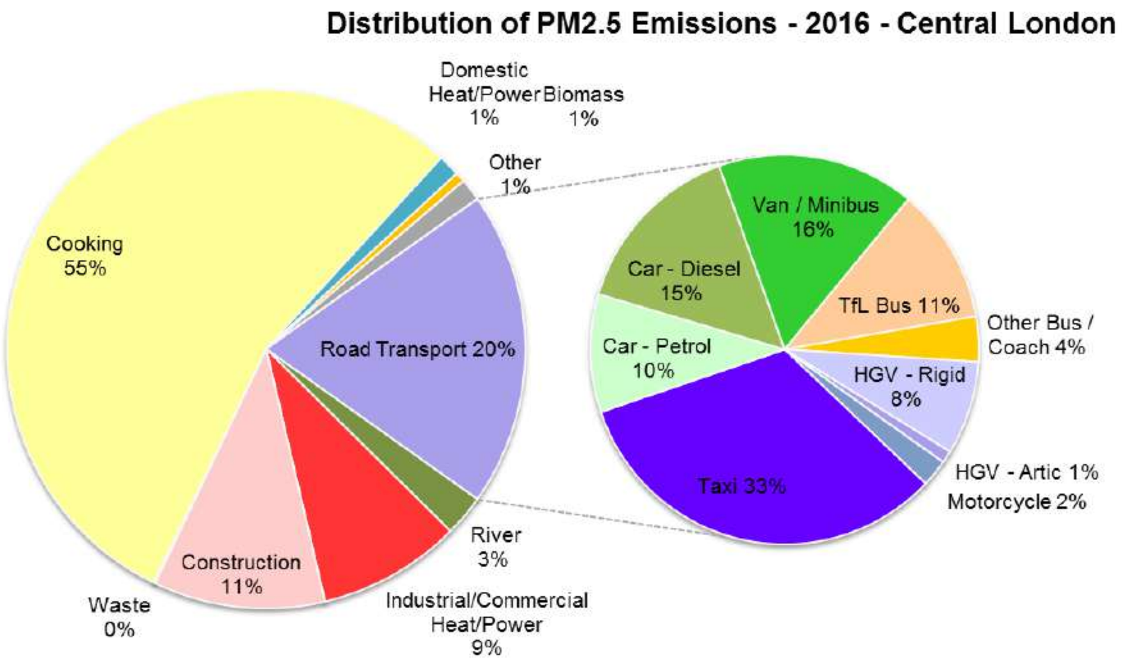

Figure A 6. Source apportionment of $PM_{2.5}$ emissions in central London (LAEI 2016)

Appendix 3. NO_2 data

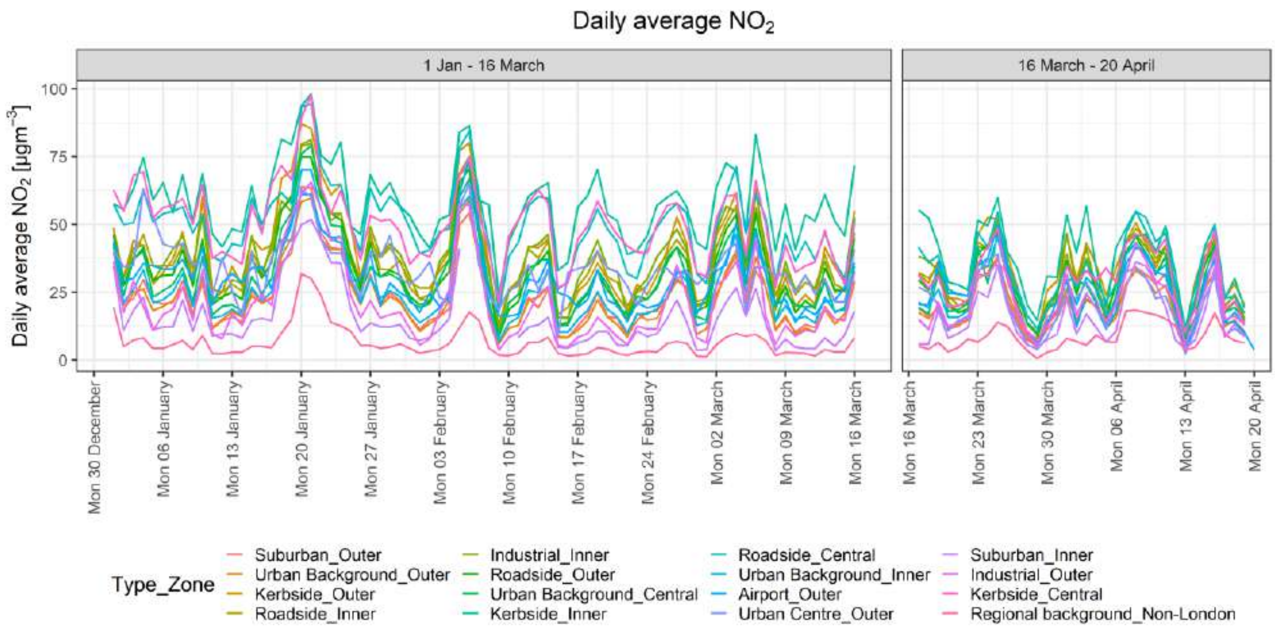

Figure A 7. Daily average NO_2 in London [2020]

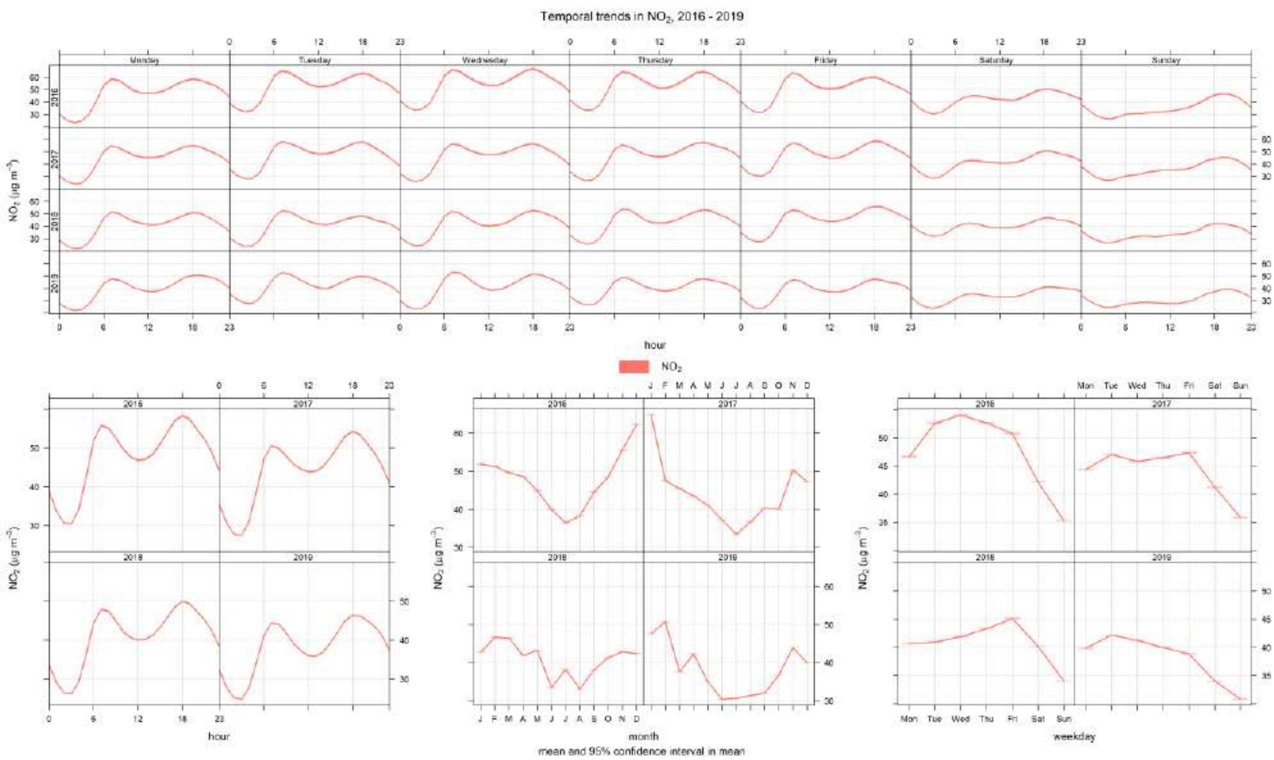

Figure A 8. Temporal trends in NO_2 in London [2020]

Table A 1. Change in daily average NO_2 since 16 March, grouped by site type and location

Type, Location	Change in daily average [μgm^{-3}]	Change in daily average [%]	Number of sites
Kerbside, Central	-23.5	-47%	1
Roadside, Central	-19.3	-38%	6
Kerbside, Inner	-17.1	-30%	4
Urban Centre, Outer	-12.7	-38%	1
Industrial, Inner	-8.6	-23%	3
Roadside, Inner	-7.8	-18%	30
Urban Background, Central	-6.5	-20%	4
Kerbside, Outer	-6.4	-15%	5
Airport, Outer	-6.0	-18%	2
Roadside, Outer	-6.0	-17%	22
Suburban, Outer	-4.4	-16%	7
Urban Background, Outer	-4.1	-18%	10
Urban Background, Inner	-3.2	-14%	15
Suburban, Inner	-1.6	-11%	1
Industrial, Outer	-1.3	-2%	2
Regional background, Non-London	0.0	+31%	3

Appendix 4. NO_x data

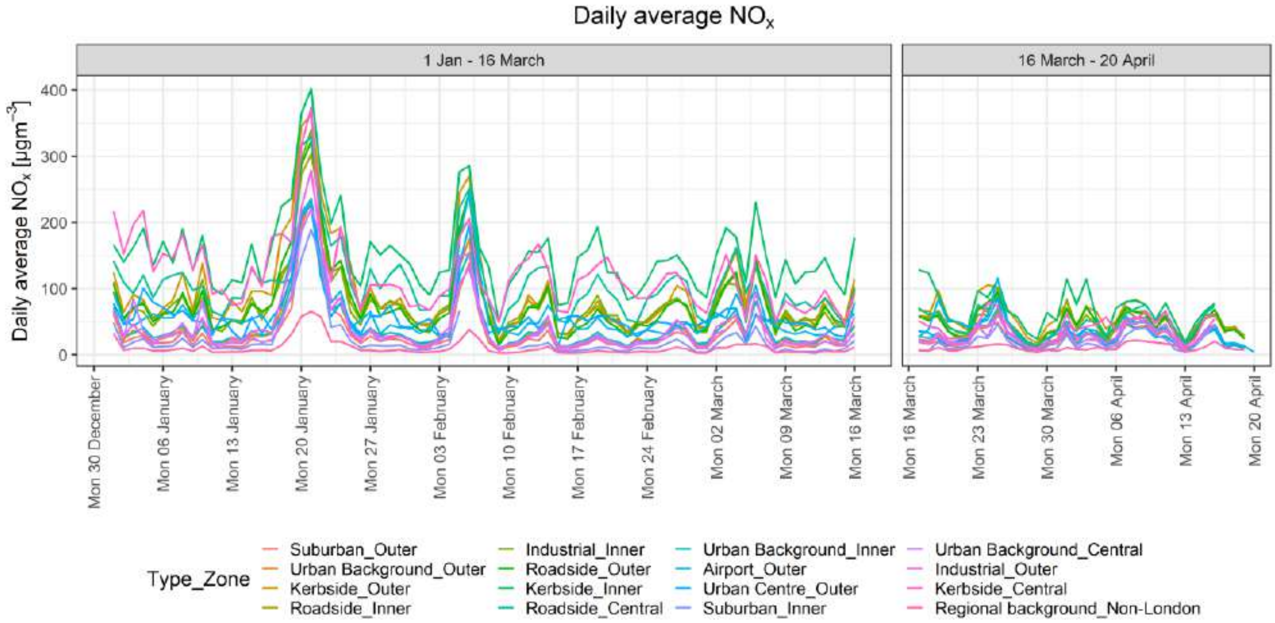

Figure A 9. Daily average NO_x in London [2020]

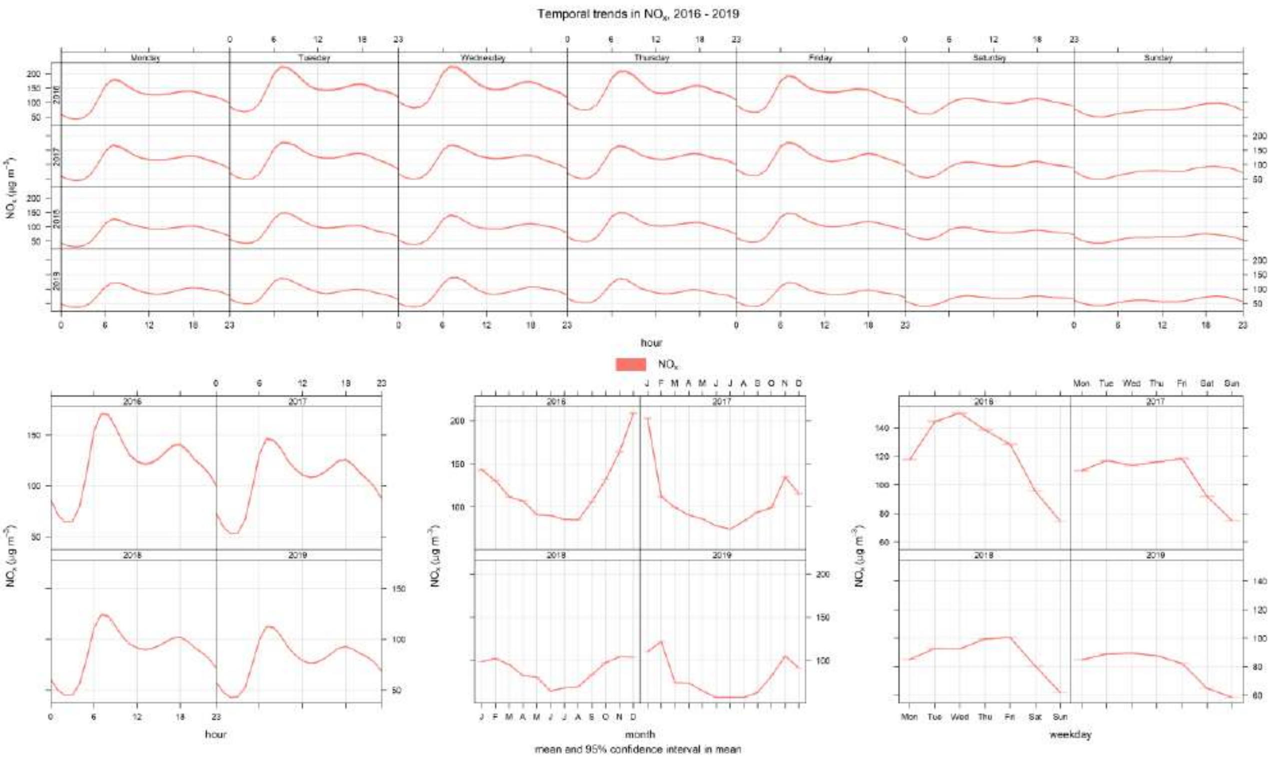

Figure A 10. Temporal trends in NO_x in London [2016 – 2019]

Table A 2. Change in daily average NO_x since 16 March, grouped by site type and location

Type, Location	Change in daily average [μgm^{-3}]	Change in daily average [%]	Number of sites
Kerbside, Central	-85	-69%	1
Kerbside, Inner	-71	-47%	4
Roadside, Central	-62	-56%	6
Kerbside, Outer	-33	-32%	4
Industrial, Inner	-32	-40%	3
Roadside, Inner	-30	-35%	29
Roadside, Outer	-29	-34%	21
Urban Centre, Outer	-28	-48%	1
Airport, Outer	-20	-32%	2
Urban Background, Central	-15	-37%	1
Urban Background, Outer	-14	-33%	10
Industrial, Outer	-14	-27%	2
Urban Background, Inner	-13	-31%	14
Suburban, Outer	-11	-32%	6
Suburban, Inner	-7	-31%	1
Regional background, Non-London	-1	8%	4

Appendix 5. O_3 data

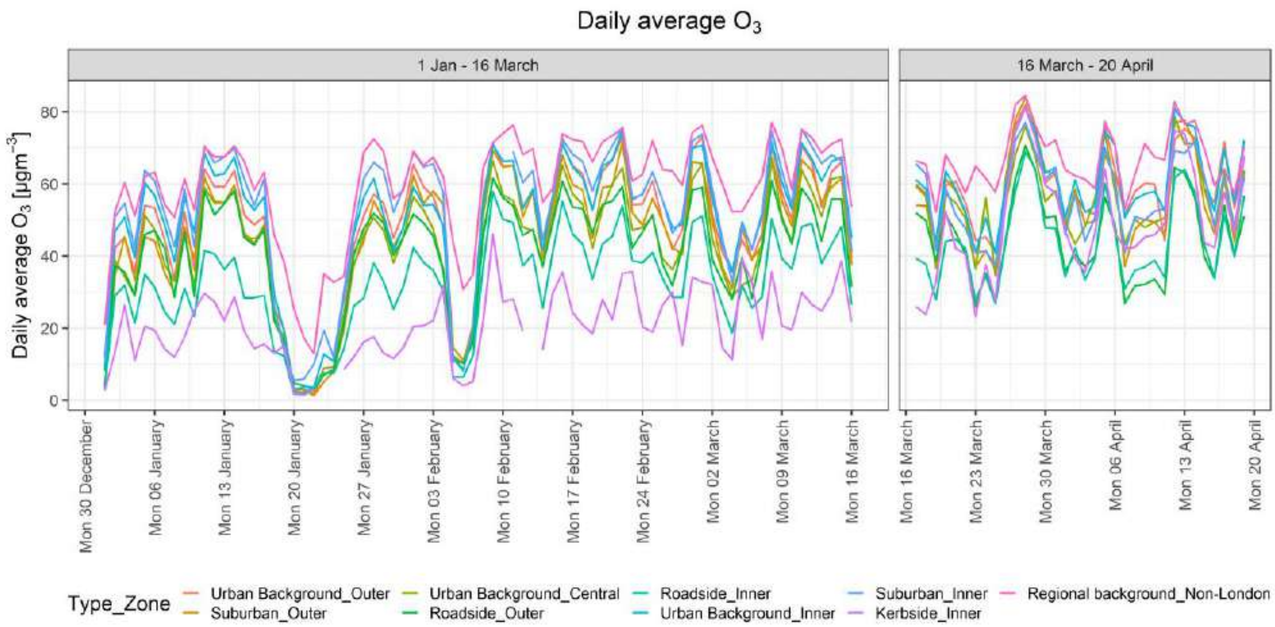

Figure A 11. Daily average O_3 in London [2020]

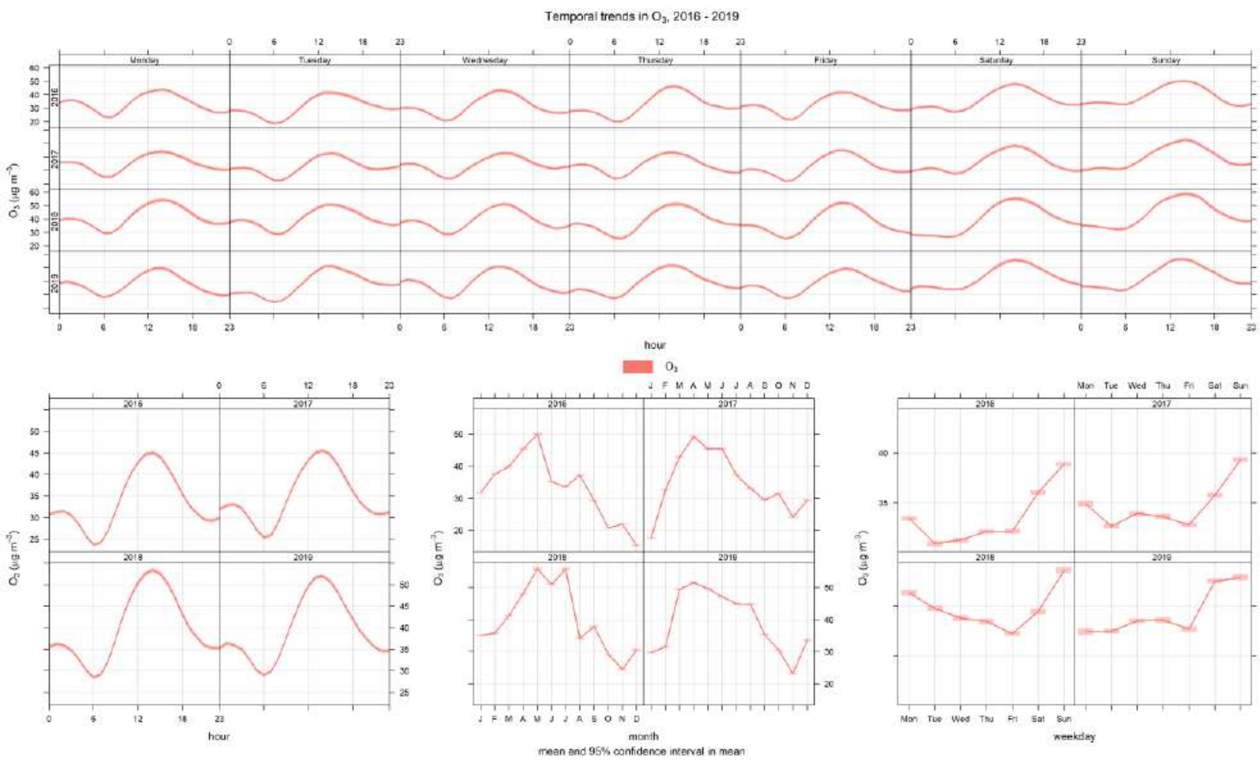

Figure A 12. Temporal trends in O_3 in London [2016 – 2019]

Table A 3. Change in daily average O_3 since 16 March, grouped by site type and location

Type, Location	Change in daily average [μgm^{-3}]	Change in daily average [%]	Number of sites
Kerbside, Inner	+24	+119%	1
Suburban, Outer	+11	+27%	3
Urban Background, Central	+10	+26%	2
Urban Background, Outer	+9	+20%	2
Roadside, Inner	+9	+32%	5
Urban Background, Inner	+8	+17%	4
Regional background, Non-London	+6	+11%	3
Roadside, Outer	+4	+12%	2
Suburban, Inner	+1	+3%	1

Appendix 6. $PM_{2.5}$ data

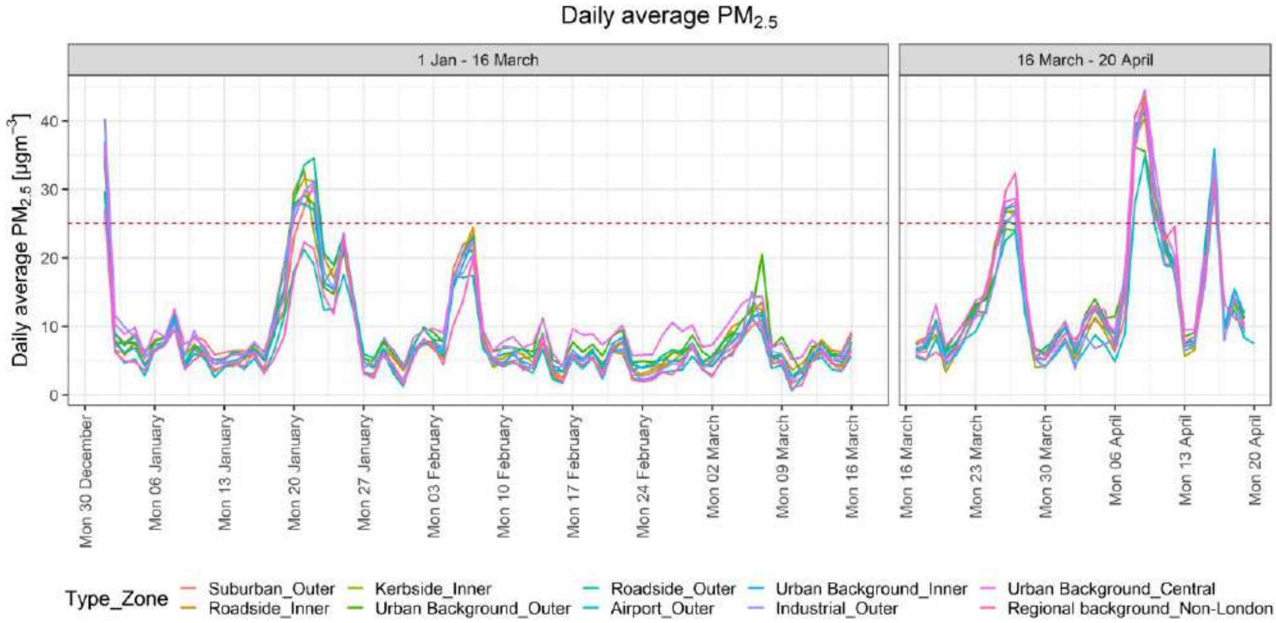

Figure A 13. Daily average $PM_{2.5}$ in London [2020]

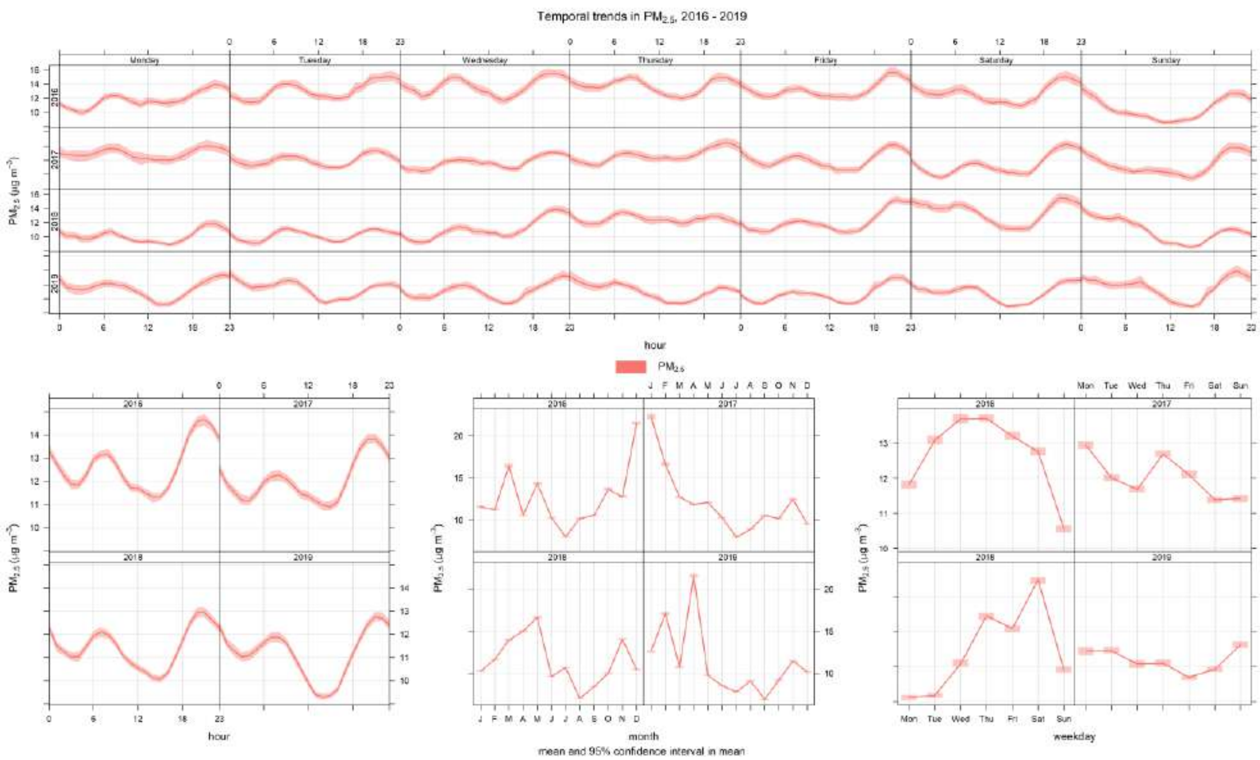

Figure A 14. Temporal trends in $PM_{2.5}$ in London [2016 – 2019]

Table A 4. Change in daily average $PM_{2.5}$ since 16 March, grouped by site type and location

Type, Location	Change in daily average [μgm^{-3}]	Change in daily average [%]	Number of sites
Regional background, Non-London	+4.6	+69%	2
Suburban, Outer	+3.3	+43%	3
Urban Background, Inner	+2.7	+37%	6
Roadside, Inner	+2.6	+31%	7
Roadside, Outer	+2.4	+30%	5
Airport, Outer	+2.3	+36%	2
Urban Background, Central	+2.3	+25%	1
Urban Background, Outer	+2.0	+22%	2
Kerbside, Inner	+1.6	+18%	2
Industrial, Outer	+1.2	+14%	1

Appendix 7. PM_{10} data

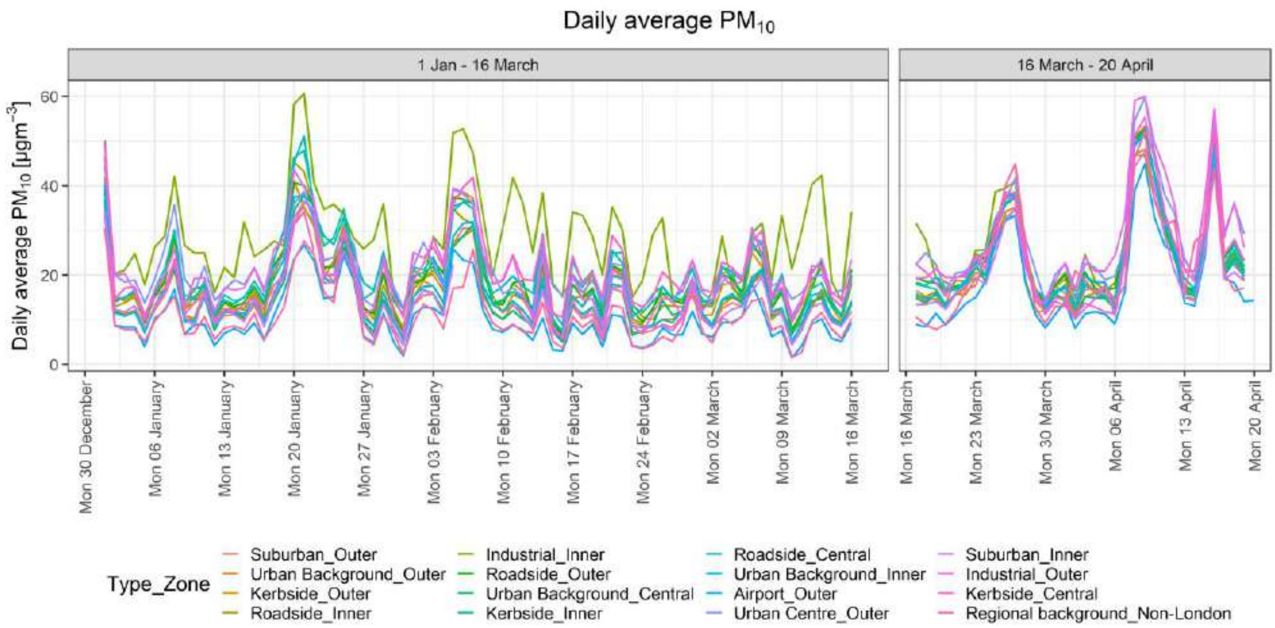

Figure A 15. Daily average PM_{10} in London [2020]

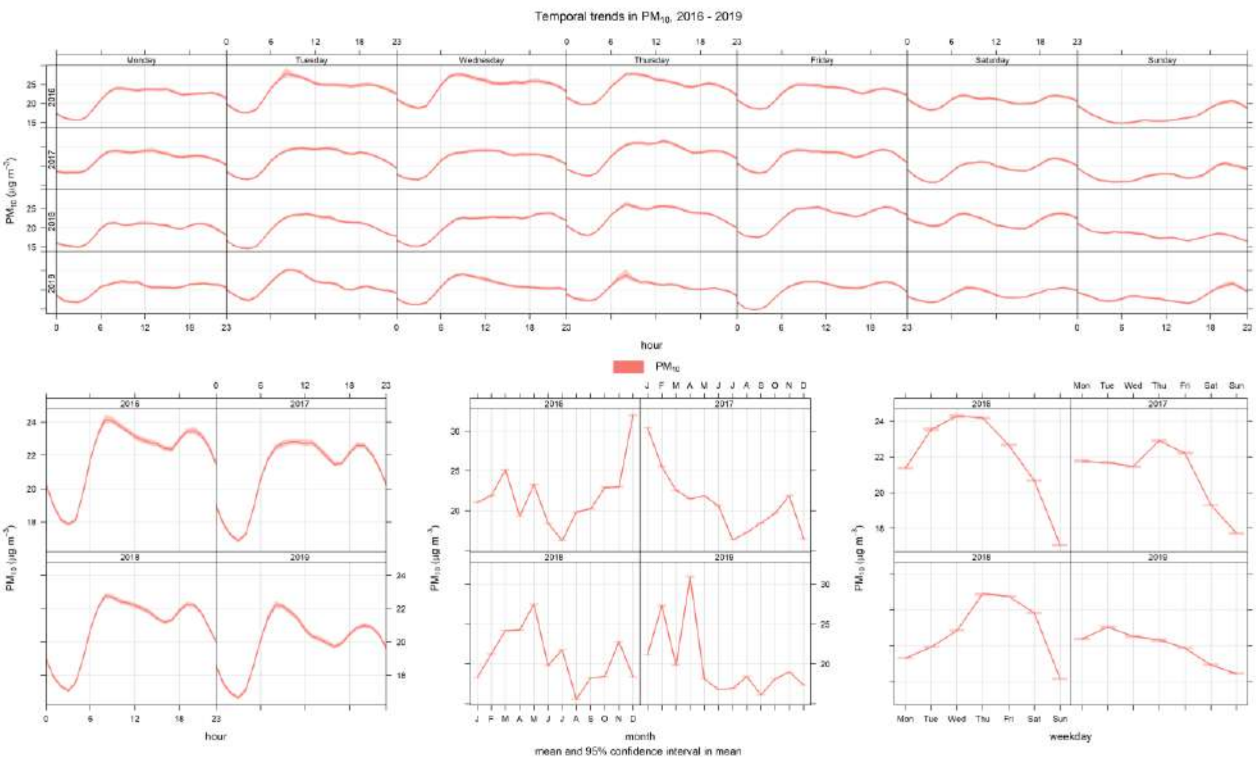

Figure A 16. Temporal trends in PM_{10} in London [2016 – 2019]

Table A 5. Change in daily average $PM_{2.5}$ since 16 March, grouped by site type and location

Type, Location	Change in daily average [μgm^{-3}]	Change in daily average [%]	Number of sites
Regional background, Non-London	+7.6	+74%	2
Suburban, Outer	+5.5	+40%	6
Urban Background, Inner	+4.3	+30%	10
Airport, Outer	+4.2	+42%	2
Urban Background, Central	+3.8	+27%	3
Suburban, Inner	+3.5	+28%	1
Roadside, Outer	+2.8	+17%	16
Urban Background, Outer	+2.8	+21%	9
Roadside, Inner	+2.8	+18%	22
Industrial, Outer	+2.6	+14%	1
Urban Centre, Outer	+2.2	+11%	1
Kerbside, Outer	+0.8	+6%	4
Roadside, Central	+0.0	+1%	4
Kerbside, Central	0.0	+1%	1
Kerbside, Inner	-0.2	0%	4
Industrial, Inner	-3.8	-4%	5

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MAYOR OF LONDON

Rt Hon George Eustice MP
Secretary of State for Environment, Food and Rural Affairs
Department for Environment, Food and Rural Affairs
Seacole Building
2 Marsham Street
London SW1P 4DF

Date: 23 April 2020

Dear George,

I am writing to share the Greater London Authority's analysis of the current changes in London's air quality due to the lockdown, which I am submitting as part of the call for evidence by the UK Government's Air Quality Expert Group.

As you know, poor air quality continues to stunt the growth of children's lungs and worsens chronic illness, such as asthma, lung, and heart disease. There is also emerging evidence linking air pollution with an increased vulnerability to the most severe impacts of COVID-19, given that it is a respiratory disease. Harvard University has published research showing a small increase in air pollution is linked to a 15 per cent increase in the COVID-19 death rate. Similarly, here in the UK, the University of Cambridge has found links between levels of air pollution and the severity of COVID-19.

The current lockdown has had the effect of dramatically improving air quality in London and across the world. A number of global cities have sought to better understand the impacts of COVID-19 lockdown measures on air quality. London is working closely with the C40 Cities Climate Leadership Group to share information and best practice.

Our evidence shows that levels of the harmful gas nitrogen dioxide (NO_2) in central London are on average about 40 per cent lower than before the lockdown – a huge reduction, caused by dramatic reductions in traffic. These reductions are in addition to those delivered by the Ultra Low Emission Zone (ULEZ) and other policies in the last four years.

However, London's poor air quality is not just about traffic pollution. Even during the lockdown, Londoners have suffered from particulate pollution episodes, sources of which include domestic burning and agricultural emissions.

We have also today published an accompanying report on the first ten months of the ULEZ, to provide important context to the COVID-19 reductions. The report shows that policies, including the ULEZ, have contributed to a reduction of 44 per cent in roadside NO_2 in central London between February 2017 and January 2020. This shows that dramatic changes were already happening to London's air quality before the lockdown and confirms the effectiveness of clean air zones in tackling air pollution.

MAYOR OF LONDON

I hope this evidence further enhances our understanding of the causes of London's poor air quality, and that we can work together urgently to act on these. It is absolutely essential that the Environment Bill is not delayed, and that it includes ambitious and legally binding targets which meet or exceed World Health Organization recommended limits.

Clearly, tackling the current crisis must be our first priority. However, as we start to think about recovery, I hope you will agree that it must be a green one, which includes the eradication of air pollution permanently and maintains the gains we have made through policies such as the ULEZ so we can continue to protect people's health.

Yours sincerely,

Sadiq Khan
Mayor of London

Cc: Paul Scully MP, Minister for London

Encs:

- Report on the estimation of changes in air pollution in London during the COVID-19 outbreak
- ULEZ report

City Hall, The Queen's Walk, London, SE1 2AA ♦ mayor@london.gov.uk ♦ london.gov.uk ♦ 020 7983 4000

MAYOR OF LONDON

CENTRAL LONDON ULTRA LOW EMISSION ZONE – TEN MONTH REPORT

April 2020

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Foreword

On 8 April 2019 the Mayor of London launched the world's first Ultra Low Emission Zone (ULEZ). Data indicates in the first ten months of the scheme it had a significant and immediate impact – although further analysis will be needed to fully assess the long-term impacts.

This report includes data from February 2017 (when the Mayor confirmed the T charge and the change in the vehicle fleet began), March 2019 (the month before the scheme was introduced) and April 2019 – January 2020 (the first ten months of the scheme).

The report has been published to provide important context for the changes reported in London's air quality as a result of COVID-19 measures. This information is important as there is emerging evidence linking air pollution with an increased vulnerability to the most severe impacts of COVID-19.

The period covered in this report pre-dates changes associated with COVID-19. These subsequent changes are addressed in a separate document available here: https://www.london.gov.uk/WHAT-WE-DO/environment/environment-publications/estimation-changes-air-pollution-during-covid-19-outbreak-0

Further information on the impact of ULEZ and the other air quality measures delivered by the Mayor will be published in due course.

Key Findings

On 8 April 2019 the Mayor of London launched the world's first Ultra Low Emission Zone (ULEZ).

Key findings from the first ten months of operation are:

- Trend analysis shows that concentrations of NO_2 at roadside sites in the central zone in February 2020 are 39 µgm^{-3} less that in February 2017, when changes associated with the ULEZ began. This is a **reduction of 44 per cent**. This is over double the reduction at inner roadside sites, of 18 µgm^{-3}, and four times the reduction at roadside sites in outer London. The smallest improvement was recorded at urban background sites in outer London, 6 µgm^{-3}. This underlines the need for expanding the central London ULEZ to the North and South Circular roads in 2021
- After the first ten months of operation, in January 2020 the **average compliance rate with the ULEZ standards was 79 per cent** in a 24 hour period (77 per cent in congestion charging hours). This is significantly higher than 39 per cent in February 2017 and the 61 per cent in March 2019 during congestion charging hours
- Analysis to determine the directly attributable impact of the ULEZ shows that, for the period January to February 2020, NO_2 concentrations at roadside locations in central London were on average 29 µgm^{-3} lower, equating to **a reduction of 37 per cent**, compared to a scenario where there was no ULEZ
- Preliminary estimates indicate that by the end of 2019 NO_x emissions from road transport in the central zone have **reduced by 35 per cent (230 tonnes)** compared to a scenario where there was no ULEZ. This is on track to achieve a 45 per cent reduction in the first year of the scheme.
- Preliminary estimates indicate that by the end of 2019 CO_2 emissions from road transport in the central zone have **reduced by 6 per cent (12,300 tonnes)** compared to a scenario where there was no ULEZ.

- None of the air quality monitoring stations located on ULEZ boundary roads have measured an increase in NO_2 concentrations since the introduction of the ULEZ indicating no issue with the displacement of traffic and related emissions
- Preliminary analysis of traffic flows indicate that the introduction of the central London ULEZ has contributed to a **reduction in traffic flows in central London from May 2019 to January 2020 of between 3 – 9 per cent** when compared to 2018, though further analysis is needed to better understand long term complex changes in traffic flows as a result of ULEZ
- From March 2019 to January 2020 there was a large reduction in the number of older, more polluting, non-compliant vehicles detected in the zone: **some 17,400 fewer on an average day, a reduction of 49 per cent** in congestion charging hours. This is higher than the 13,500 reduction reported after 6 months.
- There was a **41 per cent decrease** in the proportion of vehicles in the central zone that were non-compliant from March 2019 to January 2020 in congestion charging hours

To fully understand the impact of the scheme it is necessary to take into account pre-compliance (i.e. people and businesses preparing ahead of time for the start of the new scheme). With this in mind, the changes between February 2017 and January 2020 were as follows:

- There was a large reduction in the number of older, more polluting, non-compliant vehicles detected in the zone: **a reduction of 44,100 vehicles on an average day, equating to a 71 per cent reduction**
- There was a **96 per cent increase** in the proportion of vehicles detected in the central zone that were compliant from February 2017 to January 2020
- The average 24 hour compliance rate for all vehicles was 79 per cent in January 2020. However, there was a large discrepancy between different vehicle types. HGVs have the highest compliance of any vehicle groups (excluding TFL buses which are 100% compliant from the start of the scheme) with 90 per cent. Taxis had the lowest compliance rate with only 29 per cent.

Introduction

On 8 April 2019 the Mayor of London launched the world's first Ultra Low Emission Zone (ULEZ) in central London. This chapter of the report evaluates the impact of the scheme in its first ten months of operation (to the end of January 2020). Whilst we can determine a number of different impacts within this timeframe, further ongoing analysis will be required to understand the full impacts of the scheme over a longer period of time – particularly in relation to establishing long term changes in air quality.

A number of measures are used to assess the impacts of introducing the ULEZ. Here we evaluate the impact on air pollution concentrations, air pollution emissions, traffic flows and vehicle compliance.

What is the Ultra Low Emission Zone (ULEZ)?

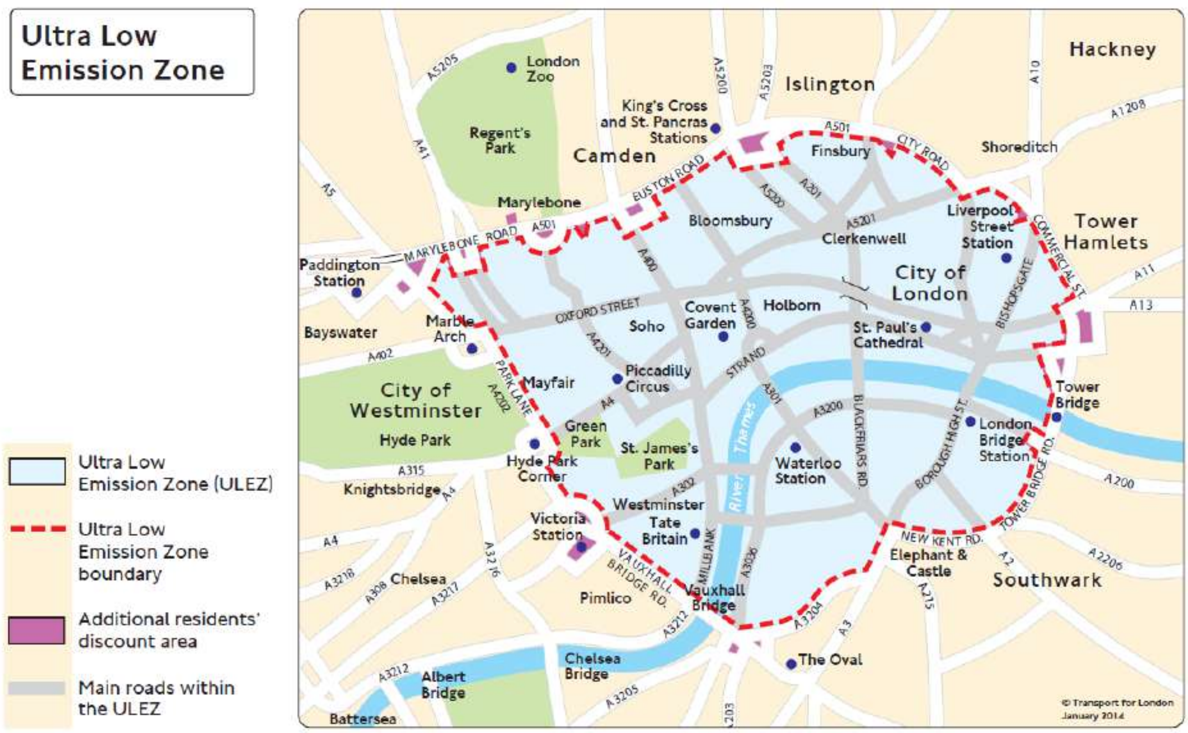

Figure 1. Map of the central London Ultra Low Emission Zone

The Central London ULEZ started on 8 April 2019 and operates in the existing central London Congestion Charge Zone. Figure 1 is a map of the area covered by the central ULEZ. Unlike the Congestion Charge (which operates Monday to Friday between 07:00 and 18:00) the ULEZ operates 24 hours a day, every day of the year except Christmas Day (25 Dec). Vehicles must meet strict emission standards to drive in the ULEZ area:

- Euro 4 for petrol cars and vans (vehicles less than fourteen years old in 2019)
- Euro 6 for diesel cars (vehicles less than five years old in 2019)
- Euro 6 for diesel vans (vehicles less than four years old in 2019)
- Euro 3 for motorcycles and other L-category vehicles
- Euro VI for lorries, buses and coaches

Vehicles that do not meet these standards must pay a charge:

- £12.50 per day for cars, motorcycles and vans
- £100 per day for lorries, buses and coaches

All TfL buses operating in the zone meet the ULEZ standards. The ULEZ replaced the T-Charge in central London and is in addition to the Congestion Charge. Alongside the ULEZ, the Private Hire Vehicle exemption to the Congestion Charge was removed on 8 April 2019 and the Ultra Low Emission Discount was replaced by the new Cleaner Vehicle Discount.

To find out more about the ULEZ or to check if your vehicle is affected please visit:

https://tfl.gov.uk/modes/driving/ultra-low-emission-zone.

This is the fourth report evaluating the impacts of the scheme. The first three reports are available from:

- Central London Ultra Low Emission Zone – First Month Report
- Central London Ultra Low Emission Zone – Four Month Report
- Central London Ultra Low Emission Zone – Six Month Report

An updated evaluation will be published once twelve months of data are available.

Assessing the impacts of ULEZ

The purpose of the ULEZ is to improve air quality in and around central London by reducing the number of older more polluting vehicles that enter the central zone. The impact of the ULEZ can be assessed using a number of different metrics including:

- Air quality monitoring[1]
- Modelling of vehicle emissions
- Number of vehicles and compliance rates
- Traffic flow data

Air pollution concentrations are affected by many different factors including the weather and regional contributions from outside London, as well as impacts from other local schemes, therefore analysis of air quality monitoring data will need to continue over time.

Vehicle compliance refers to the number of vehicles that "comply" or meet the ULEZ emission standards. Non-compliant vehicles do not meet the strict ULEZ emissions standards and have either:

- Paid the daily charge
- Incurred a penalty charge
- Not been required to pay the daily ULEZ charge as they are eligible for a 100% discount or exemption

[1] At this stage air quality data is from the London Air Quality Network and Air Quality England Network. This is because both provide data going back many years. The newly established Breathe London network will also be used for ULEZ evaluation in a separate report using different techniques.

Limitations of this analysis

To assess the impact of the scheme we have compared the number of vehicles detected in the zone and compliance rates from February 2017 and March 2019 – January 2020. In February 2017 the Mayor confirmed the introduction of the T-charge as a stepping-stone for the ULEZ and this can be seen as the start of the accelerated change in the vehicle fleet as Londoners and businesses prepared for the new schemes and buses on routes in central London began to be upgraded to become ULEZ compliant. In addition, the removal of the exemption from the Congestion Charge for private hire vehicles also commenced on 8 April 2019. TfL have also introduced new licensing requirement for private hire vehicles so that as of 1 January 2020:

- PHVs under 18 months old must be zero emission capable and meet the Euro 6 emissions standard when licensed for the first time
- PHVs over 18 months old must have a Euro 6 (petrol or diesel) engine when licensed for the first time

March 2019 is the month before the ULEZ was introduced and January 2020 is the latest available full month of data.

The ULEZ is a 24 hour scheme, however, prior to the start of the scheme in April 2019 data could only be collected during congestion charging (CC) hours – 07:00 to 18:00, Monday to Friday. When assessing the impact of the first ten months of ULEZ compared to historic months, comparison has been made based on CC hours to ensure the comparison is fair. 24 hour data for the months since the scheme has been in operation has also been provided.

As mentioned, the removal of the exemption from the Congestion Charge for private hire vehicles coincided with the launch of the ULEZ. This may also have had an effect on traffic volumes and air quality within the zone, but it is too early at this stage to separate the respective effects.

Disruptions to traffic flow in the central zone in April 2019

As explained in a previous iteration of this report, there were a number of non-typical events in central London in April 2019. These included.

- Road works (leading to signed diversions into the ULEZ)
- The Extinction Rebellion climate protests, leading to further diversions into the central zone and an unknown impact on the number of motorists choosing to drive in central London
- Easter Holidays and Bank Holidays. The timing of the introduction of ULEZ was specifically chosen to target a "quiet" week when there would be fewer vehicles in the zone

As a result, only a limited number of days were used for analysis of the first month of the scheme. Data for April 2019 presented in this report is the average over "typical days" only. However, using only typical days exclusively in the month of April has little effect on the results.

As the scheme started on 8 April, the first iteration of this report covered the period from 8 April to 5 May 2019 (to provide 4 calendar weeks of "typical days" data). For consistency this report has taken the same approach.

Unique vehicles detected in zone and relation to traffic flow

Vehicle volumes within this report relate to the daily number of confirmed unique vehicles detected in central London. Unique vehicle volumes will be different in scale to changes in traffic volumes entering or within central London for a number of reasons:

- Unique vehicle volumes do not take into account how a vehicle is used. For example, a proportion of traffic is associated with a minority of vehicles that make multiple trips a day within the zone, e.g. delivery vehicles, private hire vehicles and taxis
- Trips made wholly within the zone are currently less likely to be captured by an ANPR camera than trips crossing the boundary (for which all entry and exit points are monitored). There is currently less incentive for internal trips to cease as local residents have a 100% ULEZ discount grace period until 24th October 2021
- Analysis of changes in traffic data based on automatic traffic count sites in London is compared to the same months in 2018. However, traffic exhibits seasonal variation and further analysis will be undertaken once a full year of traffic data is available

If you want to know about estimates for changes in traffic in both central London and pan-London please see the latest Travel in London report, which looks at various sets of data for understanding traffic flow including that from TfL's automatic traffic counters: https://tfl.gov.uk/corporate/publications-and-reports/travel-in-london-reports

Further analysis is ongoing in order to understand the impacts of ULEZ including trends in changes in compliance, traffic flows, and air quality.

Air pollution concentrations

Around half of London's NO_x emissions are from road transport.[2] The purpose of the ULEZ is to improve air quality in and around central London by reducing the number of older, more polluting vehicles that enter the central zone. This will reduce the amount of NO_x emitted, which in turn will reduce nitrogen dioxide (NO_2) concentrations in the zone. Bringing London closer to compliance with the legal air quality limit values for NO_2 is a key aim of the scheme.

The analysis presented here uses data from London's automatic monitoring network. This data is publicly available from the London Air Quality Network and Air Quality England websites. Full details of the methodology for this chapter can be found in the Central London Ultra Low Emission Zone Six Month Report.

In this analysis monthly average concentrations are used to calculate trends in the period from 2010 to end of February 2020. It should be noted that measurement data from late 2019 and early 2020 have not yet been ratified. As a result, these may be subject to change following equipment tests undertaken as part of the routine audit and servicing of air quality monitoring sites.

Trends in nitrogen dioxide (NO_2)

The ULEZ was introduced part way through 2019, therefore evaluating changes with respect to the ULEZ required analysis on a shorter timescale than a year. To address this we evaluate the change in quarterly (three month) average NO_2 concentrations. It is important to note that Table 1 presents an average across several sites of each type in each zone. Data presented in Table 1 is quarterly as opposed to annual, so is not directly comparable to annual air quality limits.

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[2] London Atmospheric Emissions Inventory 2016 (LAEI 2016)

This analysis evaluates the change in quarterly average NO_2 since February 2017, when changes associated with the ULEZ began. The additional analysis estimated the proportion of reductions in NO_2 that are directly attributable to ULEZ.

Table 1 lists the quarterly average concentrations of NO_2 in London from January 2016 to February 2020 grouped by site type and London zone. The biggest reduction in average concentrations between the beginning of 2017 and February 2020 is at central roadside sites, 39 µgm^{-3}, equating to a 44 per cent reduction. This is over double the reduction at inner roadside sites of 18 µgm^{-3}, and four times the reduction at roadside sites in outer London. The smallest improvement was recorded at urban background sites in outer London, 6 µgm^{-3}.

Table 1. Quarterly average NO_2 from January 2017 to February 2020

Period	Average NO_2 [µgm^{-3}]					
	Roadside Central	Background Central	Roadside Inner	Background Inner	Roadside Outer	Background Outer
Jan – March 17	89	37	54	34	46	29
April – June 17	87	36	53	34	45	29
July – Sept 17	86	36	52	33	45	29
Oct – Dec 17	83	35	51	33	44	28
Jan – March 18	81	35	50	32	44	28
April – June 18	78	34	49	31	43	28
July – Sept 18	75	34	48	31	42	27
Oct – Dec 18	71	33	46	30	41	27
Jan – March 19	67	33	45	29	40	26
April – June 19	63	32	43	29	39	26
July – Sept 19	59	32	42	28	38	25
Oct – Dec 19	54	32	40	27	37	25
Jan – Feb 20*	50	31	38	27	36	24
Reduction *(Q1 2017 – Q1 2020)* **[µgm^{-3}]**	39	6	16	7	10	5
Reduction *(Q1 2017 – Q1 2020)* **[per cent]**	44%	16%	30%	21%	22%	17%

*Data available to 1 March 2020

Again, this is not comparable to the annual mean limit, as seen in a previous chapter there were still many sites in 2019 in inner and outer London that exceeded the legal air quality limit value for annual mean NO_2 of 40 µgm^{-3}.

As mentioned previously, air pollution is influenced by many complex factors. It is therefore important to perform additional analysis to ensure the trends reported in Table 1 were not a product of weather and seasonal factors and to attribute the proportion of the recent reduction in NO_2 concentrations within the central zone which are attributable to the ULEZ.

Changes in NO_2 attributable to the ULEZ

The ULEZ is one of the many policies to reduce air pollution in London. Other policies include the Londonwide Low Emission Zone (for heavy vehicles), investment in new cleaner buses and ZEC licensing requirements for taxis and PHVs (in addition to ULEZ measures), as well as progressively tighter EU-wide exhaust controls for new vehicles. As a result, it is not straight forward to isolate the impact of the ULEZ. For this analysis the trends in outer London (largely away from the influence of the ULEZ in central London) were used as a predictor of the change in central and inner London if the ULEZ was not in place. The change in outer London reflects the “natural churn” of the fleet, as vehicles are replaced by their owners. The changes measured in central London far exceed natural churn. Comparing the measured trends in central and outer London reveals the additional changes within the central zone, which provide an estimate for the impact of the ULEZ.

Detecting the additional change within the ULEZ by comparing trends in the zone to those in outer London has both strengths and weaknesses. Key amongst the strengths are the ease of analysis, allowing data to be analysed as it is produced, and the large number of measurement sites involved. Another strength is the use of outer London data that also acts, to some extent, as a control for the weather and seasonal factors that can confound this type of analysis. The key weakness stems from differences in the vehicle fleets in the ULEZ area compared with outer London. Traffic in the ULEZ area has a greater proportion of certain vehicle types, such as taxis, and proportionally fewer private cars than outer London[3]. Interventions on these vehicle types from other Mayoral policies would have a different impact in the ULEZ area than outside, even in the absence of the ULEZ.

Another potential limitation to the analysis presented in this chapter is changes in the number and location of monitoring sites across London over the 10-year period. More

[3] London Atmospheric Emission Inventory (LAEI) 2016, Greater London Authority 2018

detail on this can be found in the Central London Ultra Low Emission Zone Six Month Report.

A technique often used to isolate the proportion of pollution that relates to traffic sources is to subtract the background concentration from the roadside concentration. This is referred to as the "roadside increment"[4]. Changes in the roadside increment, or traffic contribution, in outer London were used as a predictor of the changes in a "no ULEZ" scenario for roadside sites in central and inner London - the rate of change in outer London is an approximation of what would see in central London if there were no ULEZ policy. The analysis in this section follows the exact method for calculating the "no ULEZ" trend that can be found in the Appendix of the Central London Ultra Low Emission Zone – Six Month Report.

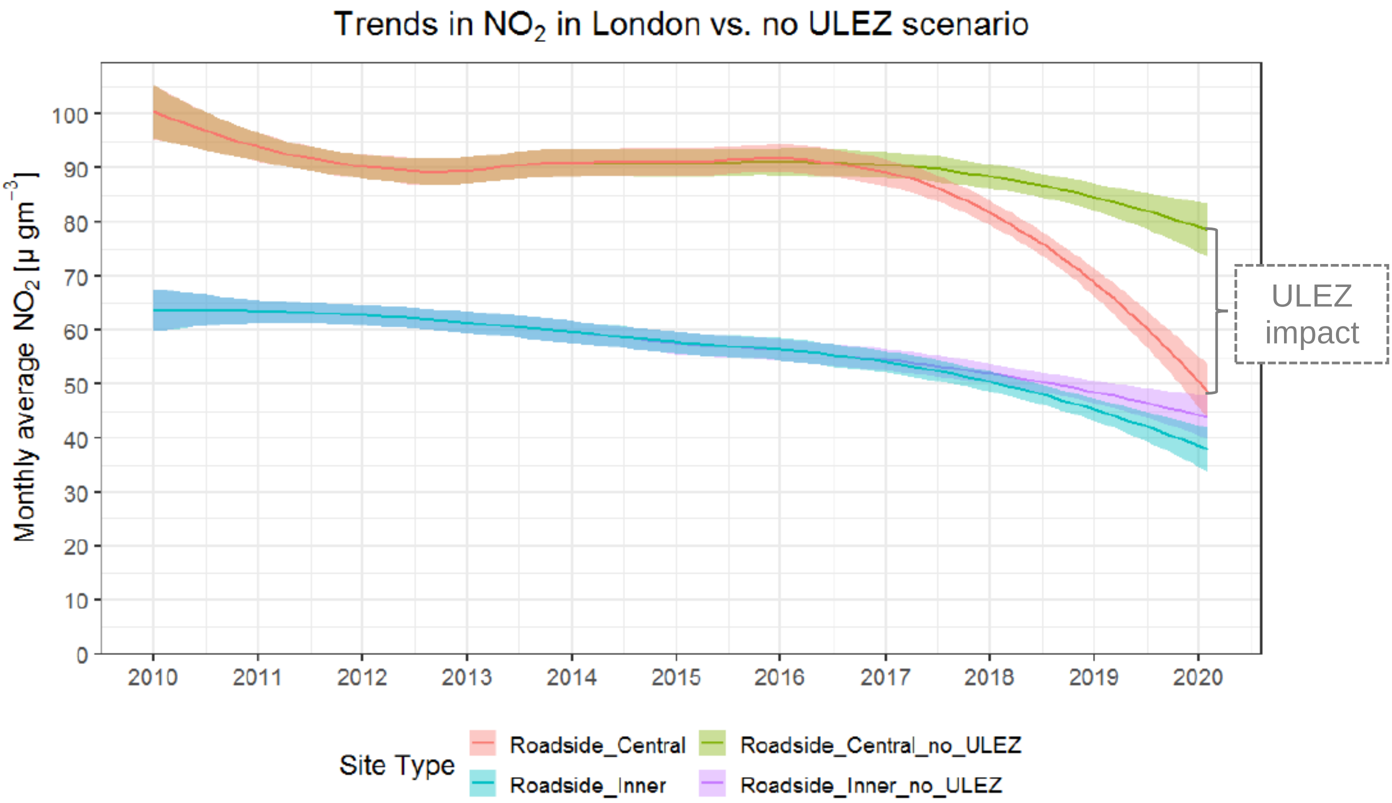

Figure 2: Monthly average NO_2 concentrations in London with and without ULEZ

[4]Font, A. & Fuller, G. (2016) Did policies to abate atmospheric emissions from traffic have a positive effect in London? Environmental Pollution, Volume 218, November 2016, Pages 463-474

Figure 2 shows the monthly average NO_2 at roadside sites in central and inner London as well as a “no ULEZ” scenario estimate for each. The “no ULEZ” reflects changes in central and inner London were they to follow the same trend as roadside sites in outer London. The divergence between the measured concentrations and “no ULEZ” scenario is much more pronounced in central London than in inner London. This shows there was a reduction in roadside concentrations in central and inner London that was far greater than the reduction measured at outer London sites.

Table 2 presents the difference between the trend in actual roadside measurements and the scenario where there was no ULEZ over three-month periods since April 2019. This can be understood as the reduction at central and inner London sites that is in addition to the changes measured at outer London roadside sites.

Table 2: Estimated reduction in NO_2 concentrations as a result of ULEZ

Period	Reduction central London roadside compared to “no ULEZ”		Reduction inner London roadside compared to “no ULEZ”	
	[μgm^{-3}]	[per cent]	[μgm^{-3}]	[per cent]
Jan – March 19	17	20%	3	7%
April – June 19	20	24%	4	9%
July – Sept 19	23	29%	5	10%
Oct – Dec 19	26	33%	5	12%
Jan – Feb 20*	29	37%	6	13%

*Data available to 1 March 2020

In January to February 2020, the most recent period for which data is available, the ULEZ is estimated to have reduced mean NO_2 concentrations at roadside sites by 29 μgm^3, a reduction of 37 per cent compared to the scenario where “no ULEZ” is in place.

A smaller reduction of 13 per cent was estimated at roadside sites in inner London. This is expected, since many vehicles driven in the ULEZ also travel in this area. This is the area that will benefit most from the expansion of the Ultra Low Emission Zone to the North and South circular roads in 2021.

Trends in NO_2 on boundary roads

When charging schemes, such as the ULEZ or Congestion Charge, are introduced in part of a city it is always important to measure the impact of the scheme not only in the zone itself, but also in the surrounding area.

There are four established air quality monitoring stations on the central London ULEZ boundary roads. Figure 3 shows that, similar to sites within the central zone, sites on the ULEZ boundary roads measured a continued downward trend in concentrations since 2017.

No sites on the boundary roads have experienced an increase in the trend of monthly average NO_2 since the scheme was introduced in April 2019. (Note, these boundary sites are categorised as inner, as opposed to central, sites).

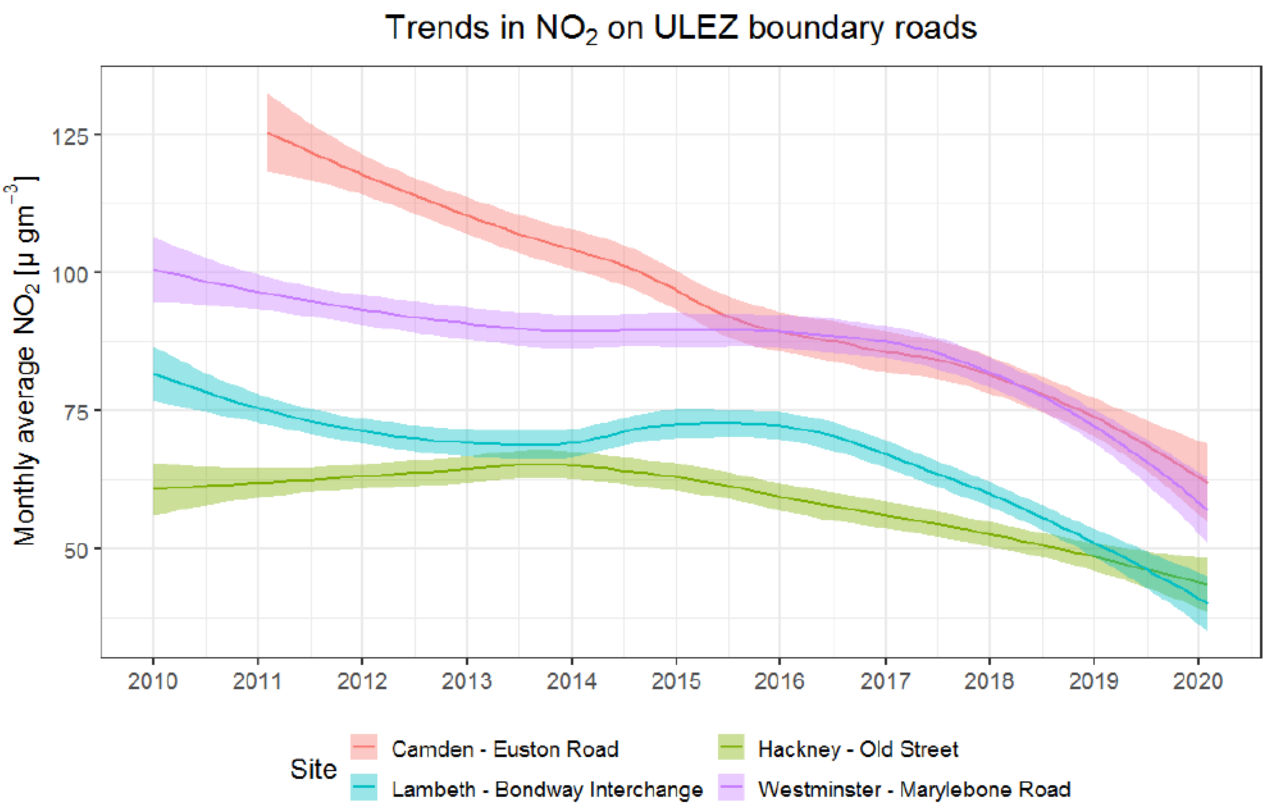

Figure 3: Monthly average NO_2 concentrations at sites on ULEZ boundary roads

This is a strong indication that there has been a positive impact on air pollution on the ULEZ boundary roads. A full picture of the impact on boundary roads will be available later in 2020 (once more data is available and the ULEZ has been in operation a full year).

Trends in fine particulate matter ($PM_{2.5}$)

As mentioned previously in this report, road transport is the largest single source of particulate matter in London, accounting for around 30 per cent of emissions. However, unlike NO_2, over half of London's concentrations of $PM_{2.5}$ come from regional, and often transboundary (non-UK) sources outside of London. There is also a large proportion of $PM_{2.5}$ emitted within London that the Mayor does not currently have the powers to address, for example wood burning. In addition, a growing proportion of road transport $PM_{2.5}$ emissions are now non-exhaust emissions including road wear, resuspension of road dust and tyre and brake wear.

Table 3. Quarterly average $PM_{2.5}$ from January 2017 to February 2020

Period	Average $PM_{2.5}$ [μgm^{-3}]					
	Roadside Central	Background Central	Roadside Inner	Background Inner	Roadside Outer	Background Outer
Jan – March 17		15	14	12	13	11
April – June 17		14	14	12	13	11
July – Sept 17		14	14	12	12	11
Oct – Dec 17		14	14	11	12	11
Jan – March 18		14	13	11	12	11
April – June 18		13	13	11	12	11
July – Sept 18		13	13	11	12	11
Oct – Dec 18		13	13	11	12	11
Jan – March 19		12	12	11	12	11
April – June 19		12	12	10	12	10
July – Sept 19		11	12	10	11	10
Oct – Dec 19		11	12	10	11	10
Jan – Feb 20*		11	11	9	11	10
Reduction *(Q1 2016 – Q1 2020)* **[μgm^{-3}]**		4	3	3	2	1
Reduction *(Q1 2016 – Q1 2020)* **[per cent]**		27%	21%	25%	15%	9%

Table 3 shows the quarterly average $PM_{2.5}$ grouped by zone and site type. Since changes associated with the ULEZ began in February 2017 there has been a 27 per cent reduction in quarterly average $PM_{2.5}$ emission in background sites located in central London. It is likely these will have been influenced by the reduction in traffic emissions, as was seen in annual average PM_{10}.

Air pollution emissions

Emissions from road transport have been modelled to estimate how NO_x emissions from vehicles have changed since the ULEZ was introduced. Full details of the methodology can be found in the Central London Ultra Low Emission Zone Six Month Report. Emissions reductions are calculated as the reduction in emissions using current compliance rates compared to a “no ULEZ” scenario for the period October to December 2019. These are estimates based on the first three quarters of operations, a full update after a full year will be included in a report evaluating the first 12 months of the scheme.

Reductions in NO_x emissions

Preliminary estimates indicate that between October to December 2019 NO_x emissions from road transport reduced by 35 per cent (or 230 tonnes of NO_x) compared to a scenario where there was no ULEZ. Modelling done by TfL as part of the ULEZ consultation process estimated that introducing the ULEZ would result in a 45 per cent reduction in NO_x emissions from road transport in the central zone. After the first three quarters of a year in operation the ULEZ is on track to meet its 45 per cent target.

Reductions in $PM_{2.5}$ emissions

Similarly, it has been estimated that between October to December 2019 $PM_{2.5}$ emissions from road transport reduced by 6 tonnes, a reduction of 15 per cent compared to a no ULEZ scenario. As discussed, total $PM_{2.5}$ emissions are more sensitive to changes in vehicle kilometres due to the dominance of non-exhaust particles. This will be addressed by policies in the Mayor’s Transport Strategy that will reduce traffic volumes by encouraging mode shift from car to walking, cycling and using public transport., The Mayor aims for 80 per cent of all trips in London to be made on foot, by cycle or using public transport by 2041.

Reductions in CO_2 emissions

CO_2 emissions in the central zone are estimated to have reduced by 12,300 tonnes, a reduction of 6 per cent, compared to a scenario with no ULEZ in place. This is equivalent to the lifetime carbon savings of over 800 solar PV installations in London. CO_2 emissions are also more sensitive to changes in vehicle kilometres due to the dependence on fuel use.

Summary of emissions reductions

Table 4 presents the summary of emissions reductions by pollutant. In future analysis, once more data is available, fleet composition estimates will be revised to take account of a full year of data and consider other changes in vehicle types, such as fuel type, and further assessment of traffic flows. Further emissions calculations will be carried out for a one-year evaluation report including the impact of the central London ULEZ on road transport NO_x, $PM_{2.5}$ and CO_2 emissions in both inner and outer London.

Table 4. Summary of emissions reductions in central zone

Pollutant	Comparison to “no ULEZ” scenario, Oct – December 2019	
	Reduction [tonnes]	Reduction [per cent]
NO_x	230	35%
$PM_{2.5}$	6	15%
CO_2	12,300	6%

Traffic flows

Transport for London uses automatic traffic count data at representative sites across London to monitor changes in traffic flows. These sites provide total traffic flows (for all vehicles) for each hour of the day. In this analysis the sites have been averaged over each month to allow estimates of changes in traffic flows in central, inner and outer London to be determined.

Traffic flows change across the year reflecting seasonal patterns such as holiday periods. Therefore, the best way to evaluate a change in traffic flow is to compare to the same period in previous years. In Table 5 monthly data for 2019 has been compared to 2018 and the percentage change in average flows calculated.

Table 5: Change in average 24 hour traffic flows in London from 2018 to 2019

	All days of week			Weekdays			Weekends		
Comparison 2019 to 2018	**Central**	**Inner**	**Outer**	**Central**	**Inner**	**Outer**	**Central**	**Inner**	**Outer**
January	0%	-1%	2%	0%	-1%	2%	-1%	-1%	2%
February	0%	-1%	2%	0%	-1%	2%	0%	-2%	2%
March	2%	2%	4%	1%	2%	3%	4%	3%	6%
April	-2%	-2%	2%	-2%	-1%	2%	-3%	-2%	1%
May	-3%	-1%	1%	-2%	-2%	1%	-6%	0%	1%
June	-5%	0%	0%	-5%	0%	0%	-6%	1%	0%
July	-5%	-1%	1%	-5%	-2%	1%	-5%	0%	1%
August	-8%	-4%	1%	-7%	-4%	0%	-9%	-3%	3%
September	-9%	-2%	0%	-9%	-2%	0%	-11%	-1%	0%
October	-9%	0%	-2%	-8%	0%	-2%	-11%	-1%	-2%
November	-7%	0%	-2%	-6%	0%	-2%	-9%	0%	-2%
December	-6%	-1%	0%	-5%	-1%	0%	-8%	-1%	0%
January	-8%	0%	0%	-7%	0%	0%	-10%	0%	1%

The table shows that in early 2019 there was very little change in average traffic flows in central and inner London when compared to 2018, whilst there was around 2 per cent increase in outer London. Traffic in inner and outer London between April and July varied by up to a couple of percent compared to the same months in 2018. However, after March reductions in average traffic flows of around 3 – 9 per cent are reported in central London when compared to the previous year. Similar estimates have been seen across both weekdays and weekends.

This is an indication that the introduction of the ULEZ is contributing to a reduction in traffic flows in central London. Across the year the average change comparing 2019 to 2018 is estimated to be a 4.5 per cent reduction in central London. However, it is too soon to fully attribute these changes solely to ULEZ, as more data is required for analysis over a longer period.

When comparing weekdays, a similar pattern is seen – whereby changes in central London in 2018 are greater than those for inner London. For weekends, the difference appears to be greater still. This is likely to reflect the fact that weekends are now subject to a charge for the first time, unlike congestion charging which only affects weekdays.

Analysis of changes in traffic flows across different times of the day has also been analysed. The results are similar to those seen for 24 hour data. However, the data suggests more substantial differences between 2018 and 2019 in the evening, late evening and night time hours – which are hours where charges have not been applied before.

Traffic flow changes are still preliminary, and data will continue to be collected over the coming months in order to understand if trends are sustained, and how these vary across the different times of day and weekends, and on specific roads across the network.

Number of vehicles and compliance rates

FIRST MONTH – changes in vehicle numbers and compliance (March 2019 – April 2019)

Table 6 compares vehicle numbers and compliance rates for the month immediately before the scheme was introduced (March 2019) and the scheme's first month of operation (April 2019). As explained earlier in this chapter, this excludes non-typical days.

The changes below capture the more immediate effect following the launch of the scheme and do not take into account those who changed their behaviour ahead of time in preparation for the scheme.

Table 6. Average number and proportion of compliant vehicles detected in the zone per 'typical' day during CC hours March 19 – April 19

Month	Number of vehicles driving in the charging zone per day during CC hours			Proportions of vehicles driving in the charging zone during CC hours	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
Mar - 19	91,035	35,578	55,457	39.1%	60.9%
Apr – 19	89,380	26,195	63,185	29.3%	70.7%
Change	-1,655	-9,383	7,728	Decrease of 9.8 percentage points	Increase of 9.8 percentage points
% change	-1.8%	-26.4%	13.9%	-25.0%	16.1%

*not representative of traffic flow

Key impacts of the first month of the scheme compared to the previous month:

- In the first month of operation (excluding non-typical days) the compliance rate with the ULEZ standards in congestion charging hours was around 71 per cent. This is much higher than the 61 per cent in March 2019
- There was a large reduction in the number of older, more polluting, non-compliant vehicles detected in the zone: some 9,383 fewer on an average 'typical' day, a reduction of over a quarter

FIRST TEN MONTHS – changes in vehicle numbers and compliance (March 2019 – January 2020)

Table 7 compares vehicle numbers and compliance rates for the month immediately before the scheme was introduced (March 2019) and the scheme's first ten months of operation. This excludes non-typical days for April 2019. The table below captures the more immediate effect following the launch of the scheme and does not take into account those who changed their behaviour ahead of time in preparation for the scheme, this is captured in the pre-compliance data presented later in this report.

Table 7. Average number and proportion of unique compliant vehicles detected in the zone during CC hours March 19 – January 20

Month	Number of vehicles driving in the charging zone per day during CC hours			Proportions of vehicles driving in the charging zone during CC hours	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
March 19	91,035	35,578	55,457	39.1%	60.9%
April 19	89,380	26,195	63,185	29.3%	70.7%
May 19	88,796	25,610	63,186	28.8%	71.2%
June 19	87,113	24,549	62,564	28.2%	71.8%
July 19	83,899	23,054	60,844	27.5%	72.5%
August 19	80,128	21,133	58,994	26.4%	73.6%
Sept 19	85,854	22,133	63,721	25.8%	74.2%
Oct 19	82,776	21,239	61,537	25.7%	74.3%
Nov 19	84,797	21,222	63,575	25.0%	75.0%
Dec 19	84,032	20,533	63,499	24.4%	75.6%
Jan 20	78,754	18,182	60,572	23.1%	76.9%
Change March 19 – Jan 20	-12,281	-17,396	5,115	Decrease of 16 percentage points	Increase of 16 percentage points
% change	-13%	-49%	9%	-41%	26%

*not representative of traffic flow

Key impacts of the first ten months of the scheme compared to March 2019 (the month before the scheme was implemented):

- In January 2020 the compliance rate with the ULEZ standards was 77 per cent. This is much higher than the 61 per cent in March 2019.

- From March 2019 – January 2020 there was a large reduction in the number of older, more polluting, non-compliant vehicles detected in the zone: some 17,396 fewer on an average day, a reduction of around 49 per cent.

- There was around a 41 per cent decrease in the proportion of vehicles in the central zone that were non-compliant between March 2019 and January 2020.

PRE- COMPLIANCE – changes in vehicle numbers and compliance (February 2017 – March 2019)

Table 8 below shows the change in the number of vehicles detected in the zone and the compliance level between February 2017 and March 2019. This data was released in April 2019 to coincide with the launch of the scheme.[5]

Table 8. Average number and proportion of unique compliant vehicles detected in the zone per day during CC hours February 17 – March 19

Month	Number of vehicles driving in the charging zone per day during CC hours			Proportions of vehicles driving in the charging zone during CC hours	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
Feb 17	102,493	62,310	40,184	60.8%	39.2%
March 19	91,035	35,578	55,457	39.1%	60.9%
Change Feb 17 – March 19	-11,458	-26,732	15,273	Decrease of 21.7 percentage points	Increase of 21.7 percentage points
% change	-11%	-43%	38%	-35.7%	55.4%

*not representative of traffic flow

As Table 8 indicates, the proportion of compliant vehicles detected in the Central London ULEZ zone rose from 39 per cent in February 2017 (when the Mayor confirmed the introduction of the T-charge) to 61 per cent in March 2019. This represents a 55 per cent increase in the proportion of compliant vehicles detected in the zone.

[5] https://www.london.gov.uk/press-releases/mayoral/ulez-launches-in-central-london

The proportion of vehicles that are compliant is the best way of comparing changes in the vehicle fleet, given the number of unique vehicles detected in the zone also changed over this period.

PRE- COMPLIANCE and LATEST MONTH – changes in vehicle numbers and compliance (February 2017 – January 2020)

Table 9 shows the change in vehicle compliance from February 2017 to January 2020. This is presented as an absolute change in the number of vehicles detected, the change in the percentage of vehicles that are compliant, and also the change in the proportion of vehicles that are compliant.

Table 9. Average number and proportion of unique compliant vehicles detected in the zone during CC hours February 17 – January 20

Month	Number of vehicles driving in the charging zone per day during CC hours			Proportions of vehicles driving in the charging zone during CC hours	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
Feb 17	102,493	62,310	40,184	60.8%	39.2%
Jan 20	78,754	18,182	60,572	23.1%	76.9%
Change Feb 17 – Jan 20	-23,739	-44,128	20,388	Decrease of 38 percentage points	Increase of 38 percentage points
% change	-23%	-71%	51%	-62%	96%

*not representative of traffic flow

Key findings for the first ten months of the scheme compared to February 2017, taking pre-compliance into account:

- From February 2017 to January 2020 there was a large reduction in the number of older, more polluting, non-compliant vehicles detected in the zone: some 44,128 fewer on an average day, a reduction of 71 per cent.
- There was a 96 per cent increase in the proportion of vehicles detected in the zone that met the ULEZ standards between February 2017 and January 2020. As mentioned previously, the proportion of vehicles that are compliant is the best way of

comparing changes in the vehicle fleet, given the number of unique vehicles detected in the zone also changed over this period.

Comparison between congestion charge hours and 24 hour data

To ensure a fair comparison with historic data the previous analysis compares data for CC hours only. Table 10 below includes vehicle numbers and compliance rates for CC hours and 24 hour average daily vehicles detected in the zone for January 2020. The 24 hour compliance rate in January 2020 was 79 per cent.

Table 10. Comparison of average unique daily vehicles for January 2020 for CC hours and 24 hour data

Time	Number of vehicles driving in the charging zone per day			Proportions of vehicles driving in the charging zone	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
CC hours	78,754	18,182	60,572	23.1%	76.9%
24 hour	106,664	22,255	84,409	20.9%	79.1%

*not representative of traffic flow

As was the case in the preceding months, the majority of unique vehicles detected in the zone (around three quarters) were detected during CC hours. There was a slight increase in compliance rate between CC hours and 24 hour data, this indicates that vehicles entering the zone in the evening and on weekends were less likely to be older more polluting vehicles.

Table 11. Average number and proportion of unique compliant vehicles detected in the zone over a 24 hour period from April 2019 – January 2020

Month	Number of vehicles driving in the charging zone per day			Proportions of vehicles driving in the charging zone	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
April 19	121,664	32,137	89,527	26.4%	73.6%
May 19	117,289	30,146	87,144	25.7%	74.3%
June 19	118,021	29,434	88,588	24.9%	75.1%
July 19	116,082	28,562	87,520	24.6%	75.4%
August 19	108,932	25,802	83,130	23.7%	76.3%
Sept 19	116,601	27,044	89,557	23.2%	76.8%
Oct 19	114,035	26,240	87,795	23.0%	77.0%
Nov 19	116,930	26,366	90,564	22.5%	77.5%
Dec 19	113,597	25,293	88,304	22.3%	77.7%
Jan 20	106,664	22,255	84,409	20.9%	79.1%

*not representative of traffic flow

Table 11 above shows the number of unique vehicles detected in the zone and compliance rate for an average day (24 hours) from April to January 2020. For all months the 24 hour compliance rate was higher than the CC hours compliance rate.

As discussed, data before April 2019 was collected during congestion charging (CC) hours only and we are therefore unable to compare 24 hour data to a time before the ULEZ was introduced

Charge payments and penalty charges

On an average day in January 2020 around 22,255 non-compliant, unique vehicles were detected in the zone. Of these:

- Around 10,628 (48 per cent) paid the charge (2,611 ULEZ web or call centre payments, 5,142 Auto Pay payments and 2,875 ULEZ Fleet charge payments)

- Around 1,894 (9 per cent) were in contravention of the scheme and incurred a penalty charge

- Around 9,733 (44 per cent) were not required to pay the daily ULEZ charge as they are eligible for a 100% discount or exemption

Compliance by vehicle type

Table 12 shows the daily average 24 hour compliance rate in January 2020 broken down by vehicle type.

Table 12. 24hr compliance rate in January 2020 by vehicle type

Vehicle type	Number of vehicles driving in the charging zone per day			Proportions of vehicles driving in the charging zone	
	Unique vehicles detected in zone*	Non-compliant vehicles	Compliant vehicles	Non-compliant vehicles	Compliant vehicles
All Vehicles	106,664	22,255	84,409	20.9%	79.1%
Cars	78,684	13,968	64,716	17.8%	82.2%
Cars (excluding taxis)	*69,724*	*7,609*	*62,115*	*10.9%*	*89.1%*
Taxis only[6]	*8,961*	*6,359*	*2,602*	*71.0%*	*29.0%*
Vans	18,808	6,837	11,971	36.4%	63.6%
HGVs	3,376	327	3,049	9.7%	90.3%
TfL buses	1,621	0	1,621	0%	100%
Non-TfL Bus/Coach	452	105	347	23.2%	76.8%
Other *	2,847	154	2,693	5.4%	94.6%
Unknown	877	865	12	**	**

*Other vehicle category includes motorbikes, mini-buses, TfL buses and non-road going vehicles
**Unknown means vehicle type cannot be determined (e.g. foreign vehicles). These default to non-compliant unless registered.

Table 12 shows the highest compliance rate is for HGVs at 90 per cent, next is car (excluding taxis) with a compliance rate of 89 per cent. The data shows that the compliance rate for cars in general is very high, but when grouped with taxis it falls to 82 per cent.

The Mayor has taken steps to support taxi drivers in the move to cleaner cabs with TfL's Taxi Delicensing Scheme launched in 2017, with payments of up to £5,000 to retire the oldest taxis from London licensing. The scheme was enhanced in 2019 to offer top level payments of £10,000. There are now over 3,370 ZEC taxis. In late 2019 the first ever fully

[6] Taxis refers to black cabs only and does not include Private Hire Vehicles

electric London black cab, the Dynamo, was launched and just a few months later there are already 18 in circulation.

To ensure London returns to being on track to reduce emissions from taxis by 65 per cent by 2025, TfL confirmed last year that the age limit for black cabs will be reduced to 12 years for Euro 3, 4 and 5 taxis by 2022. From November 2019, the current 15-year age limit will apply to the anniversary of the date when the vehicle was licensed, with a proposed reduction in the age limit to 14 years from November 2020 and an annual reduction of one year each year until the 12-year age limit is reached. Euro 6 taxis, those converted to liquid petroleum gas (LPG) which reduces NOx emissions from taxis by over 70 per cent, and ZEC taxis will retain the 15-year age limit. TfL retains the ability to grant exemptions to the age limit requirements on a case by case basis.

For more information please see the Taxi and Private Hire pages of the TfL website.

Appendix 1: Methodology

The method for this report is the same as that included in the Appendix of the Central London ULEZ - Six Month Report.

We are grateful to Dr Gary Fuller, King's College London who kindly provided peer review support and comments on this methodology.

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From: **SM-Defra-AQ Secretariat** aq.secretariat@defra.gov.uk
Subject: Greater London Authority - AQEG CfE - 23 April
Date: 29 April 2020 at 12:39
To: aqeg@ncas.ac.uk

SS

From: Elliot Treharne [mailto:Elliot.Treharne@london.gov.uk]
Sent: 23 April 2020 16:57
To: SM-Defra-AQ Secretariat <aq.secretariat@defra.gov.uk>
Cc: Rosalind O'Driscoll <Rosalind.O'Driscoll@london.gov.uk>
Subject: COVID-19 air quality analysis and ULEZ reports

Dear Sir / Madam
Please see attached a letter to George Eustice from the Mayor of London submitting our response to the AQEG call for evidence on the COVID-19 lockdown's impacts on air quality. This shows there have beenhuge reductions in Nitrogen Dioxide since the lockdown. Central London roadside locationshave seen a fall in daily average NO2of around 40 per cent. We've done a press release explaining the results in more detail which is below.
In addition, we have also submitted a report on the latest analysis on the impact of the ULEZ before the lockdown – a reduction of 44 per cent in roadside NO_2 in the central London.This is important context to understand the impact of the COVID-19 lockdown as there had already been significant reductions. The impact of the ULEZ has been significant and it provides further evidence of how powerful a policy intervention Clean Air Zones are.
I hope you find the reports helpful and am happy to discuss if you have any questions.
Thanks,
Elliot

Elliot Treharne
Head of Air Quality, Environment
GREATER**LONDON**AUTHORITY
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elliot.treharne@london.gov.uk
My preferred pronouns are he/him

STRICTLY EMBARGOED 1600 THURSDAY 23 APRIL 2020

Dramatic improvements in air quality on London's roads

- Mayor of London responds to Government's call for evidence oncurrentchanges in air pollution during the COVID-19lockdown
- Levels of harmful gas nitrogen dioxide (NO_2) at some of London's busiest roads are on average about half what they were before lockdown
- This is in addition to thesignificant reductions delivered bypolicies including the world's first Ultra Low Emission Zone (ULEZ), whichcontributed to a 44 per centreductionin roadsideNO_2in the central zone prior to lockdown

The Mayor of London, Sadiq Khan, has today published new evidence which shows dramatic improvements in air qualityas a result of thehalvingof traffic in London due to the coronavirus lockdown.This is in response toenvironment ministers' call for evidencewhich will feedintothe Government's response to COVID-19.

TheMayor hasalreadymade huge strides in cleaning up London's air, including

TheMayor hasalreadymade huge strides in cleaning up London's air, including introducing the world's first Ultra Low Emission Zone (ULEZ) in central London. Reductions measured in recent weeks are in addition to the significantimprovementsdeliveredsince 2016.A report on these improvements has been published today to provide important context to the COVID-19 reductions.The report also confirms the effectiveness of clean air zones in tackling air pollution.

In 2020, before measures to address the COVID outbreak were introduced, hourly average levels of harmful gas nitrogen dioxide NO_2 at all monitoring sites in central London had already reduced by more than a third (35 per cent) compared to the same period in 2017. Since 16 March 2020 there has been an additional reduction of 27 per cent.

Poor air qualitystunts the growth of children's lungs and worsens chronic illness, such as asthma, lung and heart disease. There is also emerging evidence linking air pollution with an increased vulnerability to the most severe impacts of COVID-19*.

The report shows:

- There have beenhuge reductions in NO_2, especially at roadside sites. Central London roadside locationshave seen a fall in daily average $NO_2$of around 40 per cent. These reductions are in addition to those already delivered by the ULEZ.
- One of London's busiest roads, Marylebone Road, has seen a reduction in daily average $NO_2$of 48 per centand Oxford Street has seen a reduction of 47 per cent.
- Despite these improvements, London has had particulate pollutionepisodesduring lockdown. Thisexposes that London's poor air quality is not just the result of traffic pollution and further action is required on other sources, includingdomesticburningand agricultural emissions.

Evidence from theBreatheLondonair quality monitoringnetwork will also be submitted to the Department for Environment, Food and Rural Affairs (Defra), which shows similar reductions inNO_2across the city. TheBreatheLondon team haveusedWaze for Cities data to measurebigreductions in congestion.

This is part ofefforts bya number ofworldcities better to understand the impacts of COVID-19lockdown measureson air quality. London is working closely with the C40CitiesClimate Leadership Group to share information and best practice.

City Hall has also today published new data showing dramatic improvements in London's air quality across the capital since 2017.

The report reveals that the introduction of policies including the world's first ULEZ have contributed to a reduction of 44 per cent in roadside NO2 in the central London ULEZ zone**.InJanuarythere were44,100fewer polluting vehicles being driven in the central zone every day with79 per centof vehicles in the zone now meeting the ULEZ emissions standards – up from 39 per cent in February 2017***.

Around half of London's air pollution comes from road transport. Today's evidence shows how our polluted air isoftencaused by the way we choose to move around the city. Nearly half of car trips made by Londoners before the coronavirus lockdown could be cycled in around ten minutes.****

The Mayor of London, Sadiq Khan, said:"London has one of the most advanced air quality monitoring networks in the world, which has recorded how the coronavirus lockdown has dramatically improved air quality in London. But this cleaner air should not just be temporary, as Londoners deserve clean air at all times. So once the current emergency has passed and we start to recover, our challenge will be to eradicate air pollution permanently and ensure the gains we've made through policies such as ULEZ continue. It is critical that Government keeps this in mind as part of the country's recovery from the pandemic."

Senior Manager Air Quality at the EnvironmentalDefenseFund Europe, Elizabeth Fonseca, said:"People with certain serious medical conditions are at higher risk for severe illness from Covid-19, so it's critical to keep in mind the health impact of pollution, both as people are experiencing it now and long-term. Nitrogen dioxide pollution has gone down, but London recently saw huge spikes in dangerous particulate pollution. A few weeks or months' improvement of just one pollutant doesn't make lung disease and other ailments disappear."

Senior Lecturer in Air Pollution Measurement at King's College London, member of the Medical Research Council and Defra's Air Quality Expert Group, Dr Gary Fuller, said: "Breathing bad air has had an intolerable impact of Londoner's health for far too long. In our operations centre at King's we have been measuring London's air pollution for nearly 30 years. During this time we've seen deteriorations followed by a long period when some places showed slow improvement, and others slowly worsened.

For years it felt like we were at a standstill.

"But, even before the Covid lockdown, London's air pollution was undergoing a dramatic change for the better. Nitrogen dioxide in central London and along main bus routes was improving at some of the fastest rates we've ever measured. We need to remember these lessons going forward. These successes show that our city's air pollution is not an intractable problem and that actions can bring results."

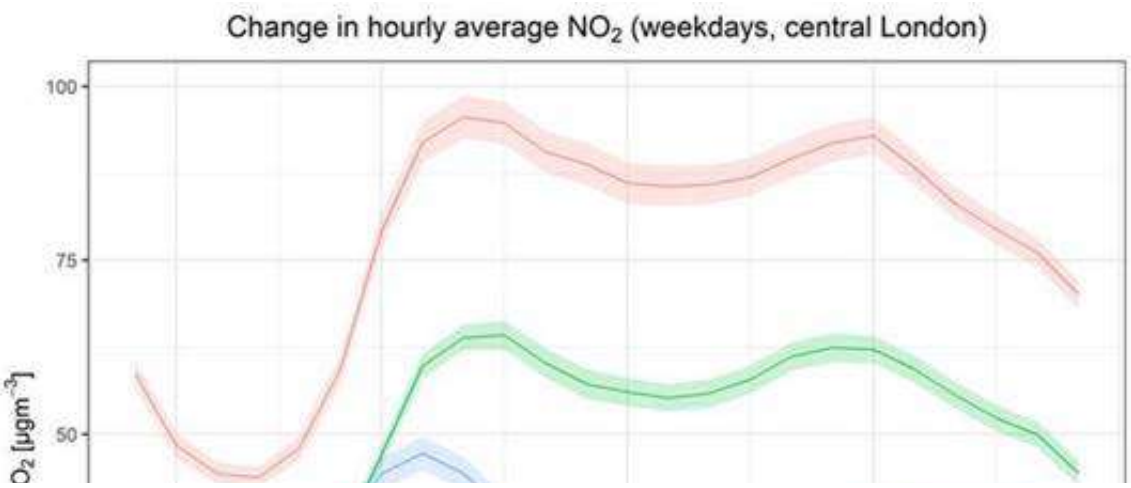
Royal College of Physician Special Adviser on Air Quality, Professor Stephen Holgate, said: "A year ago who would have believed our lifestyles would have changed so dramatically? Who would have believed it possible that the toxic air pollution in our capital city would be cut by half as a result of ULEZ and a drastic decrease in travel?

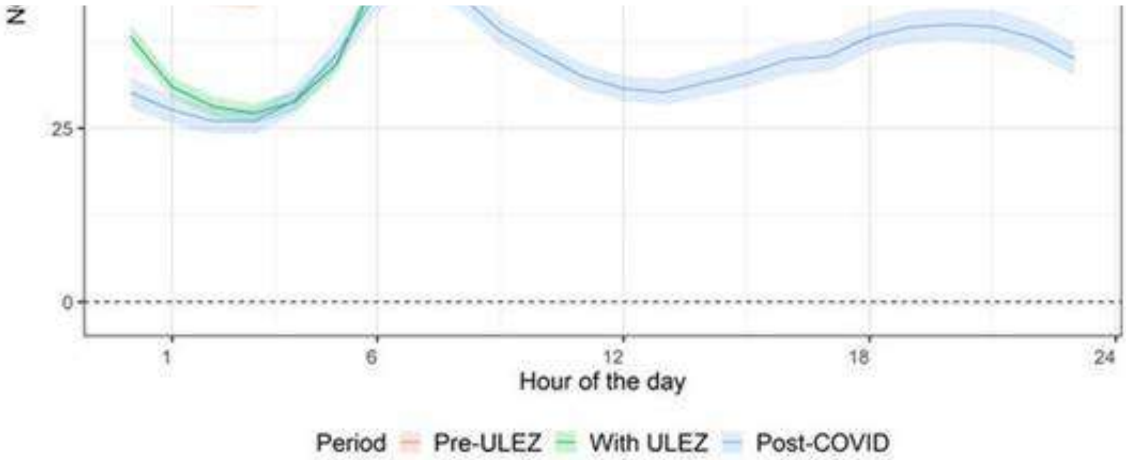
"While COVID-19 has wreaked havoc in our lives, this dreadful virus has brought the importance of outdoor space and the environment into focus. The consequences of this virus will be significant and felt for many years to come. However, as people's behaviours have changed, we have seen real improvements in air quality. We're all looking forward to the time when the lockdown is lifted, and once it does, I sincerely hope we'll be able to retain some of the new cleaner and greener habits we've developed."

ENDS

NOTES TO EDITORS

Figure 1. Change in hourly average NO2 in central London





* Harvard University research linking air pollution with an increased vulnerability to the most severe impacts of COVID-19 - https://projects.iq.harvard.edu/covid-pm Similarly, in the UK the University of Cambridge have found links between levels of air pollution and the severity of COVID-19 (https://www.medrxiv.org/content/10.1101/2020.04.16.20067405v2)
**This is measured from February 2017 to January 2020, to reflect when the Mayor publicly confirmed the Toxicity Charge (T-Charge) – the predecessor to the ULEZ - and people started to prepare for the schemes.
*** The 79 per cent compliance rates in January 2020 refers to a24 hourperiod. Before the introduction of the ULEZ24 hourcompliance data was not available so data from Congestion Charging hours was used as a proxy instead.
**** Nearly half of car trips made by Londoners before the coronavirus lockdown could be cycled in around ten minutes http://content.tfl.gov.uk/healthy-streets-for-london.pdf

&#0;. The Mayor's response to the Defra call for evidence is available here: https://www.london.gov.uk/WHAT-WE-DO/environment/environment-publications/estimation-changes-air-pollution-during-covid-19-outbreak-0
&#0;. TheULEZreport is available here:https://www.london.gov.uk/WHAT-WE-DO/environment/environment-publications/central-london-ulez-ten-month-report
&#0;. Analysis of the impact of COVID-19 measures using the Breathe London network is available: https://www.breathelondon.org/covid19
&#0;. The evidence shows that road traffic in London has reduced by around 50 per cent Londonwide since the beginning of March.
&#0;. In February 2017 the Mayor confirmed the introduction of the T-charge as a stepping-stone for the ULEZ and thiscan be seen asthe start of the accelerated change in thevehicle fleet as Londoners and businesses prepared for the new schemes and buses on routes in central London began to be upgraded to become ULEZ compliant.
&#0;. London has one of the most advanced and comprehensive network of air quality monitors. The analysis has used data from more than 100 fixed air quality monitors, alongside the Breathe London network, which has 100 state-of-the-art fixed sensor pods mounted on lampposts and buildings close to known air quality hotspots and sensitive locations such as schools and nurseries.

Maintaining air quality monitoringstationshas been designated essential workby the Environment Agency and the Greater London Authority.

NHS health information and advice about coronavirus can be found at nhs.uk/coronavirus

PDF
London_response_to_A...20.pdf

PDF
ULEZ ten month evaluat...20.pdf

PDF
Rt Hon George Eustice MP.pdf

Response to DEFRA Request for Evidence and Analysis: Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK

Prepared by Dr Doug P. Finch and Professor Paul I. Palmer, University of Edinburgh

28 April 2020

Table of Contents

Executive Summary

1 We report analyses of Defra Automatic Urban and Rural Network (**AURN**) measurements of ozone (**O_3**), nitrogen dioxide (**NO_2**), and particulate matter with a diameter smaller than 2.5 microns (**$PM_{2.5}$**). We focus our analysis on the period immediately before the pre-Covid-19 lockdown (9^{th} to 22^{nd} March 2020) and the weeks during the lockdown (23^{rd} March to 25^{th} April 2020, inclusively) and place them in context of the previous 5-year mean (2015 to 2019, inclusively).

2 We compare data from a particular week in 2020 to the corresponding weeks in previous years. This approach allows us to account for the emission change from weekdays and weekend days.

3 We find a large and consistent reduction in NO_2 across the UK during the lockdown, with a mean reduction of 13 μg m^{-3} at urban sites and of 2 μg m^{-3} at rural sites. The weeks preceding the lockdown also show that NO_2 measurements are lower than mean climatological values, however the persistency of this reduction during the lockdown suggest that lower NO_2 values at urban sites are also due to reduced emissions. Only during the lockdown are NO_2 concentrations lower than the spread of values observed in the previous five years.

4 We find that O_3 concentrations during the lockdown are generally higher than mean observed values from the preceding five years. This is consistent with reductions in NO_2 concentrations within a VOC-limited photochemical environment. However, O_3 concentrations rarely exceed the highest values found in the previous five years. Measurement sites in London and Cardiff have recorded O_3 values that exceed national air quality limits.

5 $PM_{2.5}$ measurements have been highly variable across the lockdown period, exhibiting broadly similar temporal patterns across the UK. We show this coherency in measurements is at least partly related to changes in the prevailing wind direction.

6 Our analyses use an online tool: www.ukatmosphere.org. This tool allows any user to easily extend our reported period beyond 25^{th} April 2020 and also to extend our analyses to other surface air pollutants.

UK 2015-2019 Mean Baseline Values

7. Figure 1 shows the daily mean concentrations for O_3, NO_2, and $PM_{2.5}$ for weeks 11 to 17 averaged over 2015 to 2019 at six urban AURN sites, and the mean of all UK urban and rural AURN sites. The six urban sites include those located in London, Glasgow, Manchester, Cardiff, Newcastle, and Belfast, which represent a geographical spread across the UK. We use all 22 rural and all 125 urban (traffic and background) AURN sites for the nationwide mean values.
8. We compare daily values from each week in 2020 to the corresponding weeks in previous years. This allows us to avoid any errors from comparing weekdays in 2020 with weekend days in previous years. We use climatological values to put values observed in 2020 in a wider temporal context. We use the previous 5-year mean as our climatology to minimize an impact on pollutant trends on our statistical analyses of data collected in 2020.
9. Differences between rural and urban baseline values are as expected, associated with differences in emission sources and associated atmospheric chemistry. Urban NO_2 has a mean value of approximately 30 μg m^{-3} which is 20 μg m^{-3} higher than mean rural values. Over our study period, surface O_3 increased by approximately 10 μg m^{-3} at urban (~45—55 μg m^{-3}) and rural (~70—80 μg m^{-3}) sites. An increase of O_3 is expected at this time of year due to the longer day lengths with the onset of Spring. $PM_{2.5}$ is the most variable of the three compounds we studied. Mean values for urban sites range ~10—15 μg m^{-3}, peaking with values over 20 μg m^{-3}. Rural sites follow the same pattern but with mean values ~5—10 μg m^{-3} lower than urban sites.

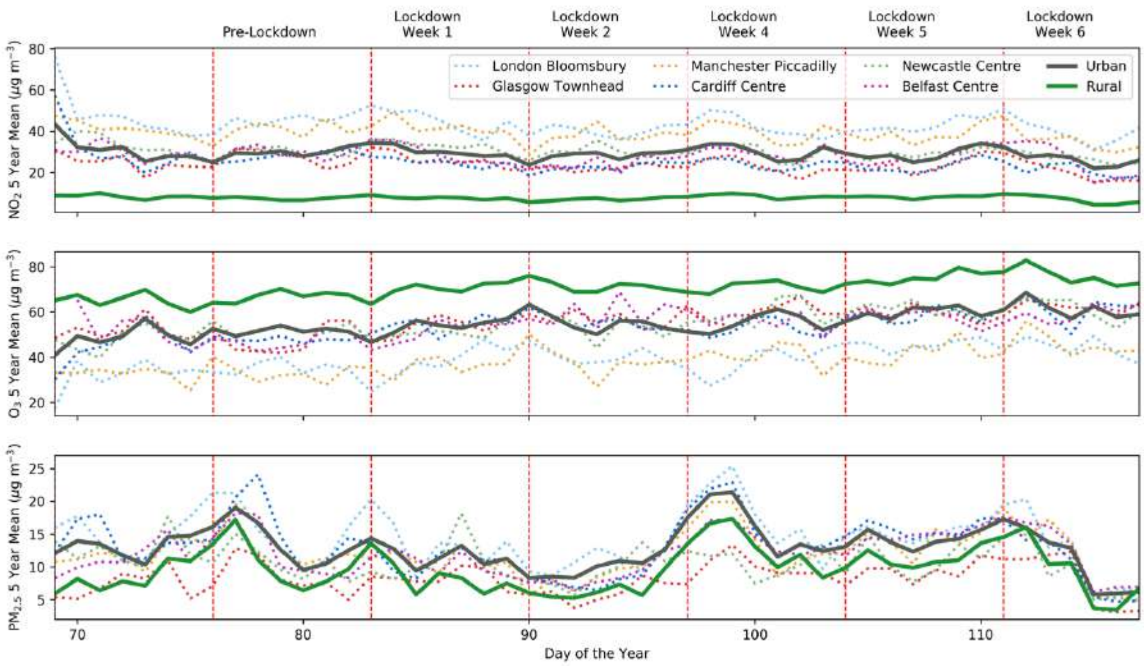


Figure 1 Daily mean concentrations (μg m^{-3}) for NO_2 (top), O_3 (middle) and $PM_{2.5}$ (bottom) for weeks 11 to 17 averaged over 2015 – 2019 for six urban centres (dotted lines), and the mean of all urban sites (solid grey) and all rural sites (solid green) in the AURN. The week lockdown labels denote the equivalent timing of the lockdown in 2020.

10. Mean values of NO_2 during the pre-lockdown and Covid-19 lockdown periods are 8 µg m^{-3} and 6 µg m^{-3}, respectively, at 22 rural AURN sites, and 29 µg m^{-3} and 16 µg m^{-3}, respectively, at 125 urban AURN sites. These changes represent a decrease of 13 µg m^{-3} over urban areas and a decrease of 2 µg m^{-3} over rural area compared with 5-year climatological values. This is consistent with a larger reduction in traffic volumes in urban areas.
11. Figure 2 shows the UK distribution of NO_2, O_3 and $PM_{2.5}$ reductions, relative to the 2015—2019 climatology, during the Covid-19 lockdown period.
12. We find that NO_2 generally decreases across all urban AURN sites, with the exceptions of Canterbury, Eastbourne, Newport and York Bootham. NO_2 generally shows a decrease across all rural AURN sites, with the exceptions of Charlton Mackrell (Somerset), High Muffles (Yorkshire), Narbeth (Pembrokeshire) and Yarner Wood (Dartmoor). The largest nationwide change in mean NO_2 during the lockdown period was observed at London Marylebone Road with a reduction of 54 µg m^{-3} (64%), from 84 µg m^{-3} to 30 µg m^{-3}.
13. We find that O_3 generally increases across the UK, with the exceptions of a number of sites in Scotland, Charlton Mackrell (Somerset), High Muffles (Yorkshire) and Derry Rosemount (N. Ireland). Mean O_3 at urban sites increased from 57 µg m^{-3} to 66 µg m^{-3} and at rural sites increased by only 74 to 75 µg m^{-3}. Aberdeen saw the largest decrease of O_3 over the UK with a drop of 26 µg m^{-3} (65 to 39 µg m^{-3}) compared to the previous five years. London Marylebone Road saw the largest increase in O_3 of 29 µg m^{-3} (26 to 55 µg m^{-3}).
14. Concurrent changes in NO_2 and O_3 are consistent with a VOC-limited photochemical environment, as expected.
15. We find that $PM_{2.5}$ is more spatially variable than the other pollutants we studied. Mean $PM_{2.5}$ increased in both urban and rural environments by only 1 µg m^{-3} and 2 µg m^{-3}, respectively. As we shown in our Appendix, variations in $PM_{2.5}$ appears to be strongly related to changes in meteorology. Changes in $PM_{2.5}$ over Scotland and Northern Ireland are small. Data are much sparser in these countries and it is therefore difficult to draw further conclusions.

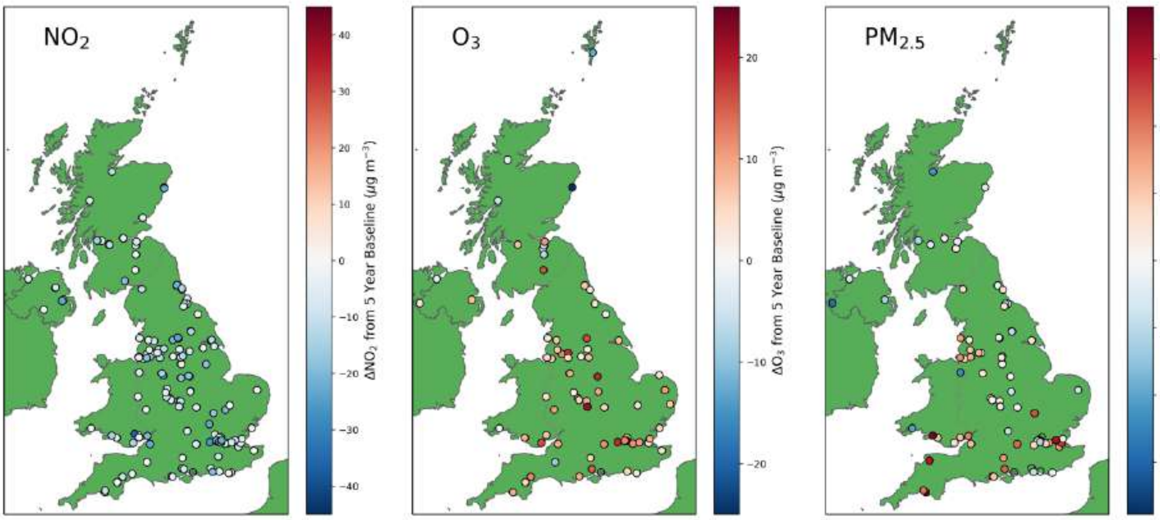


Figure 2 Spatial distributions of NO_2 (left), O_3 (middle) and $PM_{2.5}$ (right) reductions (μg m^{-3}) across the UK during the Covid-19 lockdown period relative to the 2015—2019 climatology.

City Pollutant Heatmaps

16. We show heatmaps that clearly show the changes in NO_2, O_3 and $PM_{2.5}$ at individual AURN sites across the UK during pre-lockdown and lockdown periods compared to 5-year climatological values (Figure 1). Climatological values are linked to values in 2020 by their week numbers (Paragraph 8).
17. Figure 3 shows example heatmaps for sites also shown in Figure 1: London Bloomsbury, Glasgow Townhead, Manchester Piccadilly, Cardiff Centre, Newcastle Centre, Belfast Centre, and also the mean values for urban and rural sites across the UK. A separate plot shows sites across London (Figure 4) because changes at those locations are generally much larger than other sites over the UK.

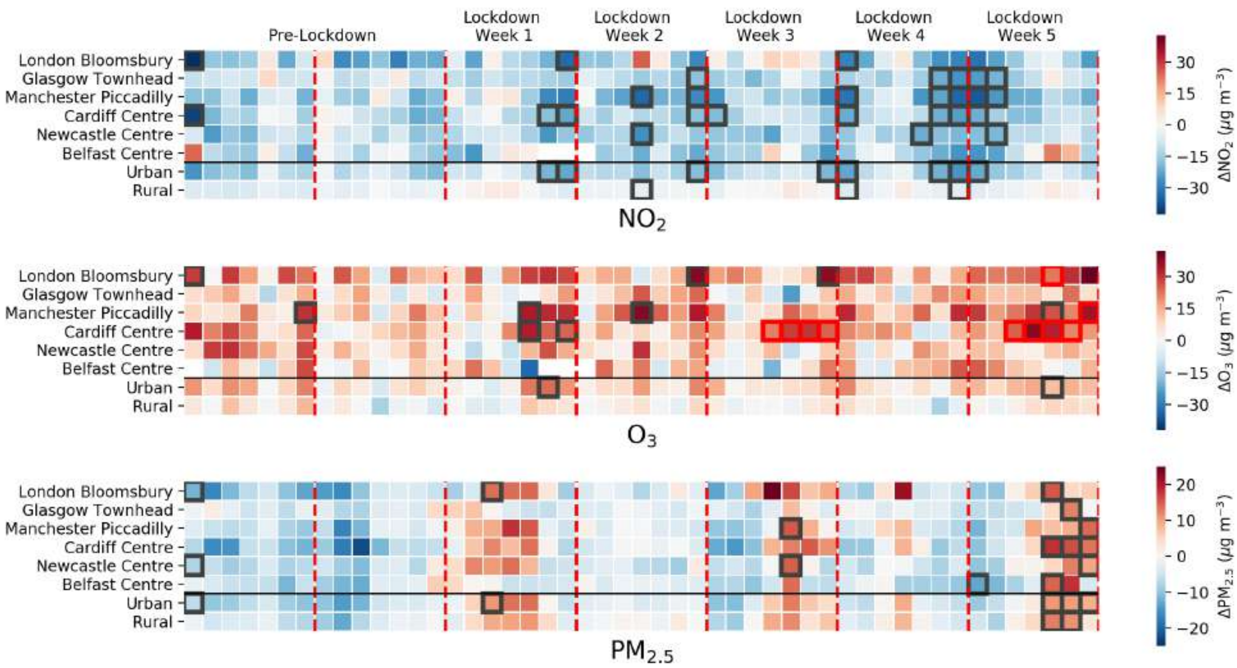


Figure 3 Heatmap showing changes (μg m^{-3}) in NO_2 (top), O_3 (middle) and $PM_{2.5}$ (bottom) from 5-year baseline (Figure 1) at six urban centres across the UK and for all urban and rural AURN sites. Grey boxes highlight days which fall outside the 5-year range and red boxes highlight days when O_3 values exceed the national air quality limit.

18. Days when concentrations are above or below the spread of 5-year climatological values are denoted by a grey box.
19. Days when concentrations of O_3 exceed the UK regulatory limit (daily maximum of a rolling 8-hour mean more than 100 μg m^{-3}) are highlighted by a red box. Similar criteria are less easy to apply with NO_2 and $PM_{2.5}$ by virtue of how their regulatory limits are defined[1].
20. The heatmap for NO_2 shows concentrations have been consistently lower than the 5-year climatological values from pre-lockdown weeks to week 5 of the lockdown. Daily values only become significantly smaller than the climatological spread of values in week 2 of the lockdown and only intermittently up to week 5 of the lockdown.
21. The heatmap for O_3 has steadily increased since the lockdown begun. We find that fewer days lie outside the spread of climatological values than we find for NO_2. This implies that we

[1] The national air quality objectives state that the annual mean $PM_{2.5}$ is not to exceed 25 μg m^{-3} (except for Scotland where it is not to exceed 10 μg m^{-3}) and the annual mean NO_2 is not to exceed 40 μg m^{-3} or a 1 hour mean to exceed 200 μg m^{-3}.

are generally observing O_3 values higher than the 5-year mean climatology but mostly within the range of values observed in the previous 5 years.

22. We find a number of sites exceed regulatory O_3 standard during the lockdown period: Cardiff (8 days), London Bloomsbury (1 day), and Manchester Piccadilly (1 day). Exceedances are more likely to happen during Spring months when O_3 increases but the exceedances during the lockdown lie above climatological mean values.
23. Previous studies have highlighted the role of meteorology in changing surface air pollutants. The duration and photochemical consistency between changes in NO_2 and O_3 suggest that these changes are also due to reduced emissions.
24. $PM_{2.5}$ shows a different pattern of variability than NO_2 or O_3, with some weeks higher and some weeks lower than the 5-year climatological mean values. Weeks one, three and five of the Covid-19 lockdown show higher $PM_{2.5}$ across all sites, while weeks two and four (and the preceding two weeks) of the lockdown show lower $PM_{2.5}$ than 5-year climatological mean values. We show in Figure 5 these large nationwide swings in $PM_{2.5}$ covary with mesoscale changes in weather patterns.
25. If we assume the main change in emissions over the lockdown period is associated with the decrease in urban traffic volume, we infer that $PM_{2.5}$ variations may not be as dependent on changes in traffic emissions as NO_2 and O_3.
26. Figure 4 shows the five AURN sites across London: Marylebone Road, Bloomsbury, N. Kensington, Haringey Priory Park South, and Hillingdon. Marylebone Road shows the largest UK decrease in NO_2 of up to 80 μg m^{-3}. This is an urban traffic site and has a higher 5-year climatological mean than the other sites in London. Intermittent exceedances in O_3 occur synchronously across London but unexpectedly these do not occur when there are the largest reductions in NO_2. This suggest that these sites have experienced O_3 exceedances on these dates in the previous five years so that O_3 measurement levels during the lockdown are not unusual. Variations of PM2.5 are consistent with those with sites across the UK (Figure 3).

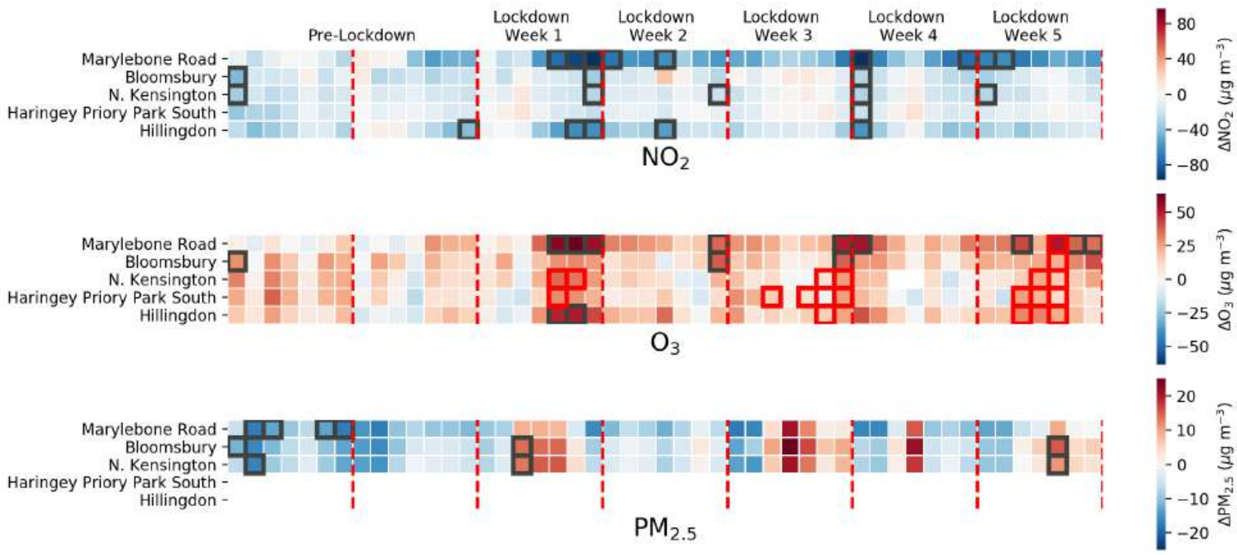


Figure 4 Same as Figure 3 but for sites across London.

27. Figure 5 shows daily mean variations in $PM_{2.5}$ and corresponding weekly wind roses for London Bloomsbury. Manchester Piccadilly, and Leeds Centre during the lockdown compared to the previous 5-year climatological mean (Paragraph 8). Plots for every AURN site measuring $PM_{2.5}$ as well as similar plots for NO_2 and O_3 can be found at https://datasync.ed.ac.uk/index.php/s/8r3ImHzU6NIfZM0 [2]
28. Wind rose plots show the distribution of daily mean wind direction and wind speed for pre- and post-Covid-19 lockdown. These numerical wind data are from model and provided by Ricardo plc.
29. All three sites show a peak in $PM_{2.5}$ during the first week of the lockdown and to different extents during weeks 4-6 of the lockdown. This is consistent with the heatmaps (Figure 3 and Figure 4). The wind roses show a clear change in the predominant wind direction during the first week of the lockdown (from westerly to north easterly), coinciding with elevated $PM_{2.5}$. This behaviour is seen across many other sites across the UK, suggesting that the elevated $PM_{2.5}$ was due to incoming air from the north east. Further work is needed to identify the origin of this elevated $PM_{2.5}$.
30. The wind direction during week four of the lockdown is less clear, however the predominant wind speed is lower than other weeks (generally below 4 ms^{-1}) indicating settled weather conditions that are associated with high levels of pollution (although this is not seen in NO_2 or O_3 concentrations).
31. Another explanation for the higher levels of PM2.5 could be an increase in garden bonfires and barbecues due to warmer weather and the lockdown giving more time to people to do activities such as gardening. However, reports of this occurring are anecdotal and composition analysis of aerosols would need to be performed to support this hypothesis.

[2] To access the plot of the AURN data use the password ‘covid19’

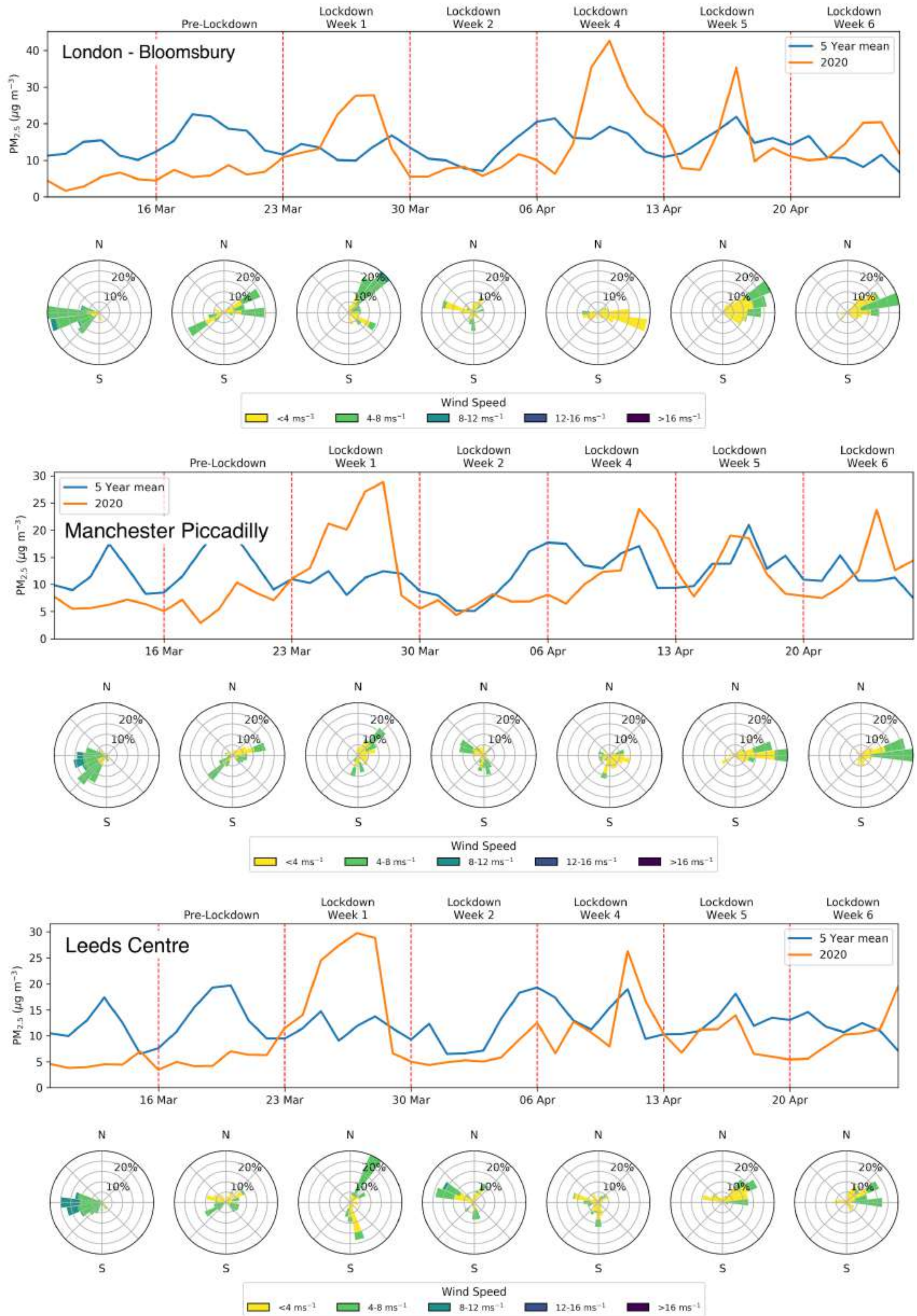


Figure 5 Daily mean $PM_{2.5}$ during the lockdown (orange) and the 5-year climatological mean (blue) and the corresponding weekly wind rose for London Bloomsbury (top), Manchester Piccadilly (middle) and Leeds Centre (bottom).

Response to Defra call for evidence "*Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK*" on point "*How might altered emissions of air pollutants over the next three months affect UK summertime air quality?*"

Understanding the impact of COVID-19 restrictions on air quality pollutants with a mixed atmospheric chemistry transport model / machine learning approach.

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Summary
We have coupled the NASA GEOS-5 Composition Forecast (GEOS-CF) Near Real-Time (NRT) global air quality system with local observations of air pollution through the use of a machine-learning bias correction approach to evaluate the change in air pollution concentrations due to COVID-19 restrictions. By combining model analyses and the equivalent observations, our method calculates "business as usual" concentrations of air pollutants at observational sites during the COVID-19 restrictions, that can take into account long-range transport of pollutants, local conditions and the weather. This is important to consider as the period of restriction has, thus far, been meteorologically unusual.

We have focused on 21 AURN sites around the South East of England and find:

- strong evidence that **NO_2 concentrations have dropped** due to the restrictions. Of the 21 sites, 20 had NO_2 measurements. 18 of these sites showed reduced NO_2 concentrations with 2 showing no change. On average a reduction of 5 ppbv (10 μg m^{-3}) was observed, which amounts to roughly 30%. Larger reductions (~40%) were seen at traffic sites.
- evidence that **PM2.5 concentrations reduced** during the restrictions by on average 3 μgm^{-3} (~20%) at the 12 sites reporting. Larger reductions of 27% were seen at some traffic sites. One of the sites showed increases.
- strong evidence of **increases in O_3** concentrations, as a result of the restrictions. Of 9 sites measuring O_3 all showed increases with an average increase of 7 ppbv (14 μgm^{-3}) or ~30%. Some sites showed much larger increases of up to 80%.
- mixed evidence of changes in **O_x (NO_2+O_3)** concentrations. Of the 21 sites, 9 measure both NO_2 and O_3. 8 sites showed increased O_x concentrations and 1 site showed decreases. On average O_x only increased by only 1.4 ppbv (3%) potentially indicating limited increases in photochemical production of O_3 but extensive re-partitioning of NO_2 and O_3 between sites. Some sites showed increases in O_x (8 ppb or 25%) indicating some photochemical production.

We interpret the **reduction in NO_2** to be due to reduced emissions of traffic NO_x. We find at traffic dominated sites (i.e. Marylebone Road) a roughly 50% (13 ppbv) reduction in NO_2 over what would have been expected without the COVID restriction. We also find a smaller decrease in PM which again likely reflects changes in traffic emissions. Given the complicated sources of PM and the complexity of the sulfate/nitrate/ammonia system further work will be needed to disentangle the response here.

We interpret the **increased O_3** as being predominantly due to an increased repartitioning of O_x into O_3 and away fro NO_2 with lower NO emissions. There is some evidence of increased O_x concentrations which might indicate increased chemical production due to the lower NO_x concentration in some regions. However, the evidence for this is weak and would require further evaluation.

Introduction

Understanding whether there has been a change in the concentration of air pollutants due to the COVID-19 restrictions which occurred in the UK after the 23rd March 2020 (informally on the 16th) is complicated. The AURN observations are still being made and a number direct comparisons with the data are possible to explore whether the concentrations are different between say the periods 16th Feb 2020-16th March 2020 and 16th March 2020-16th April 2020 or the period 16th March 2019-30th April 2019 and 16th March 2020-39th April 2020. However, these approaches are complicated by the changing nature of the atmosphere. Directly comparing concentrations within the same year ignores seasonal changes in emissions, temperatures and solar radiation. Comparing concentration between years may remove this seasonal change but does not account for different synoptic weather conditions that occur between different years. For instance, April 2020 appears to have been especially sunny, whereas it was not in 2019. Both of these direct comparison approaches will not take this into account and may inadvertently attribute a difference in concentration to the COVID-19 restrictions rather than the differing weather or seasonal patterns.

In order to understand the impact of the COVID-19 restrictions, these differences in seasonal and meteorological conditions need to be taken into account. If driven by 'real' assimilated meteorological information, Chemistry Transport Models (CTM) are able to take these influences into account (to some extent). However, they show biases due to uncertainties in the emissions of pollutants, chemical mechanisms, physical parameterizations etc. They also rely on a spatial resolution (the size of the grid boxes) which may be orders of magnitude different from the spatial resolution pollutants are measured at. Differences between measurements and observations may thus be due to the model's inability to represent sub-grid phenomena such as nearby roads or industry or the local building environment. Thus, it would be inadvisable to evaluate the impact of COVID-19 restrictions solely by comparing the difference between a model prediction of the business as usual scenario, and observations.

What is required is a method of being able to remove the bias from the model prediction by using the long-term observations of air pollution at measurement sites. That could then be used to create a "business as usual" time-series for the restriction period which would take long-range transport and other meteorological factors into account. This could then be compared to the observational data to evaluate the impact of the restrictions. Based on previous work with machine learning [Keller and Evans, 2019, Sherwen et al., 2019], our recent paper [Ivatt and Evans, 2019] (currently in peer review) provides a framework for achieving this using a machine learning approach.

Here we use Near Real Time predictions of air pollution from the NASA GEOS Composition Forecast model (GEOS-CF) for a number of AURN sites in the South East of the UK, together with a machine learning-based bias correction to calculate what the concentrations of pollutants would most likely have been during the period 16th March 2020 to 25 April 2020 (the method is described in the Appendix). We then compare these predictions to what was actually measured to be able to produce an assessment of the impact of the COVID-19 restrictions on air pollutants.

Methodology

For 21 AURN sites distributed around the South East of the UK, output from the NASA GEOS-CF system was combined with the observational data over the period of 1st Jan 2018 to 31st Dec 2019. An algorithm (see appendix) was trained to predict the difference between the model and the measurements (the bias) given a range of environmental variables (temperature, humidity, wind speed, chemical conditions, model emissions, etc.). This allows us to effectively correct the model for recurring mismatches between the model and observations that can be explained by external factors, such as meteorology, local emission sources, localized ozone titration etc.. We then continued the bias correction beyond the 31st Dec 2019 training period up to the present day. Between 1st Jan and 25th April 2020, the system is making its best guess at the concentration of pollutants assuming 'business as usual' but taking into account changes in meteorology, long-range transport etc. Differences between these predictions and the observations thus represent the impact of the COVID-19 restrictions.

In the following sections, we briefly describe the GEOS-CF model, provide an example of the methodology at two sites for NO_2 and then summarize the results. More details of the machine learning-based bias correction are provided in Appendix 1. Appendix 2 provides assessments of the differences between the model calculations and observations for all of the sites and all species.

NASA Goddard Earth Observing System (GEOS) Composition Forecast (CF) system

The NASA Goddard Earth Observing System (GEOS) provides both forecasts of and a real-time assessment of the state (meteorological and chemical) of the atmosphere, primarily to support the NASA Earth-observing programmes. Embedded within the GEOS Earth System Model, the Composition Forecasting (CF) system has a spatial resolution of 25km x 25km and so provides a global air-quality simulation, capable of exploring regional air quality issues. The gas and aerosol phase chemical components come from the open-source community atmospheric chemistry model GEOS-Chem (www.geos-chem.org). The anthropogenic emissions are simulated based on a bottom-up emissions inventory (HTAP v2.2), scaled to the year 2017 based on OMI satellite data and imposing diurnal and day-of-the-week scale factors to the monthly average base emissions.

The model provides daily, open-access 5-day air quality forecasts around the world (a graphical representation of the London forecasts are available here with the data from the forecasts publically available here) but also near-real-time (NRT) analyse that use assimilated meteorology to provide a 'best guess' of the state of the atmosphere. It is these NRT analysis data that has been used here.

Example usage

Figure 1 shows the GEOS-CF NRT NO_2 simulation at the AURN North Kensington site (NRT cyan), together with the observations (OpenAQ black). The model generally follows the observed pattern but misses some of the high and low events. This could be due to local features not represented by the model (urban canyons, location of buildings, large local sources), or due to errors in the average emissions in that grid box, errors in the meteorology etc. However, many of these biases are explicable (it is unlikely to be instrumental or model 'noise'). The dark blue line (NRT+ML) shows the results of training a machine learning algorithm on a sub-sample of the data between 1st Jan 2018 and 31st Dec 2019, to predict the model bias and then removing that bias from the model signal.

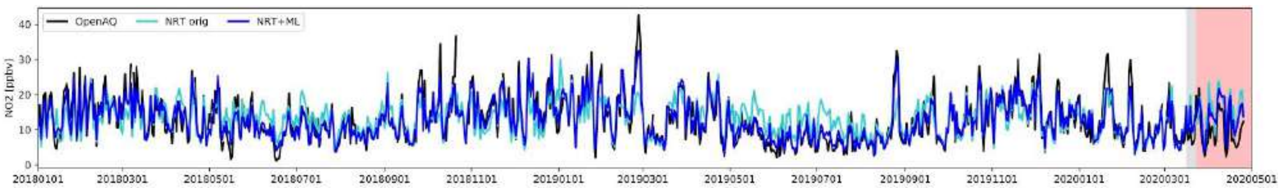


Figure 1. *NO_2 observed (black), modelled by the standard model (cyan) and modelled by the standard model together with the machine learning algorithm (dark blue) for North Kensington. The grey and red shaded area indicate the date of UK social distancing measures (16th March) and UK lockdown (23rd March), respectively. Daily mean data is shown.*

The bias correction of the NRT (Dark blue NRT+ML in the figure) data removes a significant fraction of difference between the base and the observations. While there is a clear improvement in the simulated concentrations after applying this bias correction, the base model already does a good job of simulating the observations at this site. This is reflective of the urban background nature of the North Kensington monitoring site, which is representative of the model grid resolution of 25 km^2. For other sites, this is not true.

For instance, **Figure 2** shows the comparison between the NRT model and measurements for NO_2 at the Marylebone Road site. Here the base model (cyan NRT) does a poor job at simulating the observed NO_2 concentration (black openAQ). This likely reflects the location of the site (with heavy local traffic) not being representative of the model grid. Applying the bias correction to the model removes the bias from the model prediction and the bias-corrected model follows the observations much more closely (dark blue line NRT+ML). The influence of the COVID-19 restrictions is now evident with observations falling sharply away from the bias-corrected version of the model (NRT+ML) for dates after the COVID-19 lockdown restrictions were applied. The model, on the other hand, does not show an equivalent drop in concentrations as the underlying model emissions have not been updated. In other words, the model simulates the business-as-usual scenario.

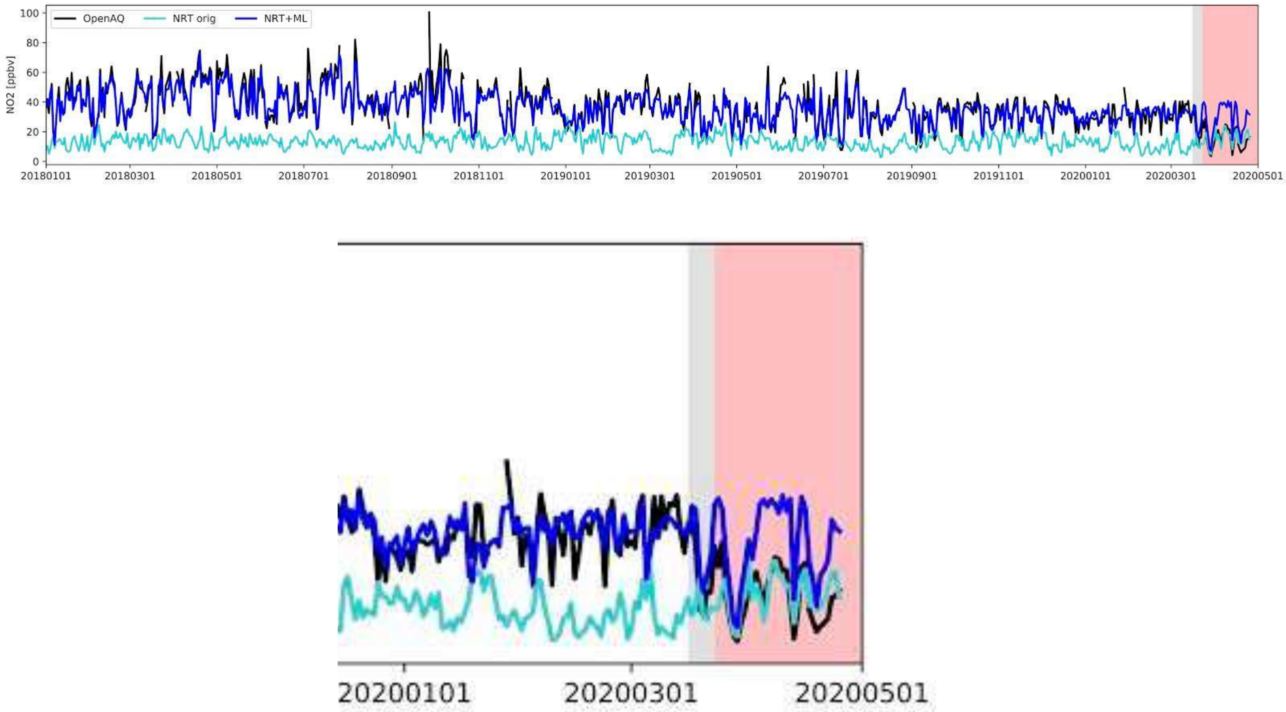


Figure 2. *NO_2 observed (black, openAQ), modelled by the standard model (cyan, NRT) and modelled by the standard model together with the machine learning algorithm (dark blue, NRT+ML) for Marylebone Road. The grey and red shaded area indicate the date of UK social distancing measures (16th March) and UK lockdown (23rd March), respectively. Daily mean data is shown. Upper panel shows the whole of the time series. The lower panel shows data after the beginning of 2020. The difference between the observational line (black) and the blue model (NRT+ML) represents the impact of the COVID restrictions.*

Differences between the corrected model (NRT+ML) and the measurements for the CODID-19 restriction period indicate the impact of fundamental changes to the environment caused by COVID-19 restrictions. **Figure 3** shows the differences between the machine learning corrected model and the observation ("the residual") since Jan 1, 2018. Positive differences (shown in red) indicate that the observations were higher than the NRT+ML model, and negative differences (blue) denote times where the NRT+ML model prediction was lower than the actual observation. Grey values indicate the residual between the base NRT model and the observations. From 1st Jan 2018 up until social distancing measures were being recommended on 16th March 2020, the model-observation residuals are relatively small and lack cohesion over time. After social distancing was implemented (16th of March), and especially after the official "lockdown" on the 23rd of March, the NRT+ML model consistently overpredicts NO_2 for the Marylebone Road site by up to 20 ppbv or more than 50%.

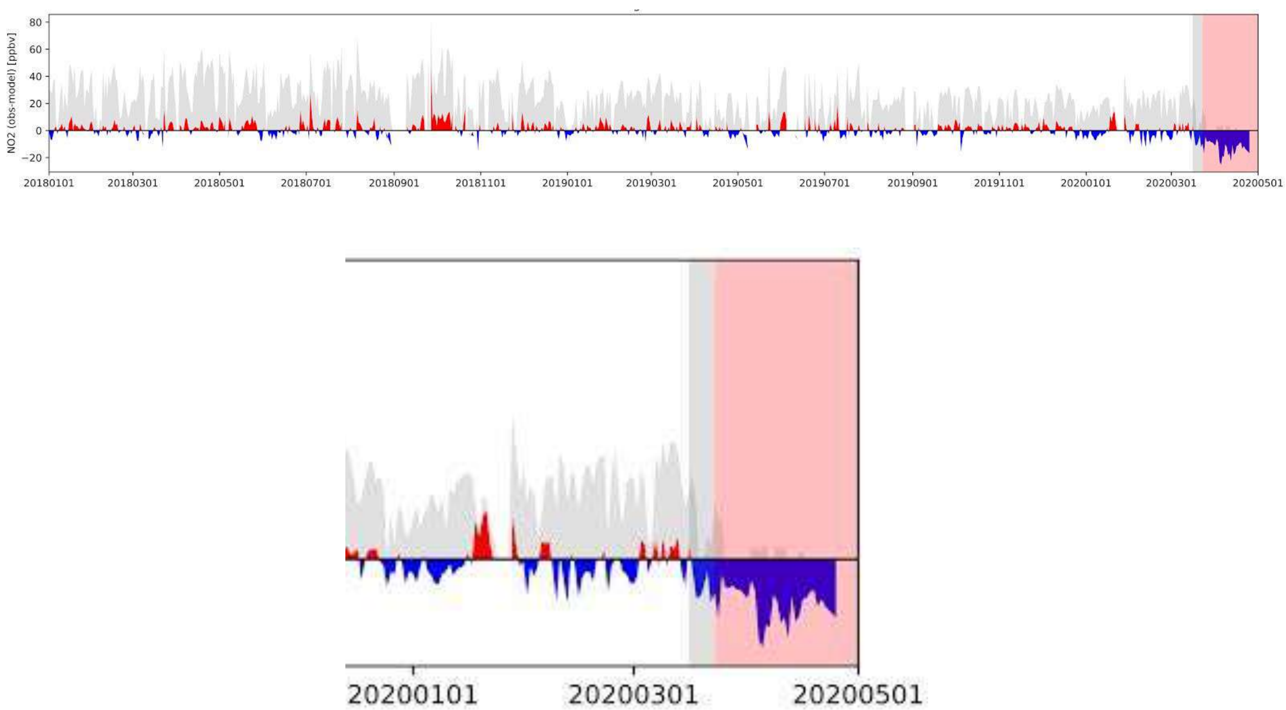


Figure 3. *Differences between measured NO_2 (black line in figure 2) and that simulated by the combined model and machine learning prediction (blue line in figure 2) at Marylebone Road. Negative (blue) colours indicate that observed concentrations were lower than predicted by the model (assuming a business-as-usual scenario), and red areas indicate times where the observations were higher than the model. Grey areas indicate differences between the measured NO_2 (black line in Figure 2) and the base model (without the machine learning) (cyan line in Figure 2). Daily mean data is shown. Upper panel shows the whole of the time series. The lower panel shows data after the beginning of 2020. The blue period after March 16th represents the influence of the COVID restrictions on NO_2 concentrations at the site.*

We have explored the impact of the COVID-19 restrictions at 21 of the Defra AURN sites around the South East of the UK. We have compared the differences between the model and the measurements after the start of social distancing on the 16th of March until April 25 with the difference between the model and the observations for the equivalent time periods in 2018 and 2019. We have analysed the results for NO_2, PM2.5, O_3 and O_x (NO_2+O_3). Not all the sites report all of the species.

Results

NO_2

Figure 4 shows a statistical description of the difference between the observed concentrations of NO_2 and that predicted by the bias-corrected model (NRT-ML) for COVID restricted period (16th March 2020 to 25th April 2020) in the dark green colour, and for the equivalent periods in 2018 and 2019 (16th March 2018 to 25th April 2018 and 16th March 2019 to 25th April 2019) in the light green colour. For the 2018 and 2019 periods, the bias-corrected model agrees closely with the observations (the median line is close to 0 for most sites). This is not surprising as the bias correction has used this data in its training. However, for the 2020 COVID restriction period, there are large differences between the predictions of the model and the observations. The observations appear systematically lower than those predicted by the model. Across all of the sites, the model predicts NO_2 concentrations higher by 5 ppbv (16.2 ppbv vs the 11.2 ppbv observed numerical values are on the plot), or approximately a 30% drop. At urban traffic sites such as Marylebone Road, Hillingdon, and Camden Kerbside, our results indicate that the reduction in traffic caused a drop NO_2 in the range of ~45-30%.

It should be noted that for many locations, the NO_2 decrease in the observations is somewhat masked by external factors, most importantly the unusually dry and warm weather. For example, for Marylebone Road, the average observed NO_2 concentration in the period 2018 and 2019 periods (16th March to 25th April) was 41.8 ppbv. This falls to 16.3 ppbv for the 2020 period, a 60% reduction. However, the NRT+ML model predicts only 29 ppbv of NO_2 for the period. This likely reflects increased photolysis and OH concentrations during the sunny weather which would naturally have reduced NO_2 concentrations. Thus overall we estimate only a 45% reduction in NO_2 concentrations at the site once these features are taken into account. At most locations, the model predicts lower NO_2 concentrations than the observed 2018-2019 mean for the period would suggest.

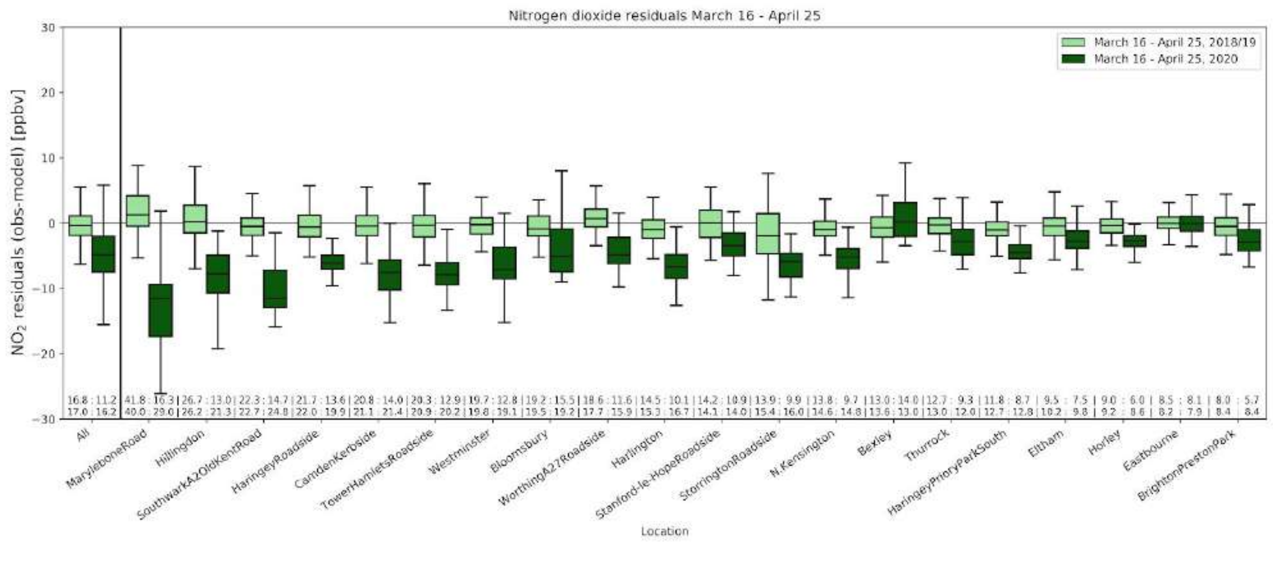


Figure 4. *Distribution of observation minus model residuals of nitrogen dioxide (NO_2) from 16th March to 25th April for a number of AURN sites in the SE of England. Dark green boxplots show the difference between observed and model-predicted NO_2 (NRT+ML) for the COVID-19 restriction period (16th March to 25th April 2020). Light green boxplots show the difference between observed and predicted (NRT+ML) for the period March 16 to April 2 for 2018 and 2019. Dark green values indicate the impact of the COVID-19 restrictions. Coloured bars indicate the 10-90% percentile, black line within the bar given the median change, error bars indicate the 5-95% percentile change. The average for all sites is shown in the leftmost boxplot. The numbers at the bottom of the figure indicate the mean concentrations as observed (upper row) and modelled (lower row), for 2018,2019 (left) and 2020 (right). Stations are ordered by pre-COVID19 average NO_2 pollution concentrations.*

O_3

Figure 5 shows the results for the 9 sites that reported O_3 concentrations. The dark purple bars represent the difference between the observed concentrations of O_3 and that predicted by the bias-corrected model (NRT+ML) for the period before the restrictions from March 16th to April 25th 2020. The light purple represents the difference in the model for the equivalent periods in 2018 and 2019. On average O_3 observations were 7 ppbv higher (~30%) than what might have been expected had the restrictions not been in place. Some sites (Thurrock and Marylebone Road) showed larger increases. The model is predicting marginally (1 ppbv) higher O_3 concentration in the 2020 period than the equivalent 2018,2019 periods which likely reflects the increased photochemistry occurring during the sunny period. However, significantly higher concentrations than that were observed suggesting a generalized increase in O3 during the period. Some sites show larger increases than others.

O_3 is a secondary pollutant, and so there is then a question as to the mechanism for this increase. Is it a change in its photochemical production (through increases in the reaction of peroxy radicals with NO) or is it a re-partitioning of O_x between NO_2 and O_3 due to lower NO emissions. Exploring the change in the O_x ($=NO_2+O_3$) concentrations over London could help answer that question. A re-partitioning would not increase O_x concentrations, whereas photochemical production would.

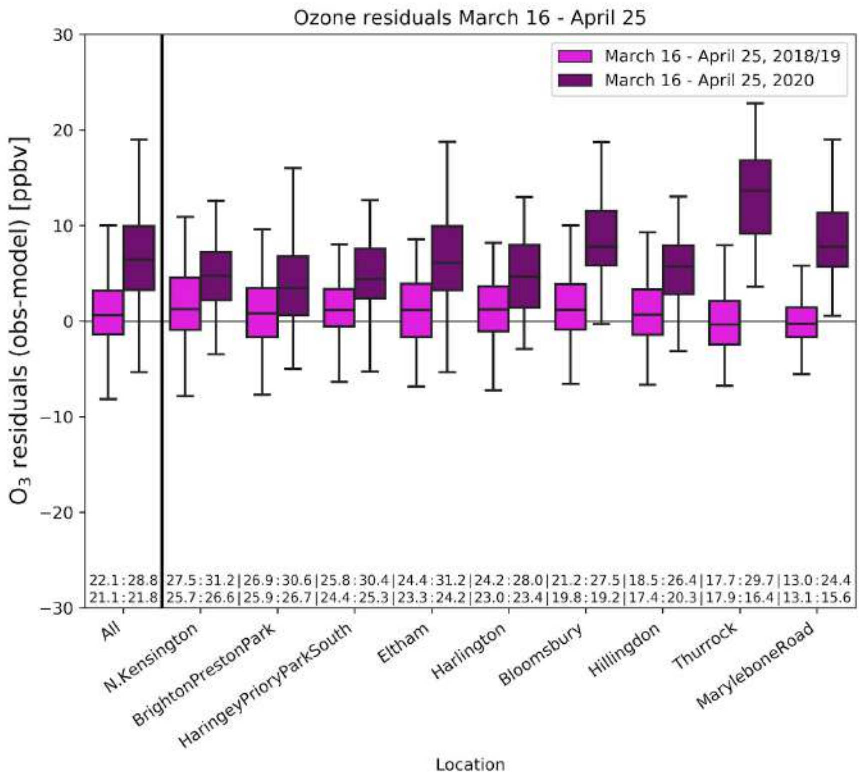


Figure 5. *Distribution of observation minus model residuals of ozone (O_3) from 16th March to 25th April for a number of AURN sites in the SE of England. Dark green boxplots show the difference between observed and model-predicted O_3 (NRT+ML) for the COVID-19 restriction period (16th March to 25th Apri, 2020). Light green boxplots show the difference between observed and predicted (NRT+ML) for the period March 16 to April 2 for 2018 and 2019. Dark green values indicate the impact of COVID-19 restrictions. Coloured bars indicate the 10-90% percentile, black line within the bar given the median change, error bars indicate the 5-95% percentile change. The average for all sites is shown in the leftmost boxplot. The numbers at the bottom of the figure indicate the mean concentrations as observed (upper row) and modelled (lower row), for 2018,2019 (left) and 2020 (right). Stations are ordered by pre-COVID19 average O_3 pollution concentrations.*

O_x

Figure 6 shows the difference between the observed concentrations of O_x (NO_2+O_3) and that predicted by the bias-corrected model (NRT+ML) for the 16th of March to 25th April for the 2018 and 2019 period (light purple) and then for the 2020 COVID restriction period (dark purple) for the sites that report both NO_2 and O_3 concentrations. There is little evidence for an increase in O_x. The average O_x is predicted to be 38.1 ppbv by the model where 39.5 is observed, an increase of 4%. However, sites show notable differences. For Marylebone Road, O_x concentrations have gone down by roughly 5 ppbv (12%). This may indicate a reduction in the direct O_x source from the emissions of NO_2 by diesel vehicles. For 4 of the sites, the changes are relatively small and not significantly different from 0. Thurock, Bloomsbury and Eltham show increases in O_x concentrations. This may indicate an increase in photochemical activity due to lower NO_x concentrations which would lead to more efficient O_3 production. However, more investigation will be needed to diagnose the chemistry occurring here.

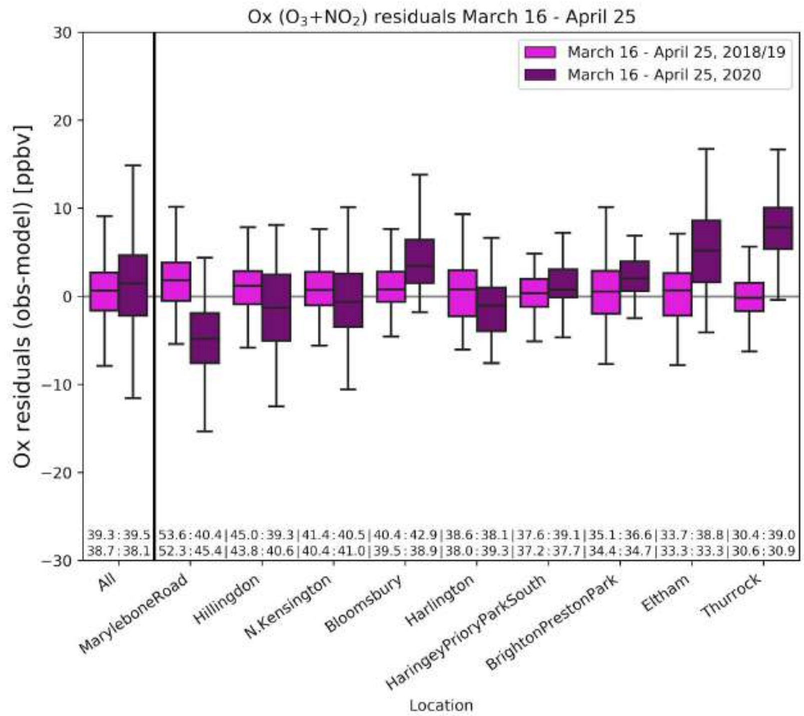


Figure 6. *Distribution of observation minus model residuals of odd oxygen (O_x=NO_2+O_3) from 16th March to 25th April for a number of AURN sites in the SE of England. Dark green boxplots show the difference between observed and model-predicted O_x (NRT+ML) for the COVID-19 restriction period (16th March to 25th April 2020). Light green boxplots show the difference between observed and predicted (NRT+ML) for the period March 16 to April 2 for 2018 and 2019. Dark green values indicate the impact of COVID-19 restrictions. Coloured bars indicate the 10-90% percentile, black line within the bar given the median change, error bars indicate the 5-95% percentile change. The average for all sites is shown in the leftmost boxplot. The numbers at the bottom of the figure indicate the mean concentrations as observed (upper row) and modelled (lower row), for 2018,2019 (left) and 2020 (right). Stations are ordered by pre-COVID19 average O_x pollution concentrations.*

PM2.5

Figure 7 shows the results for the sites reporting PM2.5. During the restriction period, mean PM2.5 concentrations were lower than the 2018,2019 average (14.5 vs 18.0 μgm^{-3}). The model prediction for the 2020 period (17.5 μgm^{-3}) is slightly lower than the 2018,2019 periods. Overall it appears that during the COVID restrictions mean PM2.5 drops by around 3 μgm^{-3} (17%) for the sites explored here. There are differences between the sites. At some sites (Marylebone Road, Bexley, etc.) PM2.5 concentrations drop by 25%. Standford-Le-Hope Roadside shows an increase of 4 μgm^{-3}. The model predicts lower concentrations during the 2020 period than the 2018,2019 mean for some sites (Marlyborne Road, Westminster). This may indicate trends at these sites which are being extrapolated forwards by the modelling approach. Future work will be needed to explore these features.

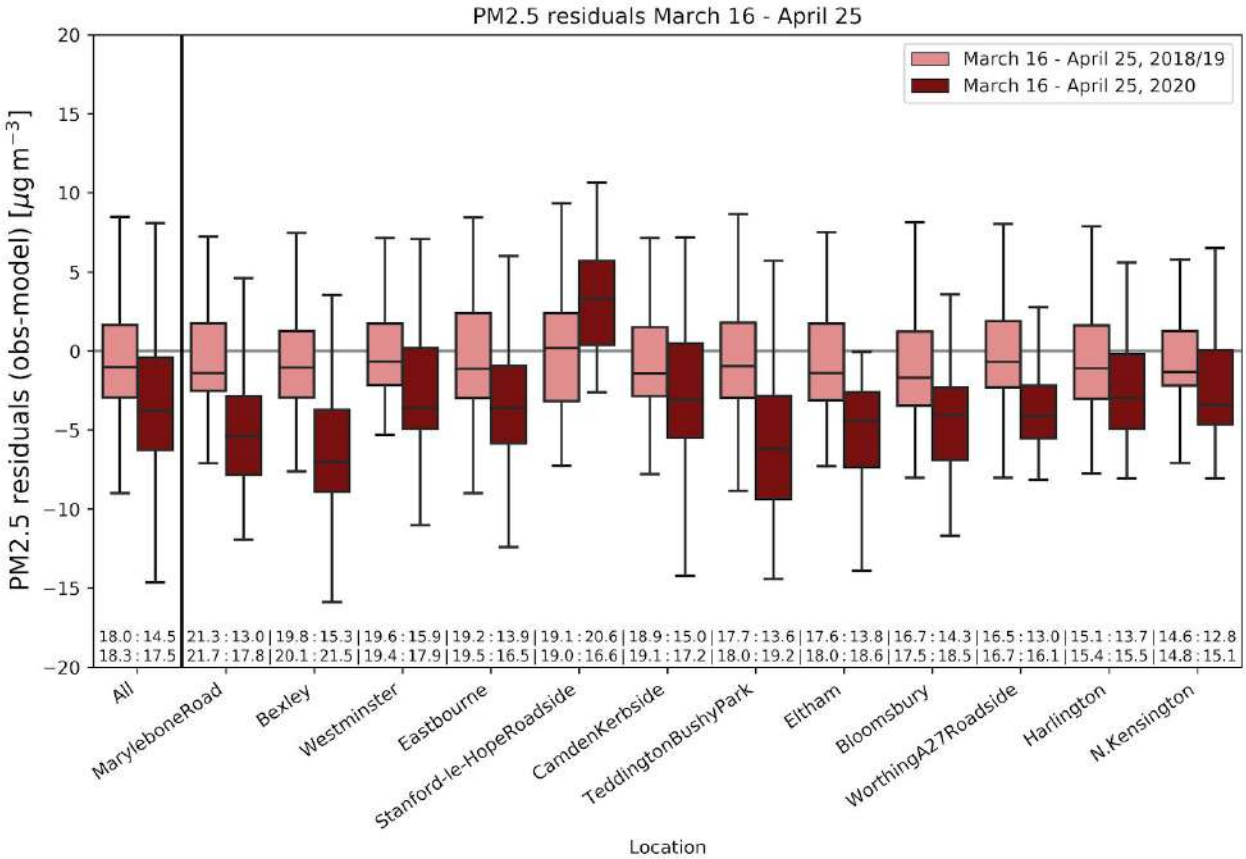


Figure 7. *Distribution of observation minus model residuals of PM2.5 from 16th March to 25th April for a number of AURN sites in the SE of England. Dark green boxplots show the difference between observed and model-predicted PM2.5 (NRT+ML) for the COVID-19 restriction period (16th March to 25th Apri, 2020). Light green boxplots show the difference between observed and predicted (NRT+ML) for the period March 16 to April 2 for 2018 and 2019. Dark green values indicate the impact of COVID-19 restrictions. Coloured bars indicate the 10-90% percentile, black line within the bar given the median change, error bars indicate the 5-95% percentile change. The average for all sites is shown in the leftmost boxplot. The numbers at the bottom of the figure indicate the mean concentrations as observed (upper row) and modelled (lower row), for 2018,2019 (left) and 2020 (right). Stations are ordered by pre-COVID19 average PM2.5 pollution concentrations.*

Appendix 1: Machine Learning Bias Correction

The idea behind the bias correction methodology is to identify - and ultimately correct for - recurring, systematic differences ('biases') between air pollution concentrations as observed by a local monitoring site and simulated by the NASA GEOS-CF model. These model biases can be caused by a wide range of factors, such as model difficulties to resolve local features (e.g., roadside emissions), errors in the simulated meteorology (e.g., boundary layer height in very cold weather), or incompleteness of the chemical mechanism. Based on previous work [Ivatt and Evans, 2019], rather than attempting to identify and quantify these biases manually, we use the XGBoost machine learning algorithm (https://xgboost.readthedocs.io/en/latest/#) to develop a statistical relationship between historical differences between GEOS-CF model prediction and actual observation at the location of interest. In essence, the machine learning model provides a 'correction factor' for the coarse model output so that it relates more closely to a localized observation. Once the machine learning model has been trained for a location, it can be applied to newly generated model output to offer an improved air quality prediction in near real-time. While not perfect, the differences between this corrected model prediction and the actual observations are expected to be normally distributed with a mean error (bias) around zero, unless there has been a fundamental change in the system that the machine learning system has not been trained on. If such a change occurs, the corrected model will keep predicting the expected concentration under the business-as-usual scenario and thus start to deviate from the true observations.

To test the impact of COVID-19 restrictions on air quality in the London area, we applied the methodology to the NASA GEOS-CF model and 21 AURN sites around London, focusing on NO_2, O_3, and PM2.5. In addition, we also looked at Ox (NO_2 + O_3) to test if there is any evidence for changes in overall photochemical production due to COVID-19.

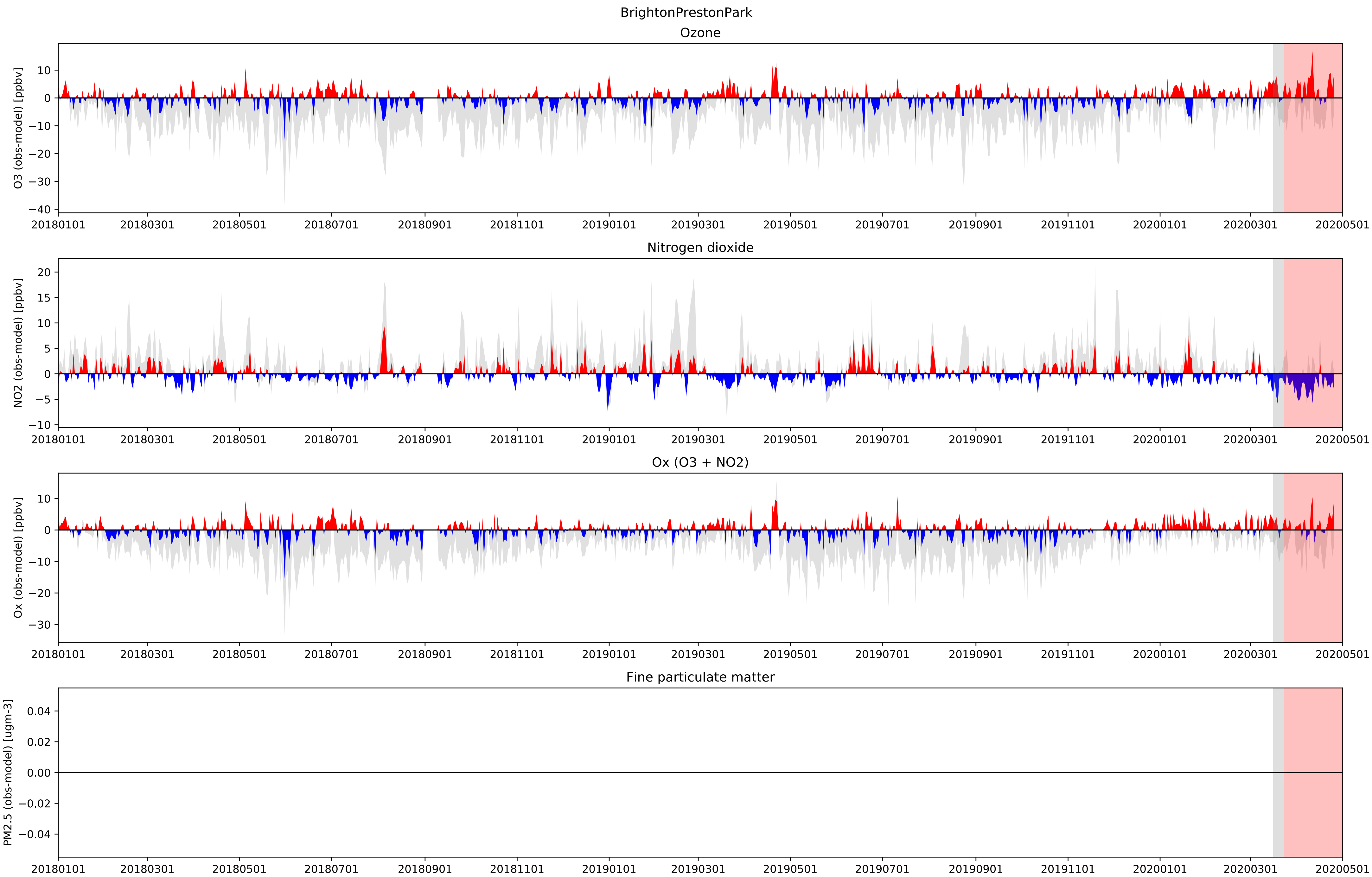
Hourly observations were obtained from the OpenAQ platform (https://openaq.org/), and the corresponding GEOS-CF model output was accessed locally from the NASA Discover supercomputer (https://gmao.gsfc.nasa.gov/weather_prediction/GEOS-CF/data_access/). For each site and species, we trained a separate bias correction model. For the training, we used 50% of the model output (and corresponding observations) between Jan 1, 2018 and Dec 31, 2019, selected randomly. We find little sensitivity in our results to a shorter training window (e.g., Jan 1, 2018 - Dec 31, 2018) or a smaller sampling size.

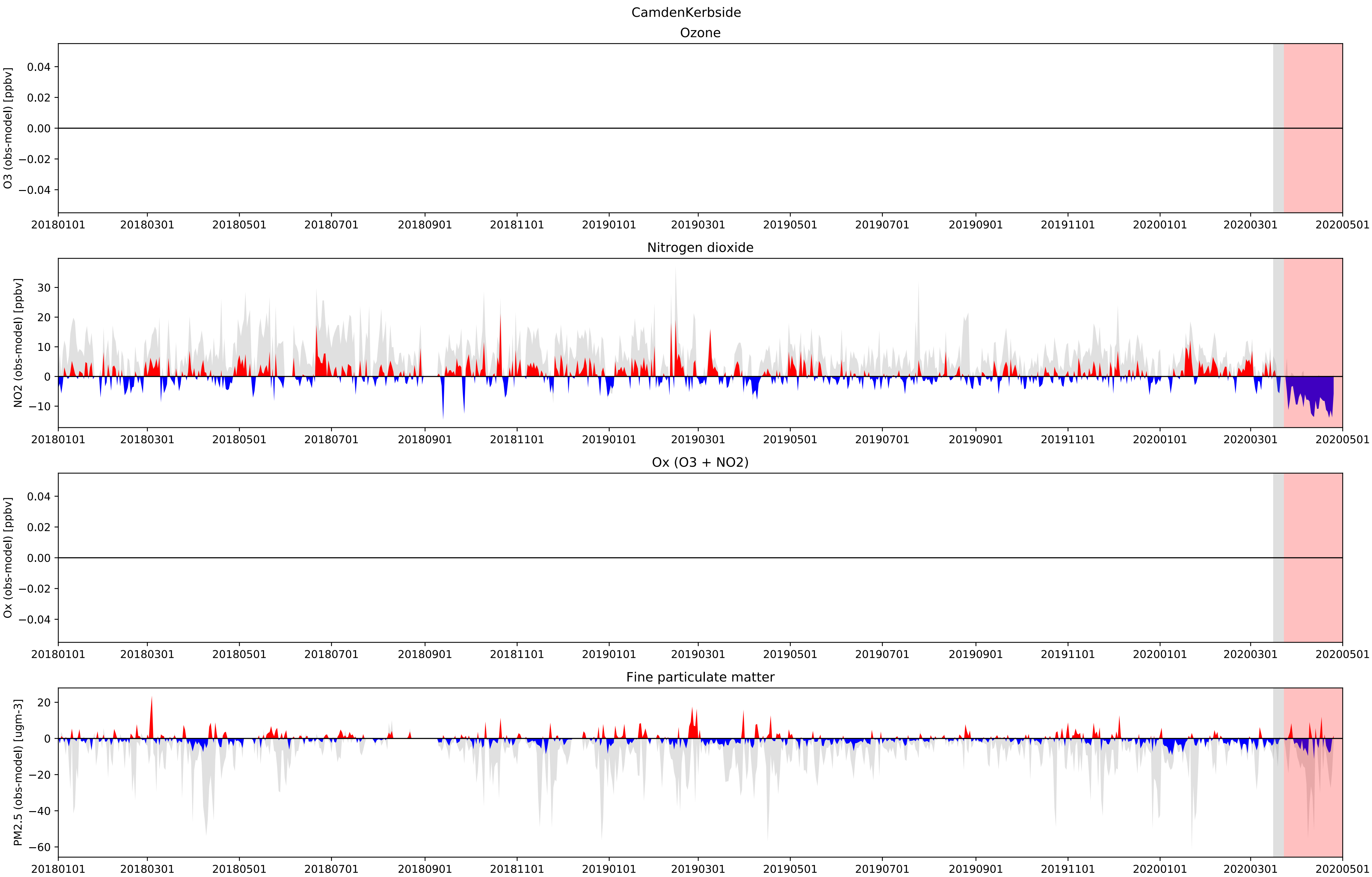
As inputs to the machine learning model we use 8 meteorological parameters as simulated by the GEOS-CF model for the given location (surface North- and Eastward wind, surface temperature, surface relative humidity, cloud coverage, precipitation, surface pressure, and planetary boundary layer height), the surface concentrations of 49 chemical species at the given location (ozone, nitrogen oxides, carbon monoxide and volatile organic compounds, aerosols), and 31 model emissions at the given location. In addition, we provide as input features the hour-of-day, weekday, and calendar days since Jan 1, 2018. These inputs allow the machine learning model to identify systematic model - observation mismatches related to the diurnal and weekly cycle of the pollutants, and also to correct for long-term trends in air pollution (e.g., due to a steady decrease in emissions not captured by the model).

Based on these inputs, we train the machine learning model to learn the differences between the observed concentration and the model predicted values. The absolute concentration difference (the model 'bias') is used as the predictor variable. It should be noted that - in cases where the residuals are heavily skewed (e.g., for PM2.5) - it might be more appropriate to transform the residuals first, e.g., using a logarithmic transformation. This will be the focus of future research. It should also be noted that this approach is designed to correct for model-observation differences that are either systematic (e.g., a general model bias), or local in scale (e.g., misrepresentation of local sources). It is not suitable for identifying and correcting infrequent outlier events, such as a Saharan dust storm reaching the UK.

Appendix 2

Figures A1 - A21 show the observation-model residuals for all 21 sites from Jan 1, 2018 to April 25, 2020 for NO_2, O_3, Ox, and PM2.5. The shaded grey area shows the residuals between the observations and the original (uncorrected) model, and the coloured areas represent the remaining observation-model differences after applying the machine learning correction. Positive values indicate times where the model underpredicted the observations, and negative values denote cases where the model overpredicted the actual observation. While the analysis is done using hourly values, we show the daily average values to smooth out some of the small-term variability. The dark grey shaded area at the end indicates the onset of social distancing recommendations on 16th March 2020, and the red shaded area indicates official lockdown after 23rd March.


BrightonPrestonPark
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


CamdenKerbside
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


Eastbourne
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
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20190501
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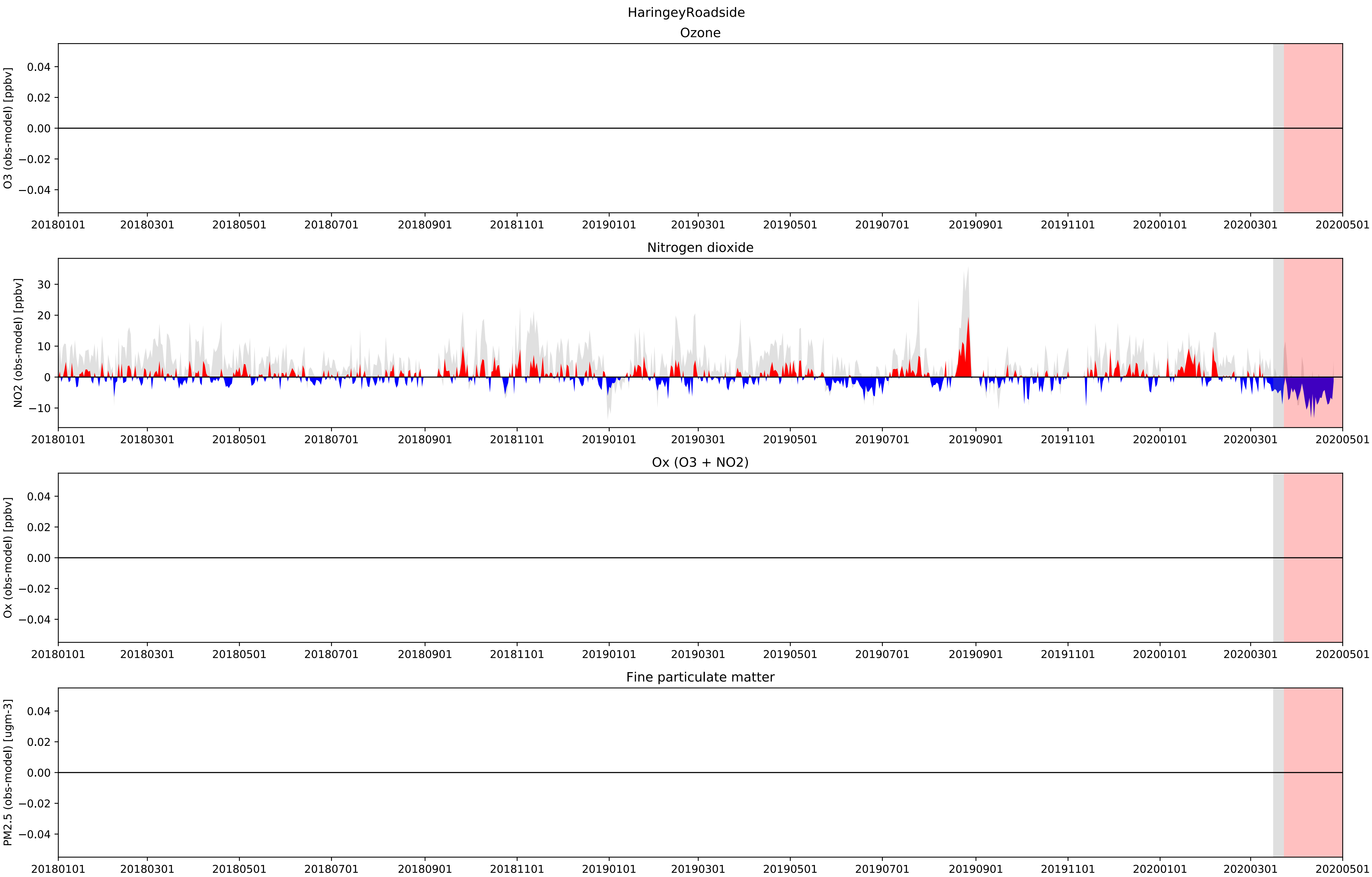
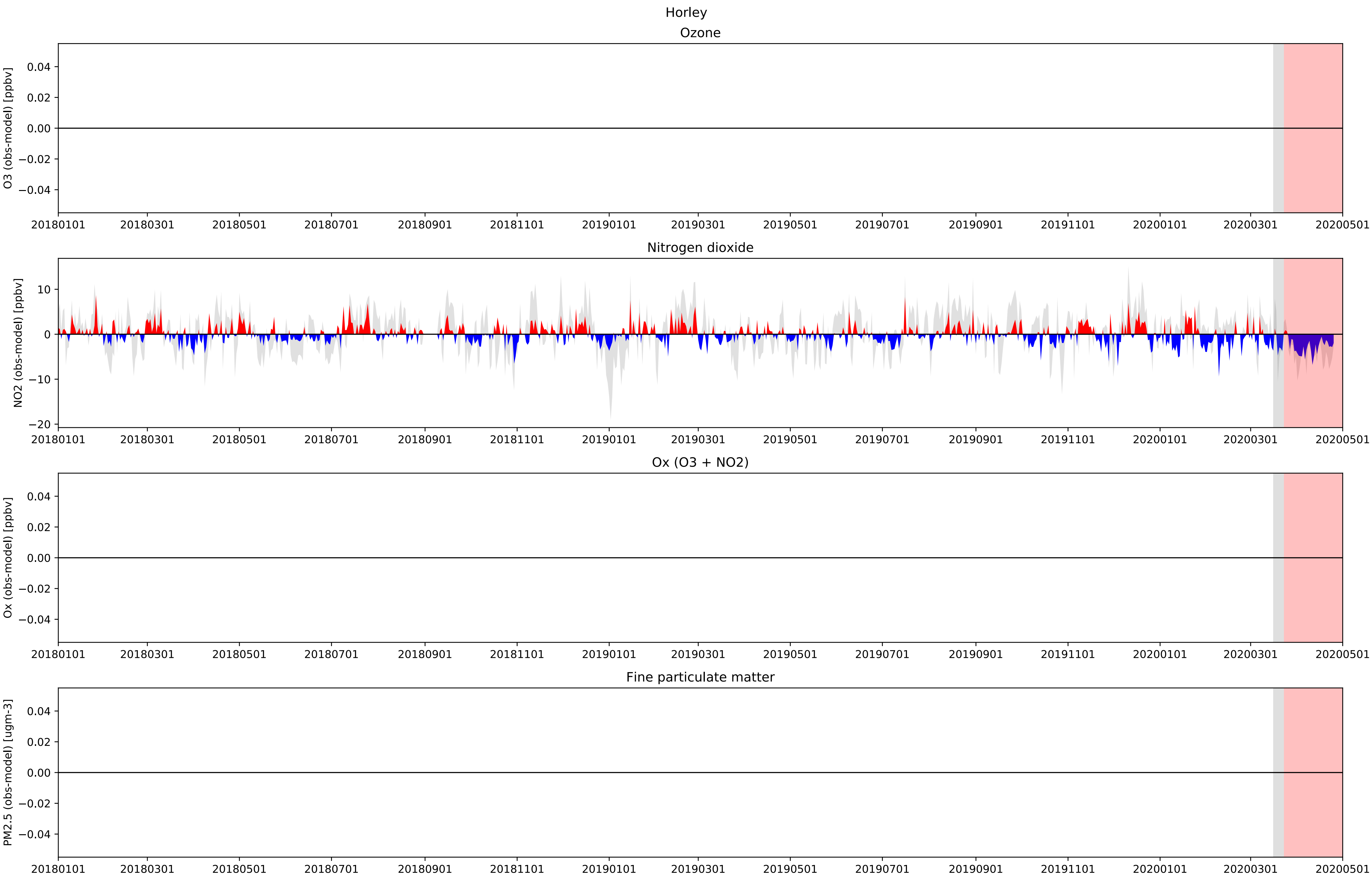
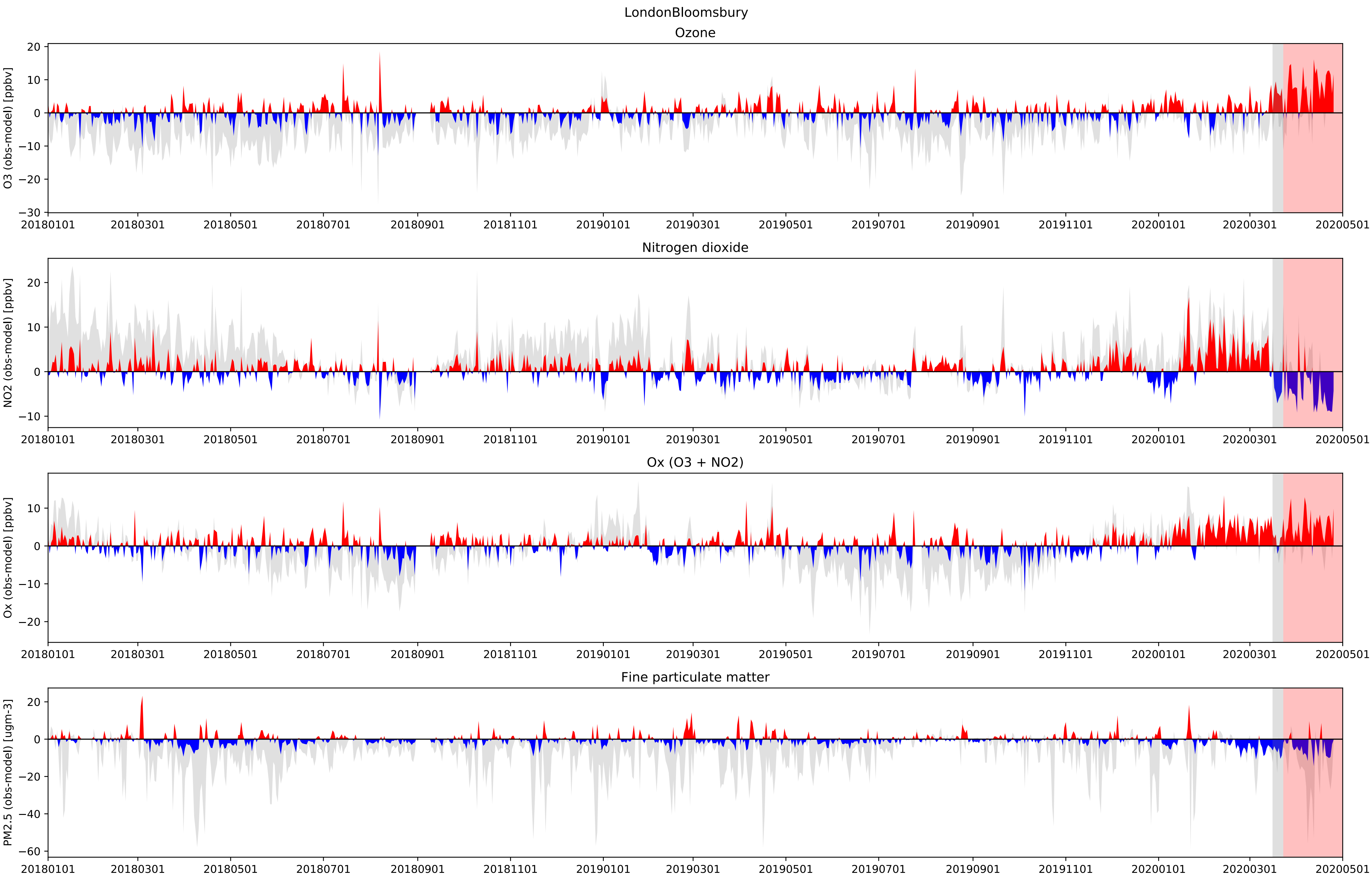
20190901
20191101
20200101
20200301
20200501



HaringeyRoadside
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


Horley
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


LondonBexley
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


LondonBloomsbury
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
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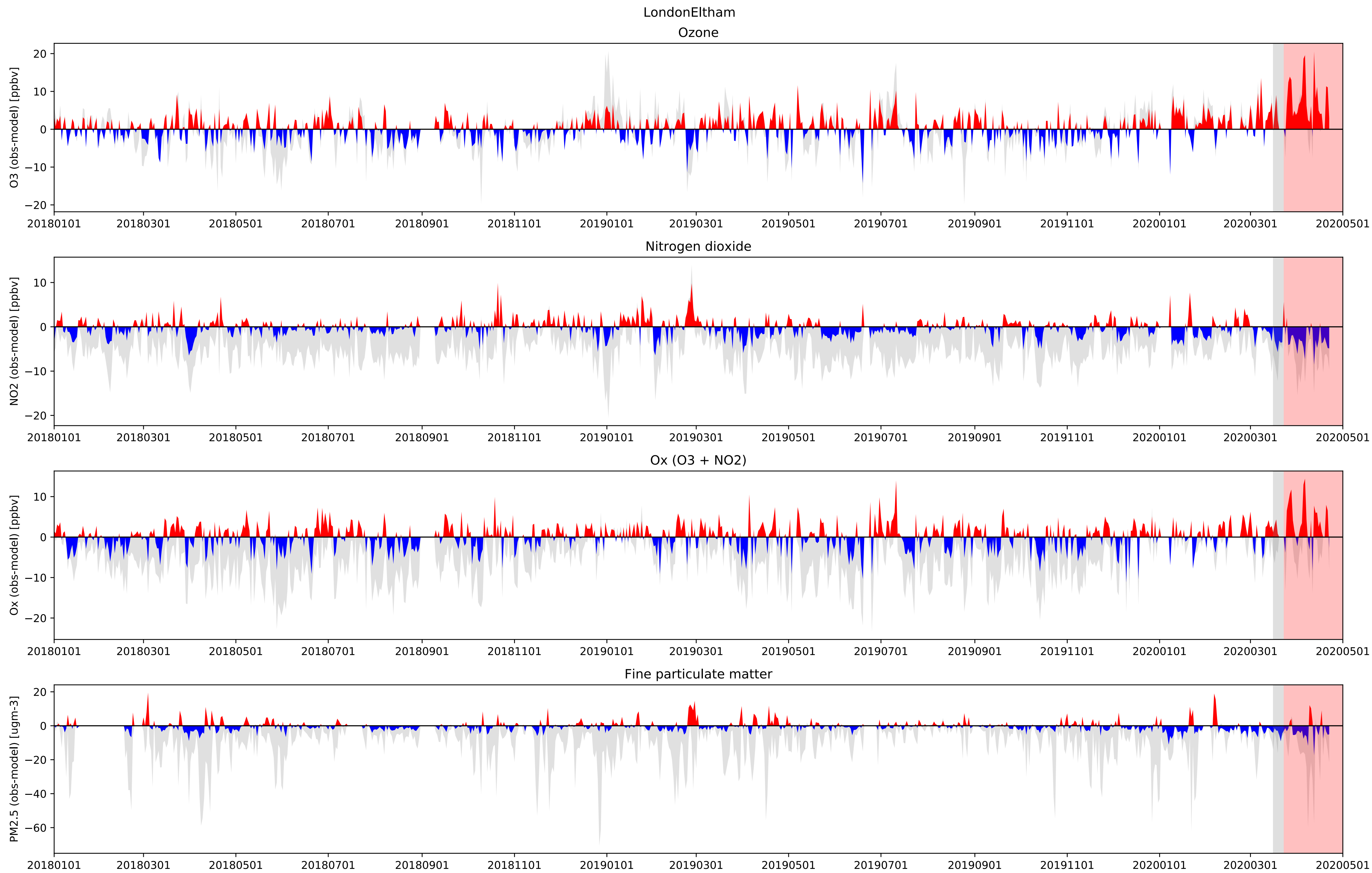
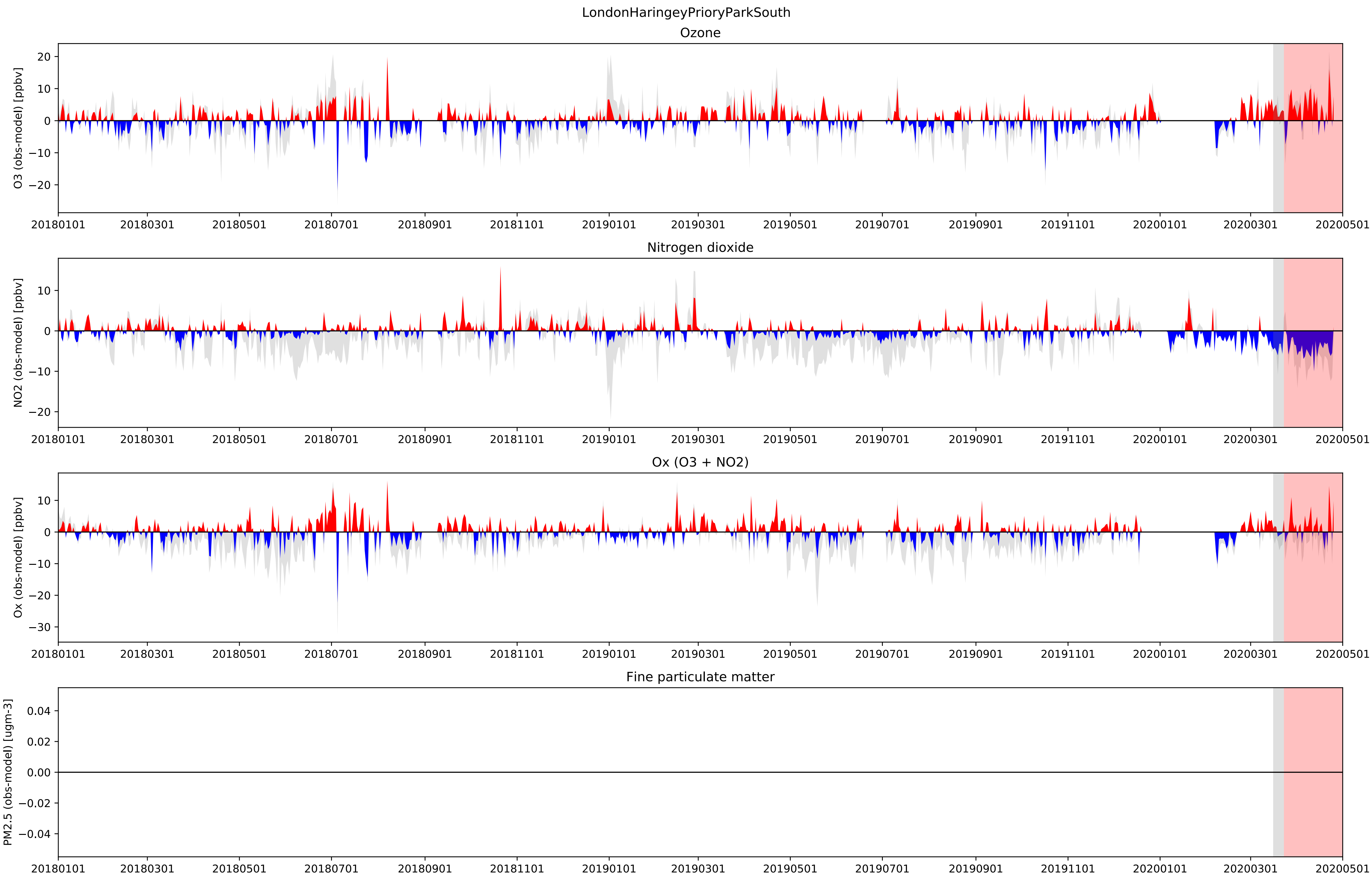
20200101
20200301
20200501


LondonEltham
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


LondonHaringeyPrioryParkSouth
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
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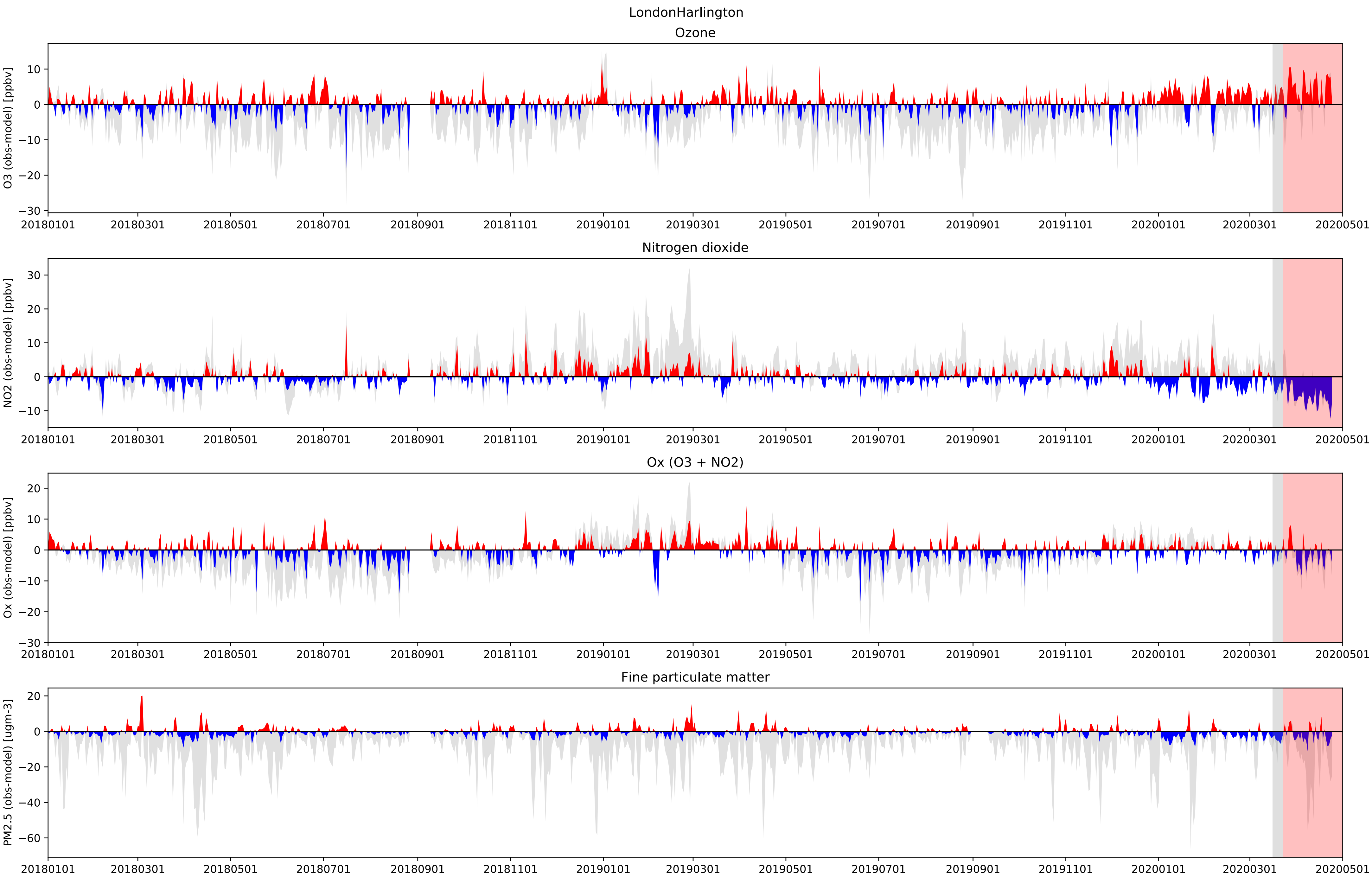
20200101
20200301
20200501


LondonHarlington
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
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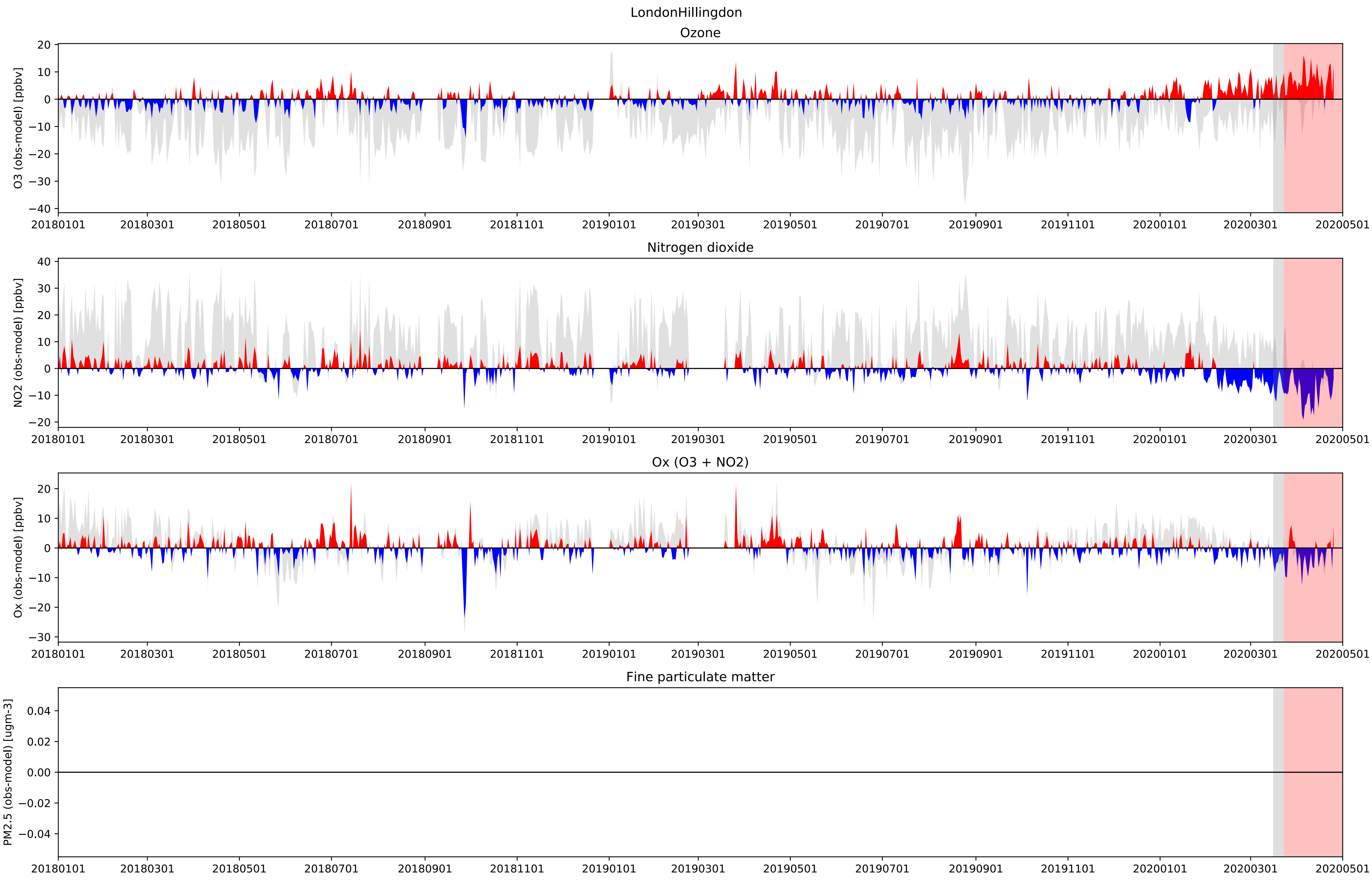
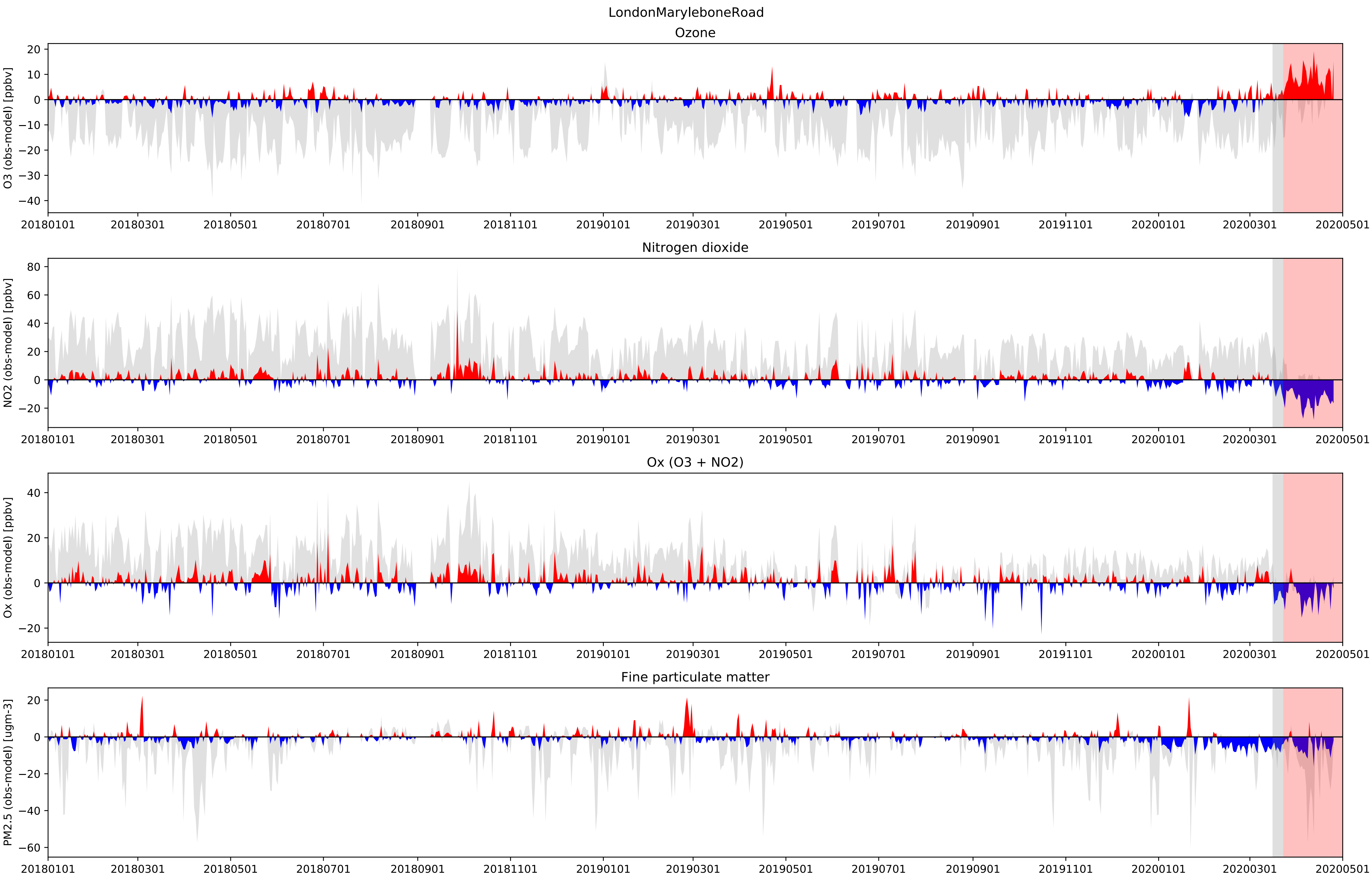
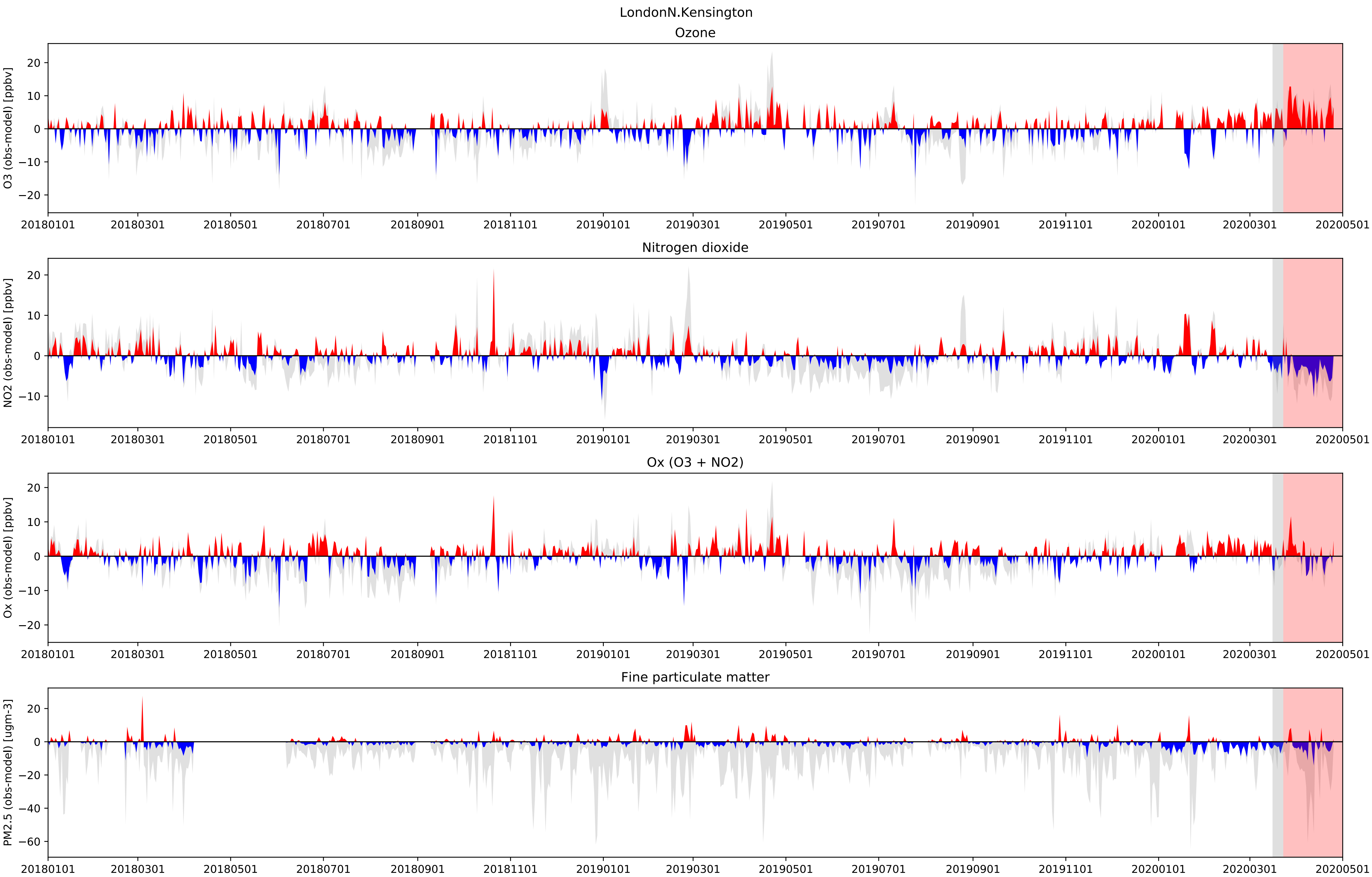
20200101
20200301
20200501


LondonHillingdon
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


LondonMaryleboneRoad
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501



LondonN.Kensington
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
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20190101
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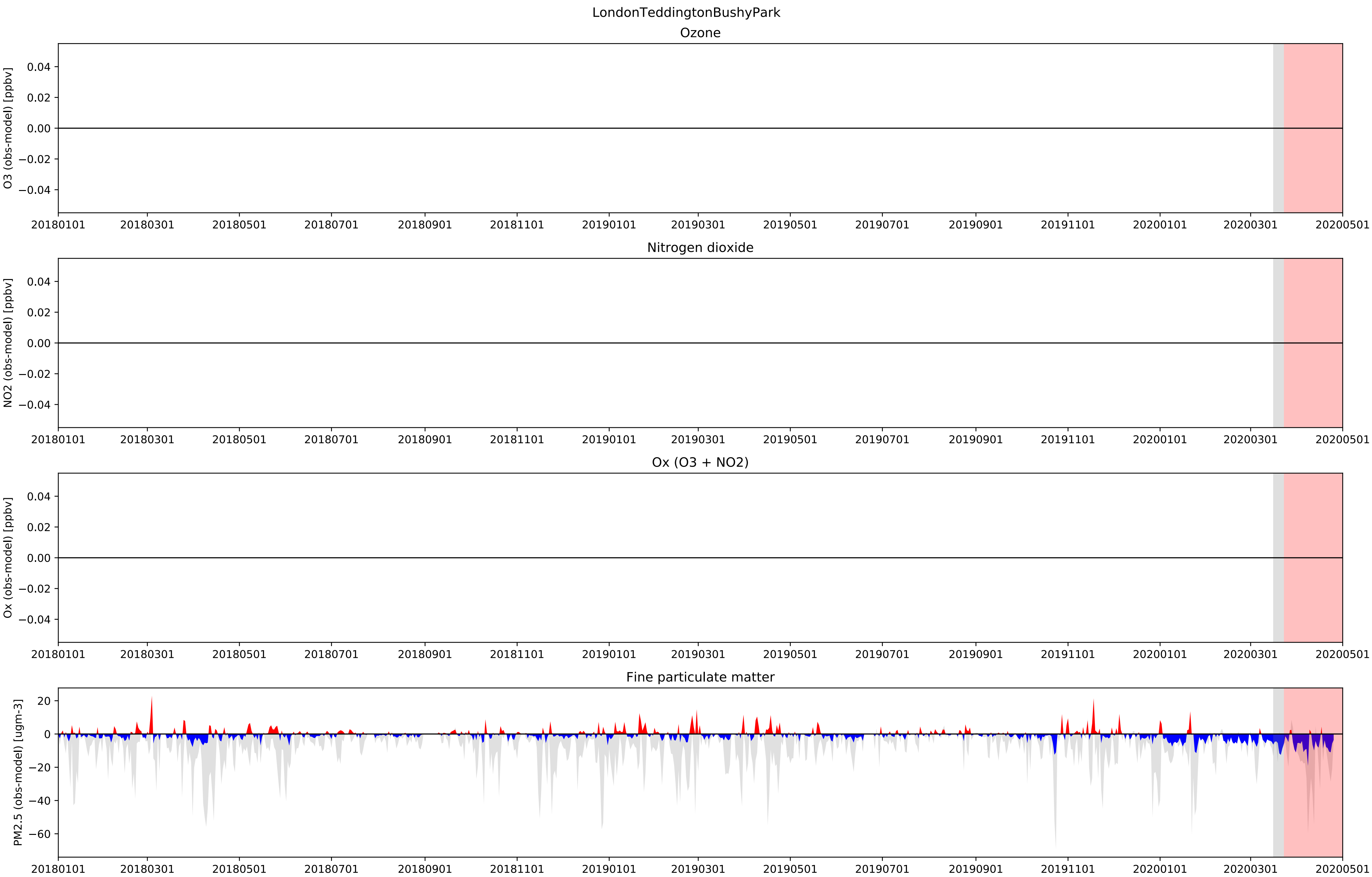
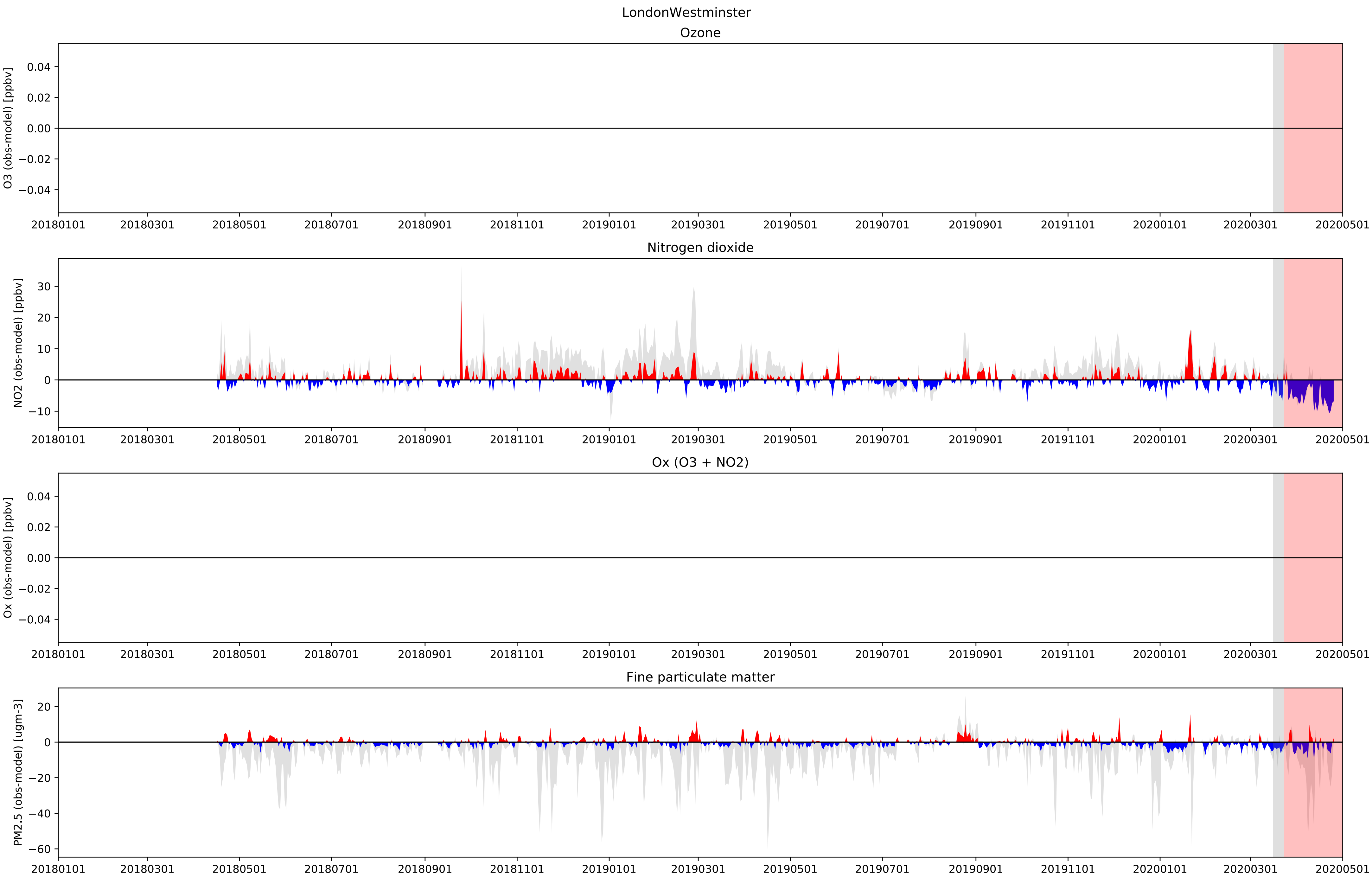
20200101
20200301
20200501



LondonTeddingtonBushyPark
Ozone
O3 (obs-model) [ppbv]
0.04
0.02
0.00
−0.02
−0.04
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20
0
−20
−40
−60


LondonWestminster
Ozone
O3 (obs-model) [ppbv]
0.04
0.02
0.00
−0.02
−0.04
Nitrogen dioxide
NO2 (obs-model) [ppbv]
30
20
10
0
−10
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20
0
−20
−40
−60
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
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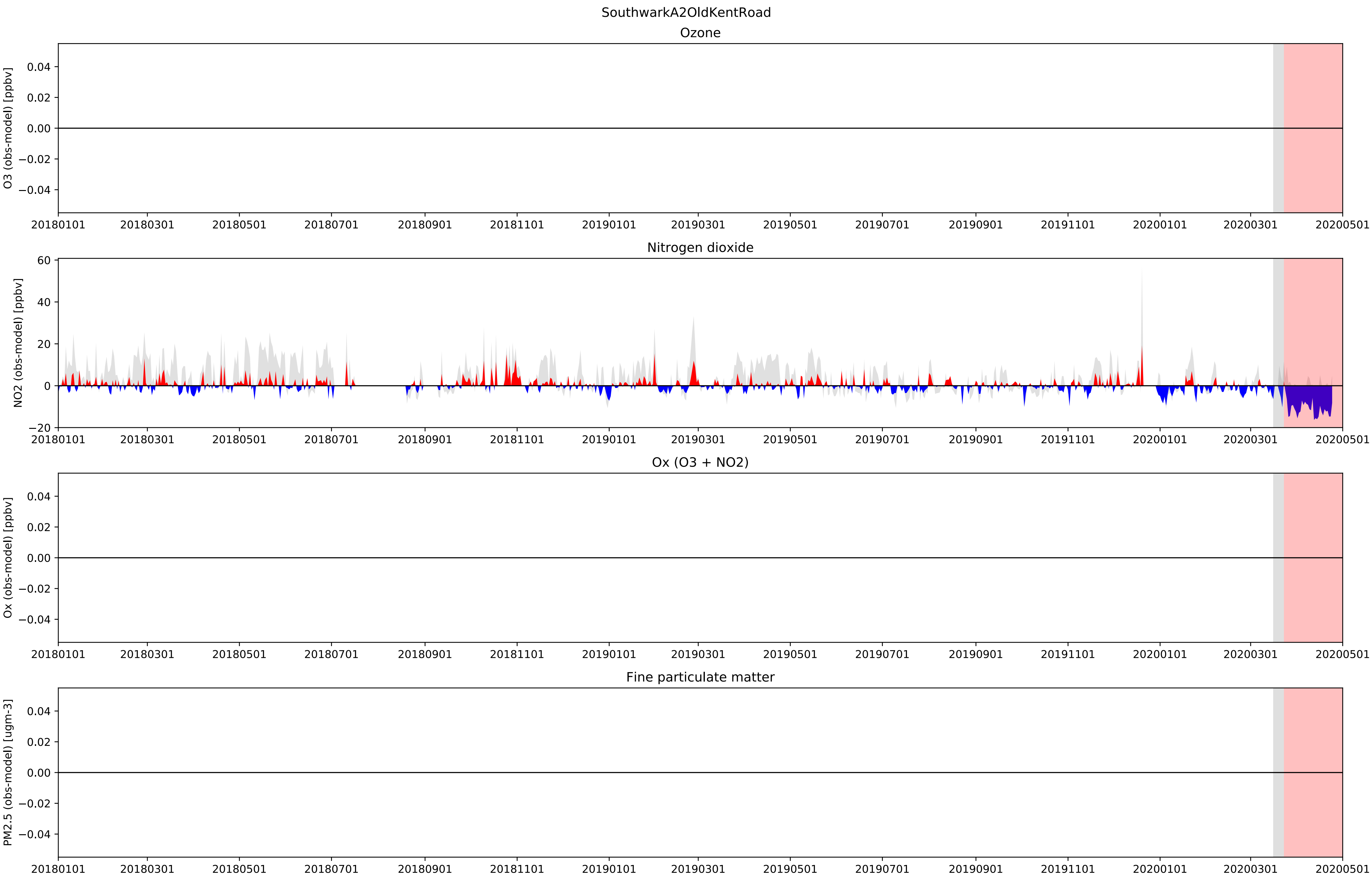
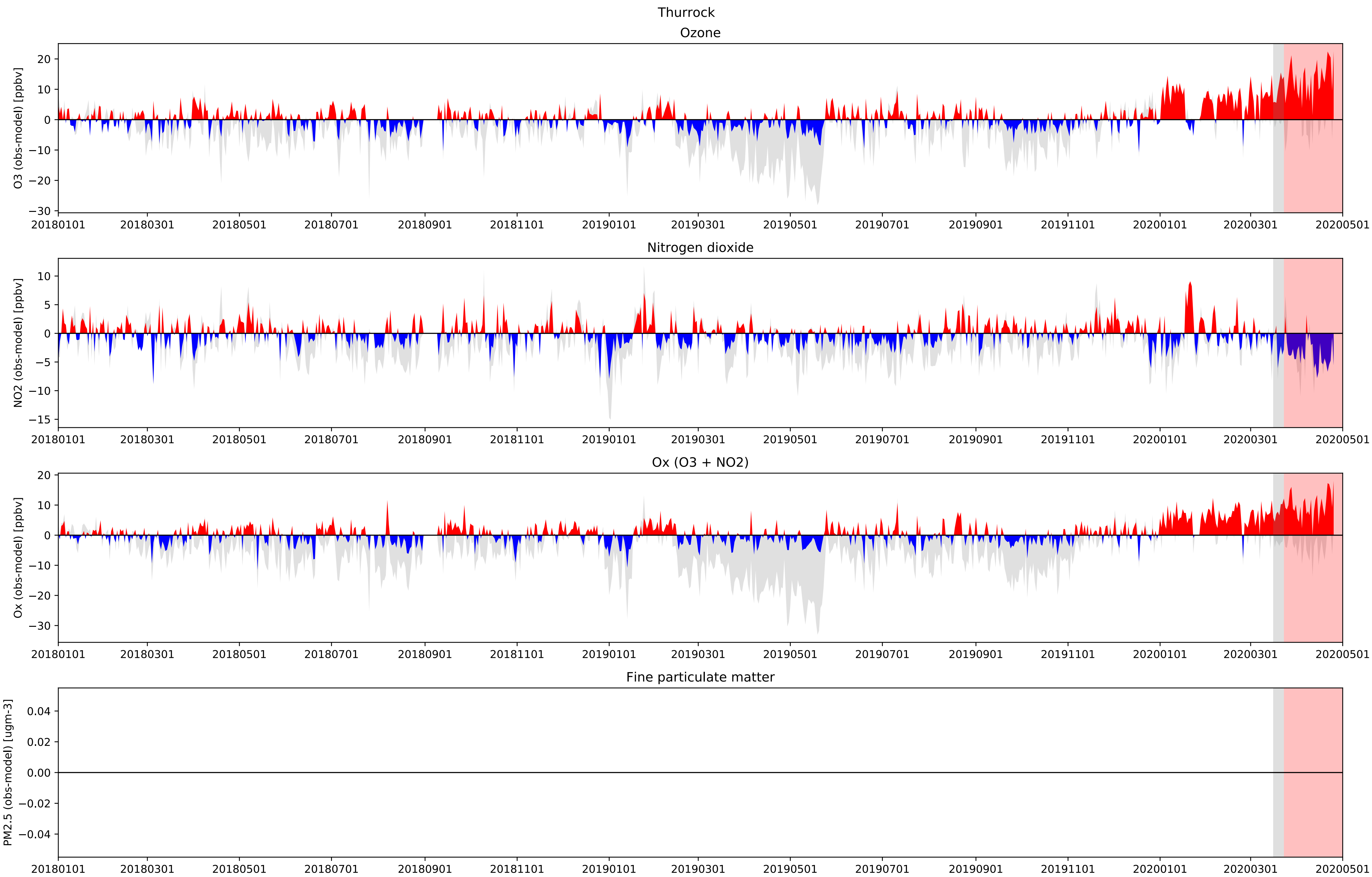
20200101
20200301
20200501


SouthwarkA2OldKentRoad
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


Stanford-le-HopeRoadside
Ozone
O3 (obs-model) [ppbv]
0.04
0.02
0.00
−0.02
−0.04
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501
Nitrogen dioxide
NO2 (obs-model) [ppbv]
20
10
0
−10
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20
0
−20
−40
−60


StorringtonRoadside
Ozone
O3 (obs-model) [ppbv]
0.04
0.02
0.00
−0.02
−0.04
Nitrogen dioxide
NO2 (obs-model) [ppbv]
30
20
10
0
−10
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501


Thurrock
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
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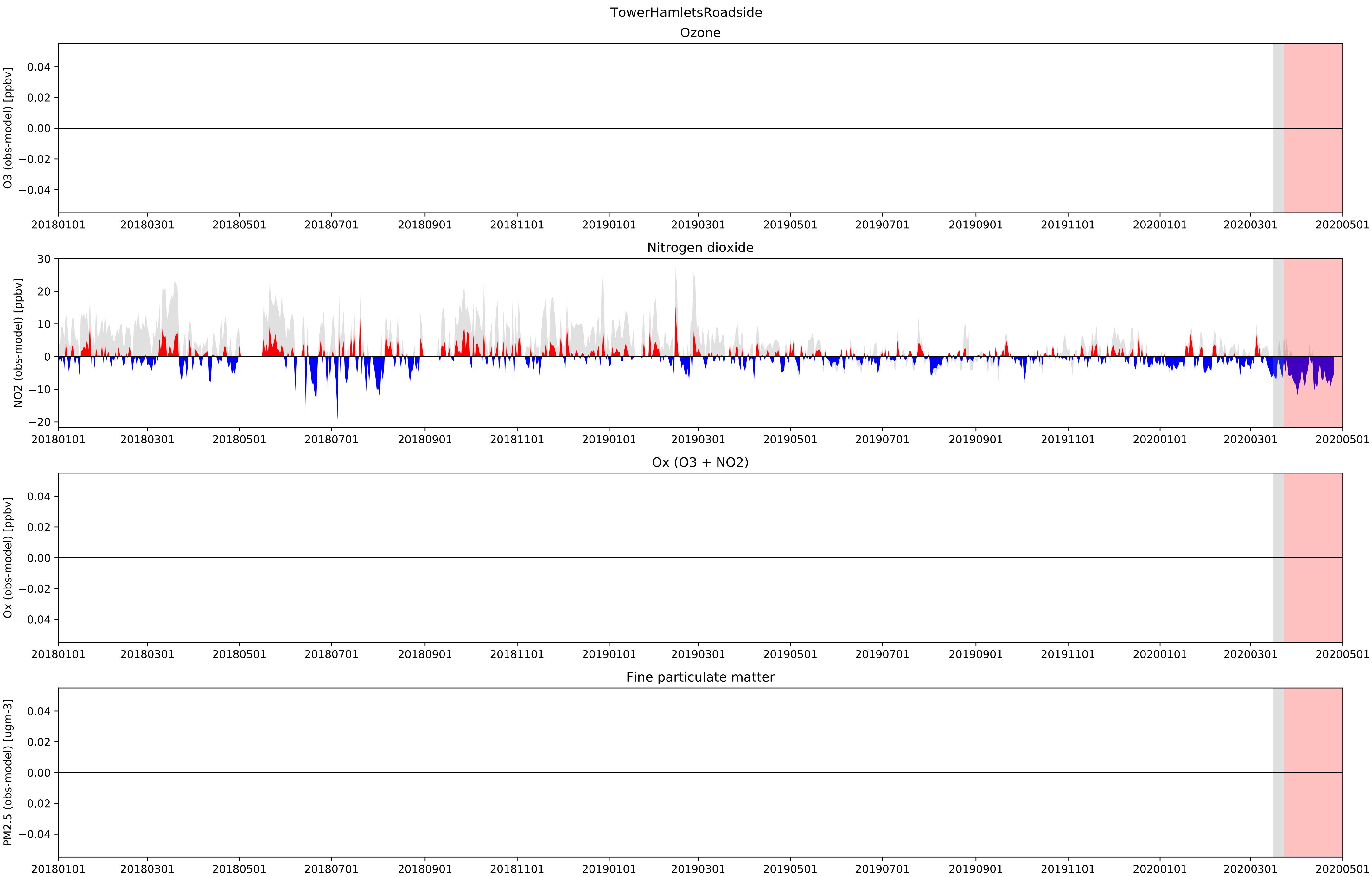
20200101
20200301
20200501


TowerHamletsRoadside
Ozone
O3 (obs-model) [ppbv]
0.04
0.02
0.00
−0.02
−0.04
Nitrogen dioxide
NO2 (obs-model) [ppbv]
30
20
10
0
−10
−20
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
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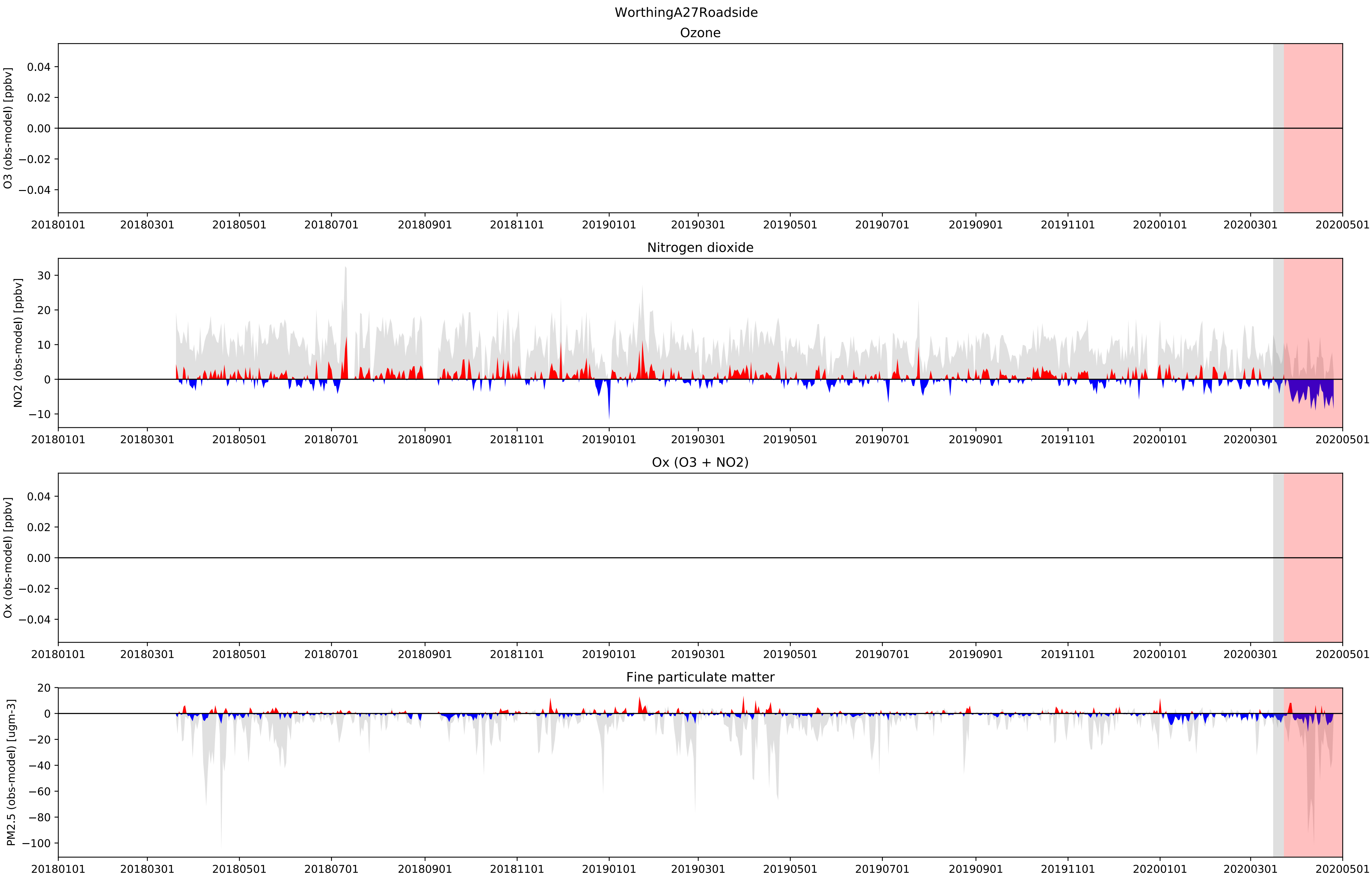
20190901
20191101
20200101
20200301
20200501


WorthingA27Roadside
Ozone
O3 (obs-model) [ppbv]
Nitrogen dioxide
NO2 (obs-model) [ppbv]
Ox (O3 + NO2)
Ox (obs-model) [ppbv]
Fine particulate matter
PM2.5 (obs-model) [ugm-3]
20180101
20180301
20180501
20180701
20180901
20181101
20190101
20190301
20190501
20190701
20190901
20191101
20200101
20200301
20200501

Contribution to the AQEG/Defra Call for Evidence

From the Centre for Atmospheric Sciences and Computer Science, University of Manchester

Contributed by: Rami Alfarra, James Allan, Khalid Alzahrani, Saleh Alzahrani, Hugh Coe, Caroline Jay, Nicholas Marsden, Gordon McFiggans, Ernesto Reyes-Villegas, Hugo Ricketts, Jonathan Taylor, David Topping, Tong Wu

The following is a summary of key points. The main data and analyses supporting these comments are listed in appendices.

The University of Manchester Air Quality Supersite (MAQS) issited on the University Fallowfield campus (53º26'39''N, 2º12'52''W) and is an urban background site located approximately equidistant between the Manchester Piccadilly Gardens and Sharston AURN sites around 4 km to the south of the centre of Manchester..

1. Meteorological Context

Westerly, wet conditions dominated UK weather throughout the start of 2020 until mid-March. The winter of 2019-2020 was unusually mild and wet as a result of a very strong Atlantic Jetstream positioned further to the south than normal for the time of year. This steered successive low pressure systems across the UK including the named storms Ciara, Dennis and Jorge through the month of February and early March. The winter was the 5th wettest on record and the 5th mildest. February was the wettest ever recorded.
(https://www.metoffice.gov.uk/about-us/press-office/news/weather-and-climate/2020/2020-winter-february-stats)
.

High pressure and easterly winds have dominated UK weather since mid-March. A high pressure system has been quasi-permanently positioned over the North Sea and Scandinavia since mid-March leading to dry conditions, high temperatures and southerly or easterly winds of variable strength. These conditions are atypical for late March and early April.

There has been a predominance of easterly and south easterly air masses during the lockdown period. 72-hour back trajectories were generated using HYSPLIT. These were initiated hourly from Manchester throughout March and April 2020 (Appendix 1). The trajectories were classified into 5 clusters. There is a clear difference in the frequency of the clusters between pre and post-lockdown, with the former dominated by westerly conditions and the latter by southerly and easterly air masses. Average PM2.5 concentrations for the different clusters show significantly enhanced concentrations in the easterly and southerly clusters.

Care should be taken where comparing trends in air pollutants as a result. We recommend caution in using trends in time series through the course of 2020 to diagnose air pollution effects of lockdown and also when comparing year to year differences between 2020 and earlier years due to atypical conditions and large changes in meteorological situation before and post lockdown.

2.Changes in concentrations of pollutants since the COVID outbreak

NO_x shows substantial reduction in most locations in the UK. Diurnal variations of NO_x at Urban Roadside sites across the AURN network show fractional reductions of between 20 and 80% of the average diurnal cycle from April over the previous 5 years (Appendix 2). There is considerable site to site variability with some locations showing far less reduction than others. A small number of sites show a modest increase, for example Edinburgh. Whether this is due to changes in the fleet, driving pattern or other causes is not clear but the reductions are not uniform. Background urban locations show less reduction with most locations showing between 10% and 60% reductions in the average diurnal cycle for April compared to the last 5 years. Again, a few locations show a modest increase. The extent of the reductions also varies with time of day with generally lower reductions observed during the morning rush hour than in the middle of the day or early evening.

There is considerable variability between sites but no systematic reduction in PM2.5 or O_3 in the UK since the start of the outbreak. The fractional differences between the diurnal cycles during April 2020 and the previous 5 years averaged across all AURN show little systematic change in ozone though there is a large amount of variability from location to location (Appendix 2). Similar data for PM2.5 shows a modest reduction of around 25% at urban traffic dominated sites. There is no discernible change in PM2.5 comparing April 2020 to previous years at urban background locations. There are two populations of rural background site that show either increases or decreases in PM2.5 of typically 50% compared to the long term April average, again with considerable variability.

Ammonia mixing ratios are elevated in central Manchester during the periods of long range transport across the UK and northern Europe during the lockdown period. The time series of ammonia in appendix 3 shows elevated mixing ratios occur at similar times to enhanced, transported particulate matter. Since springtime is a common period for agricultural muck spreading the springtime easterly and southerly winds lead to greatly enhanced ammonia during this period and will drive up concentrations of ammonium nitrate,

A major moorland fire caused very high concentrations of PM2.5 to be observed across the Greater Manchester region on April 23rd 2020. There was a moorland fire at Rakes Moss on the A628 Crowden to Glossop on Thursday around 13:00 (appendix 3). A maximum concentration of PM close to 300 μgm^{-3}, nearly all of which was PM1 and over 85% was organic mass. We would caution including this period in any analysis of the effects of the lockdown but it serves to indicate that major air quality events can occur during periods of reduced activity.

Assessment of pre and post lockdown periods as a function of air mass show substantial reductions in NO_x across all air masses but no significant variation for PM2.5 in any air mass. Average NO_x and PM2.5 from the Manchester Air Quality Supersite were calculated for each of the air mass classifications based on back trajectories before and after the lockdown. NO_x showed substantial reductions in all clusters, except those transporting air from northern Europe whereas there was no detectable reduction in PM2.5 in any of the air mass clusters (Appendix 3).

Secondary production of aerosol during regional transport has been a major component of PM2.5 during the lockdown period. Back trajectory cluster analysis was used to show elevated concentrations of PM2.5 in Manchester occurred post lockdown during the prevalent stagnant and southerly and easterly flows across the UK (Appendix 1). This was extended to assess the main chemical components of PM2.5 in different air masses (Appendix 4). High concentrations of ammonium nitrate and ammonium sulphate were observed during periods of easterly and southerly winds and during stagnant air. High levels of organic matter were also observed, which we show are at least 80% secondary in nature.

Ammonium nitrate concentrations are substantial throughout the lockdown period despite reductions in NO_x. Despite the large reductions in NOx emissions observed across the UK during April 2020, ammonium nitrate continued to make a significant contribution to fine mode PM with concentrations of ammonium nitrate between 10 and 20 μgm^{-3} during easterly and southerly conditions (Appendices 3 and 4). These concentrations are not dissimilar to those measured previously in Manchester. This arises since ammonia concentrations remain high and reductions in NO_x are insufficient to prevent appreciable reductions in ammonium nitrate.

PM2.5 concentrations in both clean and regionally polluted conditions were not dissimilar to those observed in similar springtime conditions over the last 5 years. The back trajectory cluster analysis was used to define an air mass categorisation based on local wind speed and direction information to separate conditions when regional pollution conditions dominate from periods of cleaner air. The PM2.5 concentrations during these two conditions in the lockdown period of April 2020 were compared to those of the previous 5 years. No differences were observed in either meteorological situation, implying that there was no noticeable difference between the regional background particle pollution during stagnant UK pollution events or during European outflow as a result of changes in emission due to lockdown.

3. Evidence for changes in emissions since the COVID outbreak

There is no noticeable change in the absolute concentration of black carbon before and after lockdown but there is a detectable change in the proportion of black carbon arising from wood burning. Aethalometer measurements at the Manchester Air Quality Supersite (Appendix 5) have been analysed through the pre and post lockdown period. There is no discernible difference in the total black carbon mass loading before and after lockdown occurred. A well-recognised model to discriminate between wood-burning and traffic sources of black carbon was employed to derive separate contributions from these sources based on differences in absorption with wavelength. This model shows a distinct reduction in the traffic contribution overall, particularly during the morning rush hour and throughout the daytime period. The wood burning signature increases during the evenings post-lockdown. This is consistent with the diurnal signature in particulate potassium, a known marker for wood burning, and closely related to the collapse of the daytime planetary boundary layer at the end of the day as shown from the ERAS data and heat flux measurements at the site.

Wood burning is a significant source of particulate Zinc but Copper is associated with vehicle traffic emissions. Data from MAQS show particulate Zn increases in concentration during the periods of easterly and southerly winds, whereas the Cu seems to be more locally generated (Appendix 3). The week to week variability and diurnal cycle in Zinc shows strong similarities to known markers for wood smoke emissions throughout the period. The variability in Cu follows that of NO_x and the component of BC associated with traffic. Brake wear has previously been identified as a source of particulate Cu and the MAQS data are consistent with these earlier observations.

Time series modelling of NO_2 in central Manchester predicts post lockdown reductions of NO_2 of 70% in central Manchester. A time series model of NO_2 concentrations at the Manchester Piccadilly Gardens site, an urban background site in central Manchester, was created using 5 years of meteorological information and the AURN NO_2 data (Appendix 6). Post lockdown, the model simulates the NO_2 concentrations based on meteorology and historical NO_2 values and is therefore an indicator of NO_2 in the absence of any reductions in traffic. The model predicts more than 3 times the observations for much of the lockdown period, consistent with reductions in traffic volumes of between 70 and 75% over the lockdown period in central Manchester. In a second model, the NO_2 data used to build the model was normalised by traffic volume and then the concentrations were re-weighted by the traffic counter information. In essence, this model assumes that all the NO_2 at Piccadilly Gardens can be explained by local traffic sources. The model reproduces the observations for much of the lock down period, though during periods of outflow from Europe and SE England, Manchester is a receptor of regional pollution and a large proportion (up to 60%) of the observed NO_2 cannot be explained by local sources. Similar model data is available from multiple other locations.

4. How might altered emissions of air pollutants over the next three months affect UK summertime air quality?

Photochemical production of ozone may become more efficient in urban areas. Observations of NO_x and O_3 in cities across the UK show marked decreases in NO_x and corresponding increases in O_3 during lockdown. As NO_x reduces from high values common in many towns and cities photochemical production may well become more efficient and lead to higher O_3 concentrations in summertime as temperatures and hence emissions of biogenic hydrocarbons and evaporative emissions increase. The observations from the CleanAir supersites will be valuable in constraining photochemical models of these changing environments.

Recent laboratory work demonstrates that secondary organic aerosol (SOA) production of biogenic and anthropogenic mixtures from VOCs are non-additive and hence current models are not equipped to capture SOA changes under future low NO_x conditions in summertime. Recent chamber work on the oxidation of mixtures of anthropogenic and biogenic VOC precursors of secondary organic aerosol (SOA) has indicated that the yield is non-additive. It is therefore very likely that a reduction in the emissions of anthropogenic VOC and of NO_x will lead to a change in the yield of SOA formation as the mixture becomes more biogenically-dominated and the NO_x reduces. The resultant change in PM mass will depend on the exact change in the VOC mixture, but the contributions from different SOA sources will not be linearly additive as assumed in any current AQ models that consider them. At the present time caution should be applied to any model predictions of future SOA changes in response to reductions in the NO_x and VOC mix.

5. Insights that can be gained from aerosol science on possible viral transmission mechanisms

Experience of air quality and knowledge of aerosol processes would suggest that aerosol transmission could result in direct deposition of the virus to the lungs is likely to be most significant in enclosed environments. While it has been hypothesised that ambient $PM_{2.5}$ may increase transmissibility, it is not obvious from an aerosol mechanical perspective how this would happen. Outdoors, aerosols rapidly disperse, which would reduce the concentration and thus likelihood of transmission, but they are known to linger indoors. While there is currently much focus on aerosol transmission in care settings, attention may also need to be paid to other enclosed environments such as shops and public transport. More research is needed at the interface between aerosol science and medicine to understand the viable aerosol viral emissions from mildly symptomatic or asymptomatic individuals, particularly through breathing, talking, etc.

The UK atmospheric aerosol community has instrumental capabilities and expertise to study the physical aerosols online and in real time. The Manchester team (with partners) is currently using NCAS AMOF instruments to study potential interventions designed to prevent aerosol transmission in a care setting. Online bioaerosol sensors previously used by that atmospheric community to study airborne spores, pollen and bacteria will be unlikely to detect viruses such as SARS-Cov-2, but may be of use to detect allergens that could confound COVID-19 symptom tracking or make individuals more likely to develop severe symptoms.

Appendix 1 Meteorology and Back Trajectory Analysis

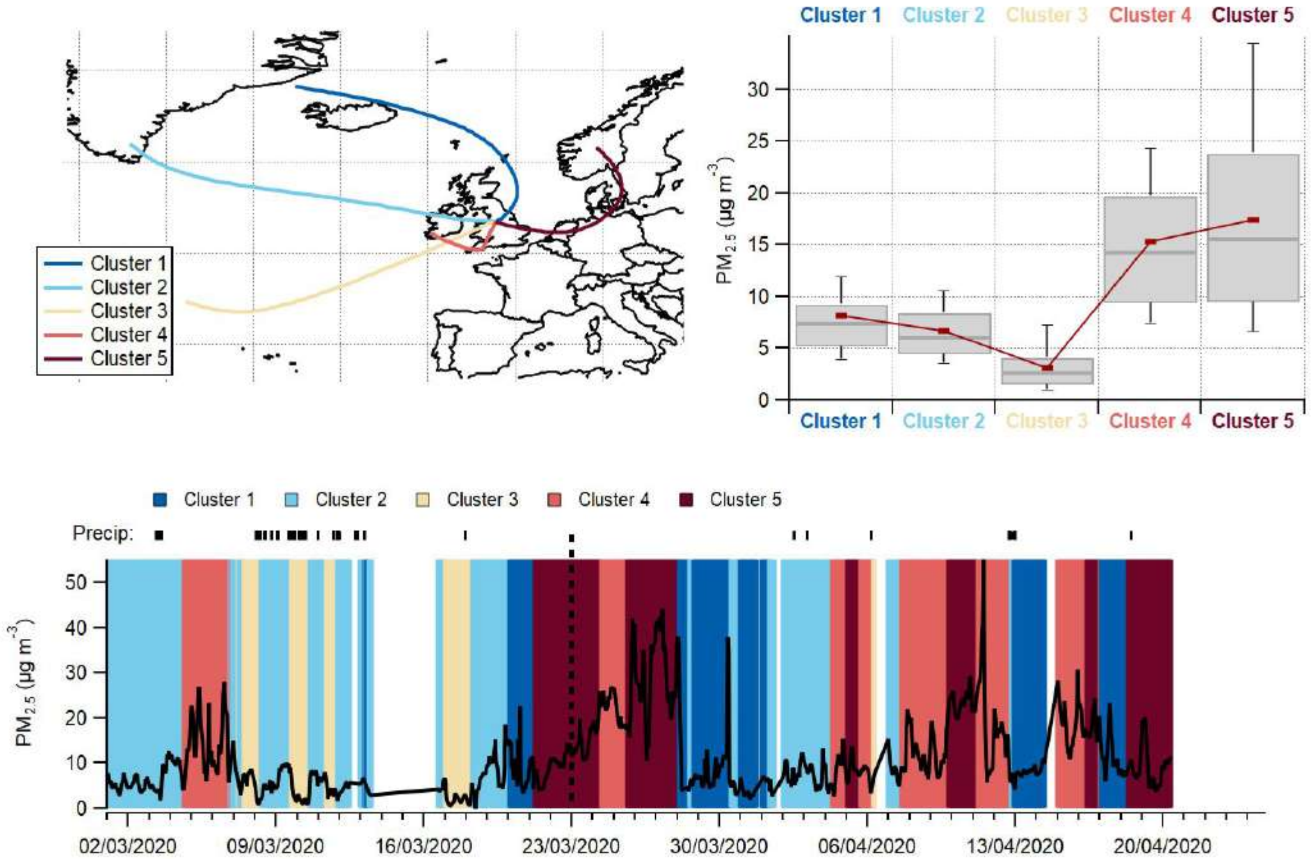


Figure 1.1 – Summary of back trajectory clusters ending in Manchester from March/April 2020

The differing levels of $PM_{2.5}$ pre- and post-lockdown have largely been determined by the different meteorological conditions causing different airmass origins. Figure 1.1 shows the results of 72-hour HYSPLIT back trajectories, which were initiated hourly from 500m AGL above the Manchester air quality supersite during March/April 2020, and divided up into 5 clusters. During March pre-lockdown the weather was dominated by a series of low pressure systems. Clusters 2 and 3 dominated, with moderate westerly winds and precipitation, and airmass origins over the Atlantic. Levels of $PM_{2.5}$ were in the range 1 – 9 µg m^{-3} in these conditions. Post-lockdown the airmass histories were much more easterly due to the prevalence of high pressure systems, with clusters 1, 4 and 5 dominating. Cluster 4 shows slack winds from the south of the UK, and cluster 5 light winds and airmasses originating over the Benelux region. These two clusters are associated with much higher levels of particulate pollution, with $PM_{2.5}$ generally in the range 8 – 30 µg m^{-3}. The high levels of pollution on the 24th – 26th March are associated with airmass origins over continental Europe. The lidar data for 25th March shows the various aerosol layers that were advected from continental Europe and subsequently mixed back into the air over Manchester (see Figure 1.2). This is a typical example of the mixing in of pollution from further afield. This mixing is driven by the heating of the ground as shown by the surface heat flux measurements in Figure 1.3.

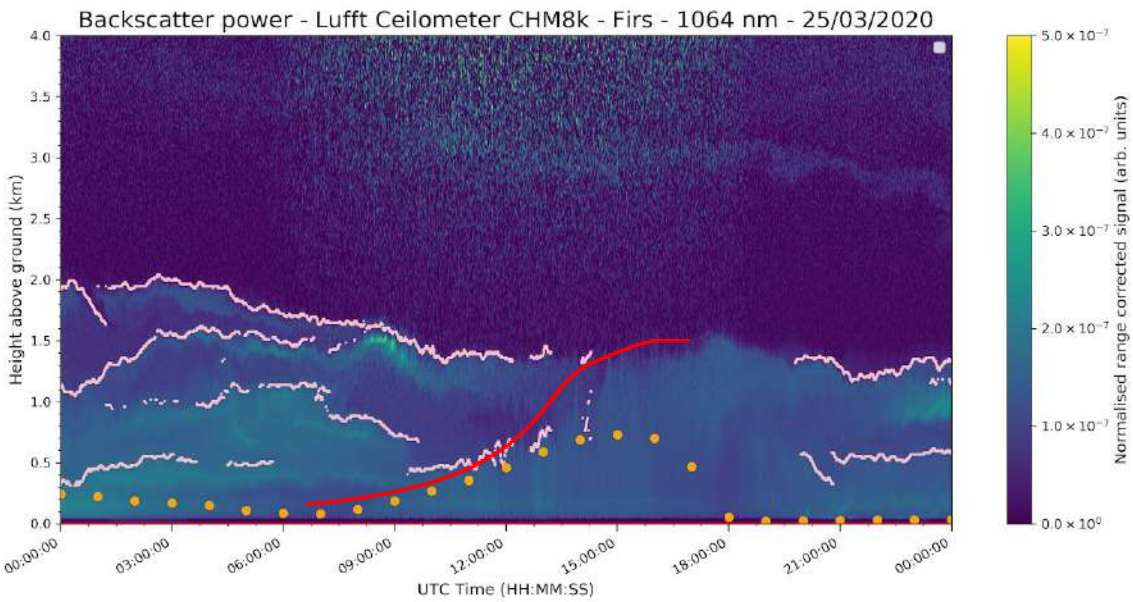


Figure 1.2 – The backscatter power as measured by the Lufft CHM8k ceilometer at the Manchester air quality supersite on 25 March 2020. Increased backscatter power signifies the presence of aerosol. The top of various aerosol layers is marked in pink. The red line shows the top of the planetary boundary layer (PBL) made visible by the thermals containing aerosol as the ground heats up. As the day progresses, the PBL grows and entrains the aerosol above it. For comparison, the PBL height as modelled by the ECMWF ERA5 data also is shown (orange dots).

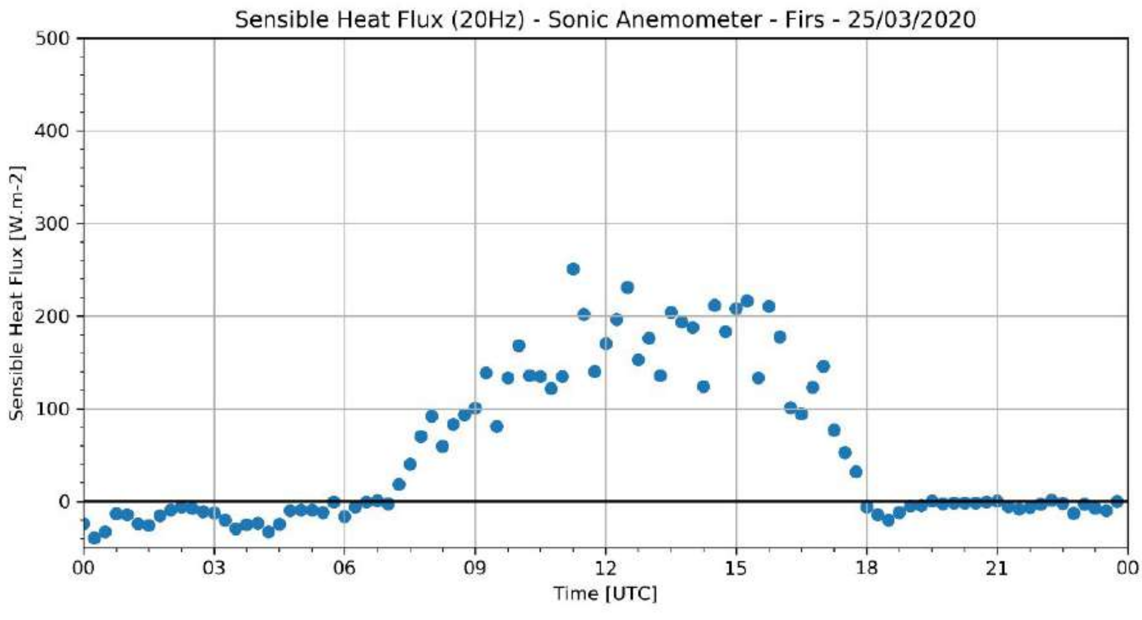
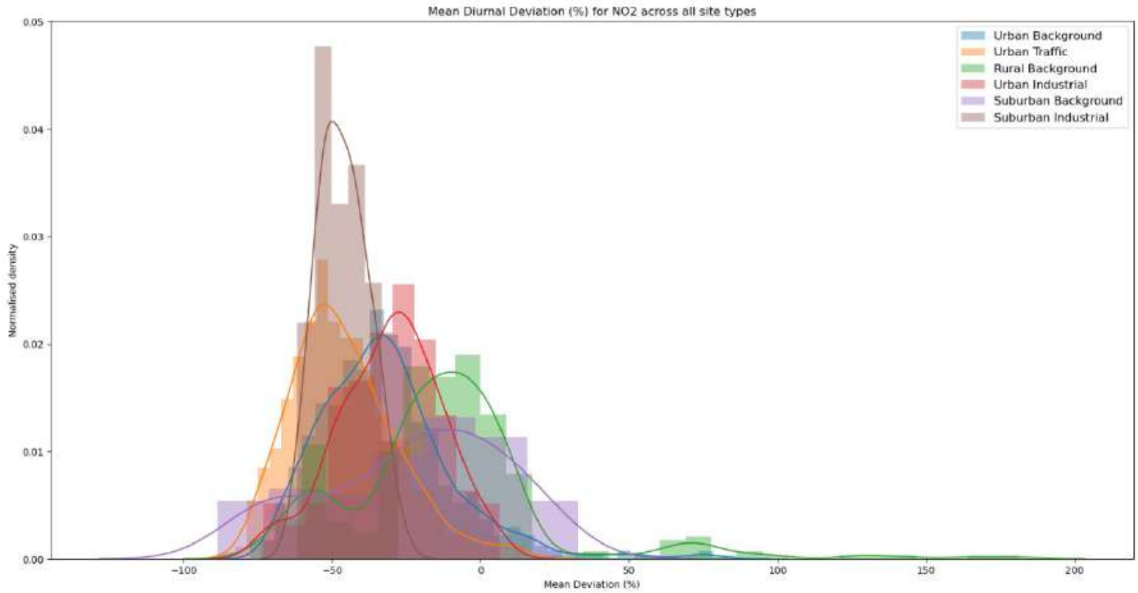
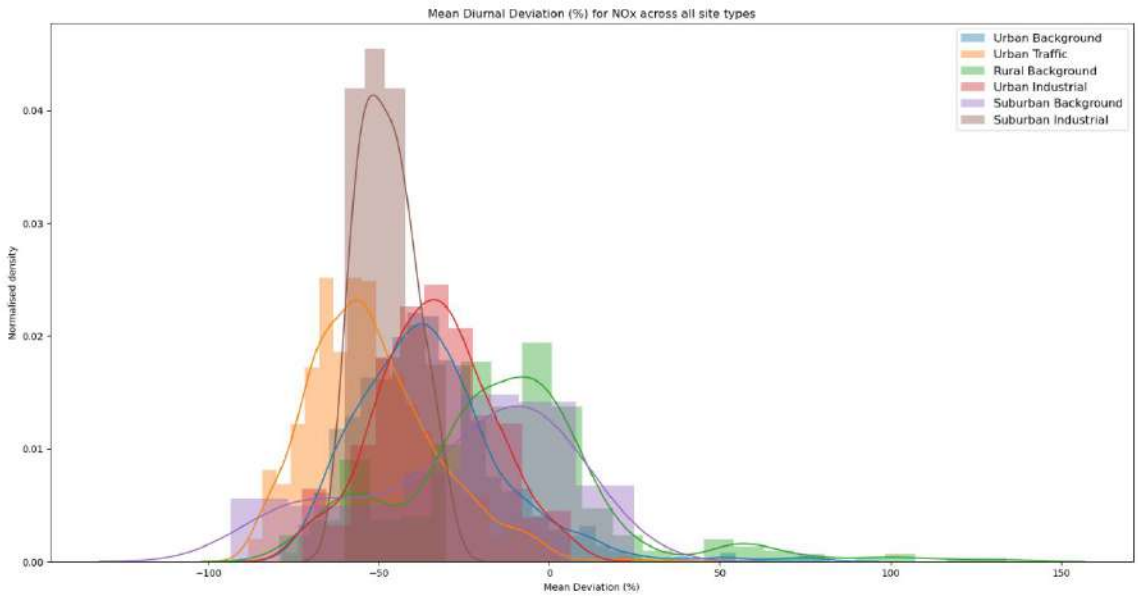


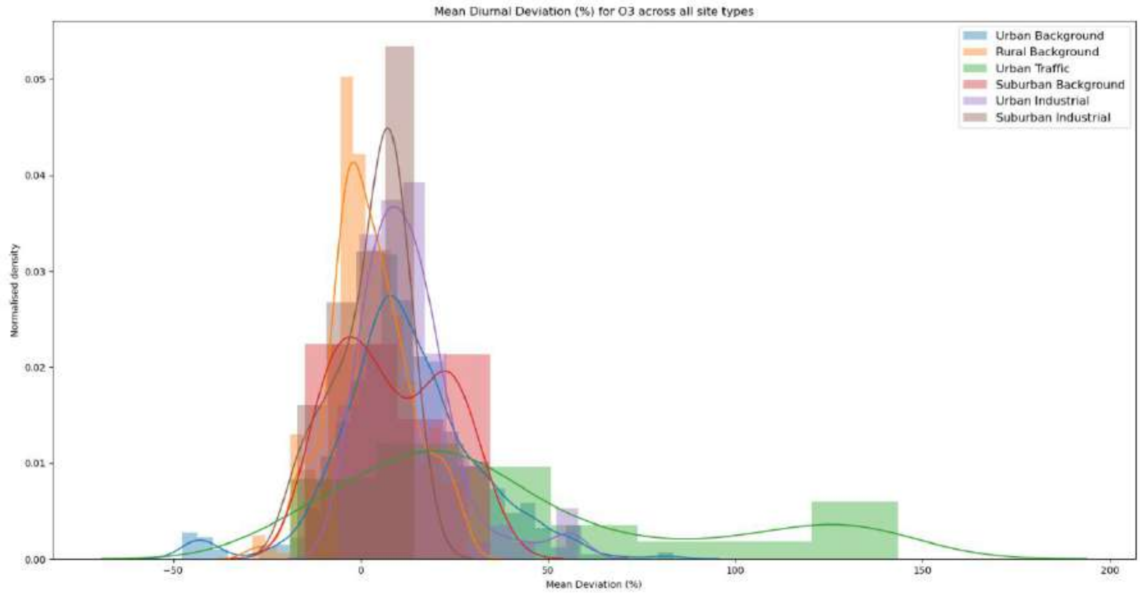
Figure 1.3 – Sensible heat flux - the amount of heat being transported from the surface into the atmosphere - as measured by the Gill WindMaster sonic anemometer at the Manchester air quality supersite

Appendix 2: AURN Network data

Figure 2.1: Histograms showing the distributions of the fractional deviation of the variation in the hourly mean values of NO_2, NO_x, O_3 and PM2.5 during April 2020 compared to the same hour in the monthly diurnal mean taken between 2015 and 2019. Each histogram represents data from all the sites in the AURN network in each site category (urban background, urban traffic, rural background, urban industrial, suburban background, and suburban industrial).






Mean Diurnal Deviation (%) for O3 across all site types
Urban Background
Rural Background
Urban Traffic
Suburban Background
Urban Industrial
Suburban Industrial
Normalised density
Mean Deviation (%)

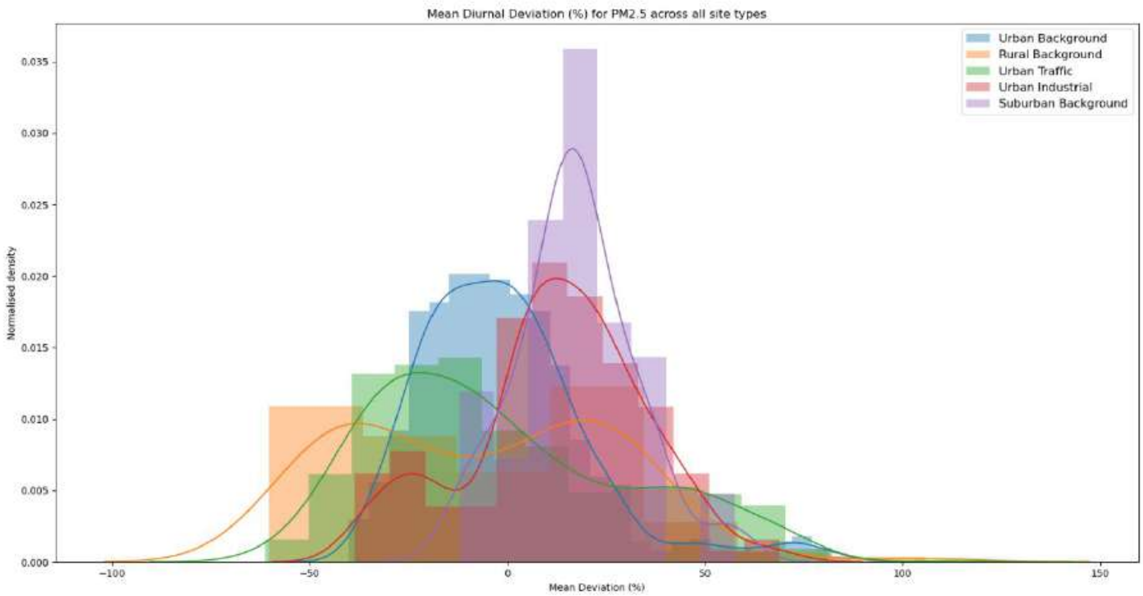
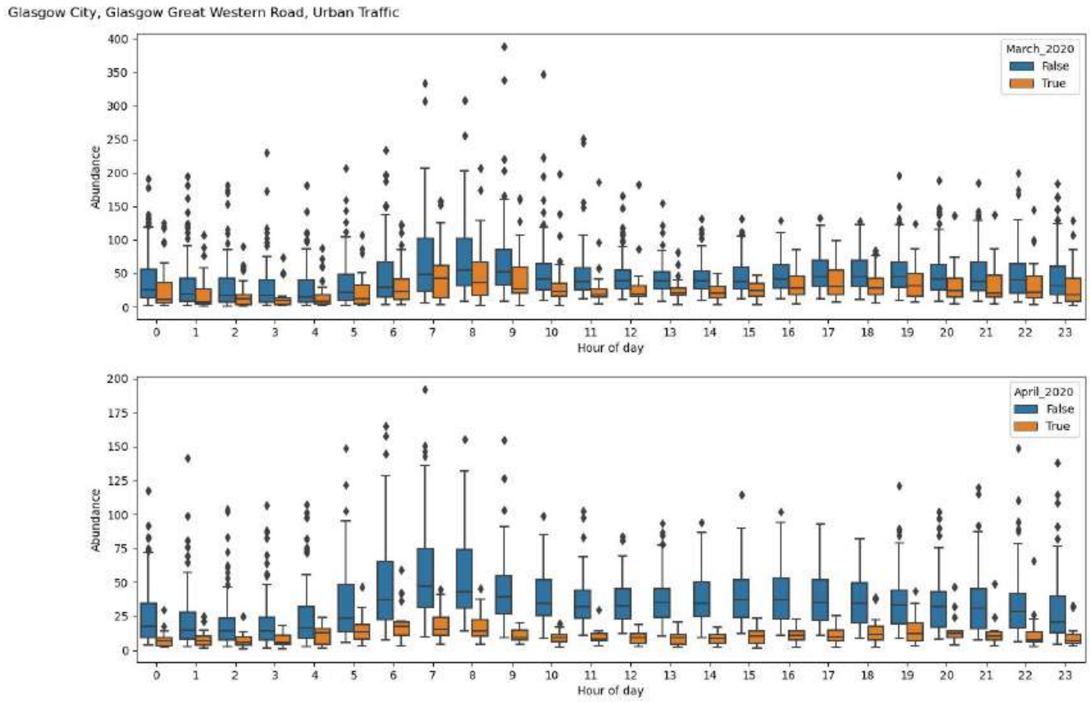
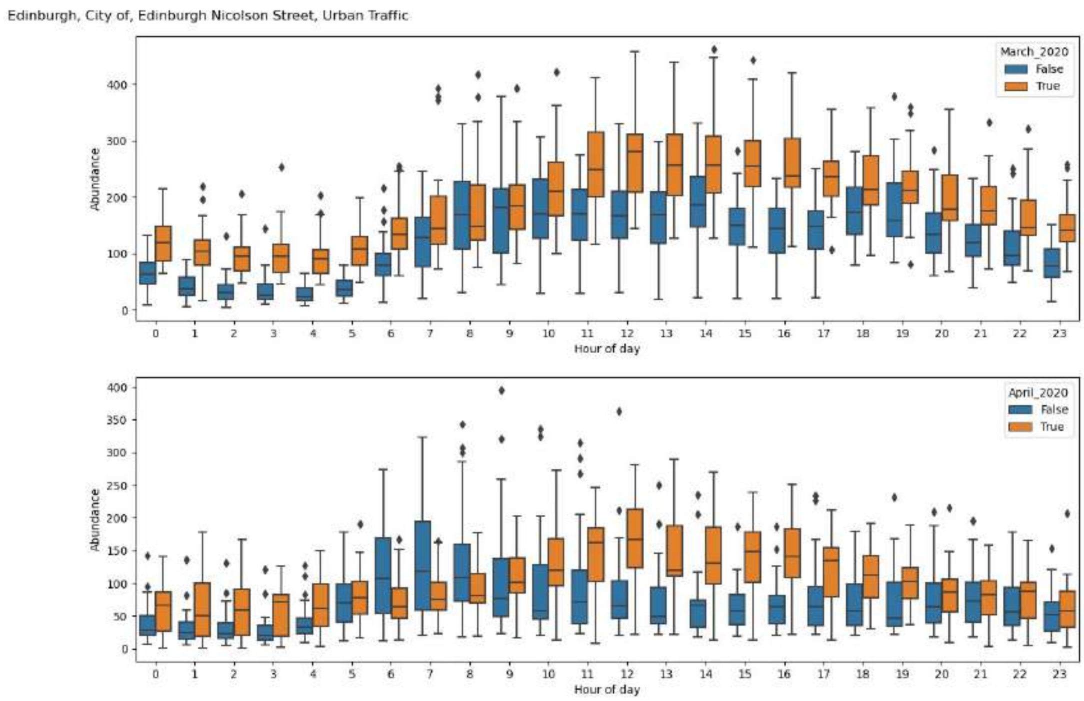
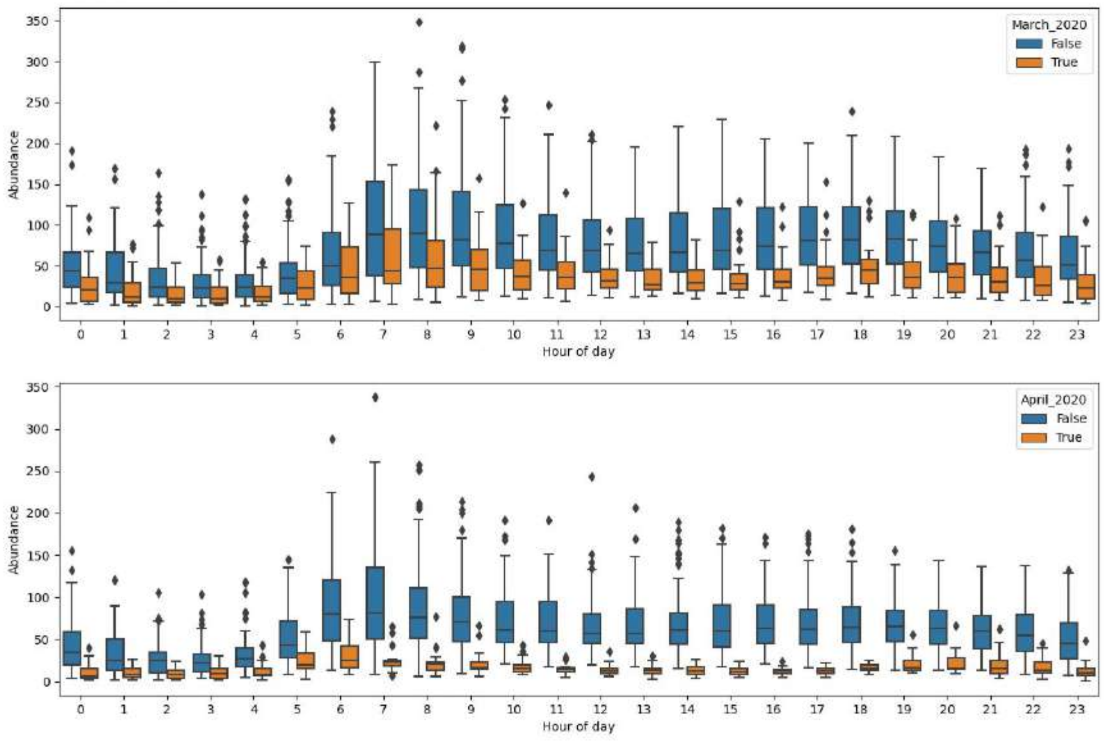

Mean Diurnal Deviation (%) for PM2.5 across all site types
Urban Background
Rural Background
Urban Traffic
Urban Industrial
Suburban Background
Normalised density
Mean Deviation (%)

Figure 2.2 (panels a, b, c and d) Examples of mean monthly diurnal cycles of NO_x for March 2020 and April 2020 (True) compared to the average of the previous 5 years (False) for 4 roadside sites: Edinburgh Nicolson Street, Glasgow Great Western Road, Leeds Headingley Kerbside and London Marylebone Road. Similar plots are available for all available AURN sites.





Leeds, Leeds Headingley Kerbside, Urban Traffic



Westminster, London Marylebone Road, Urban Traffic

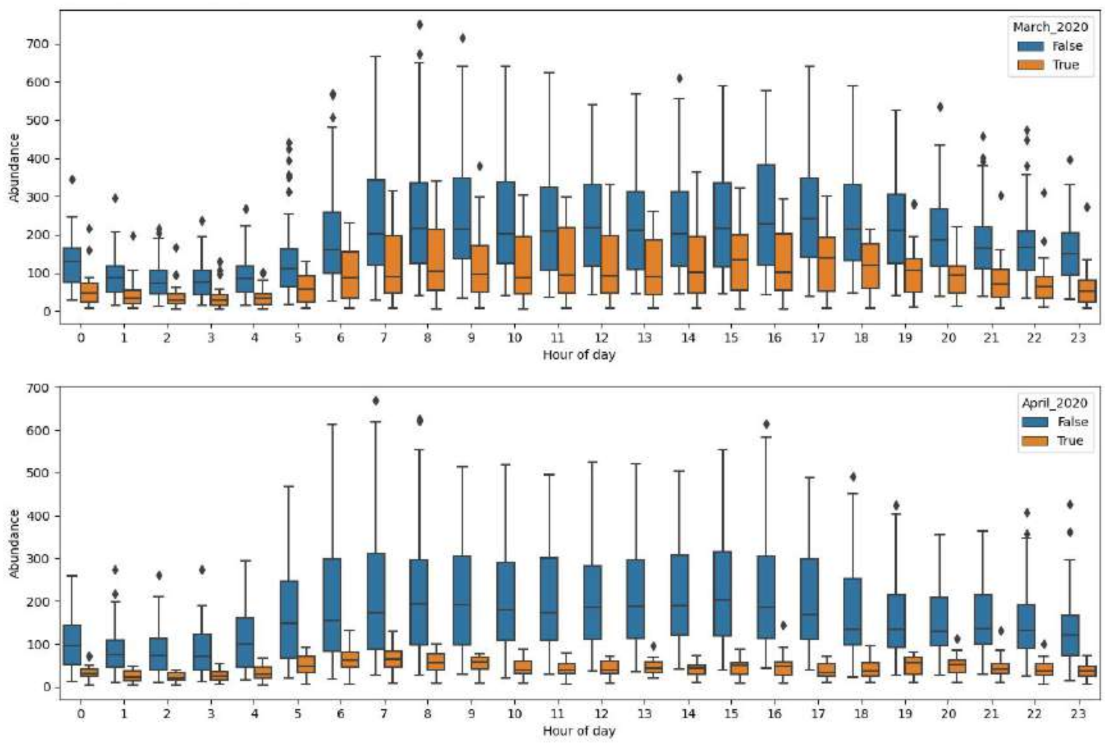
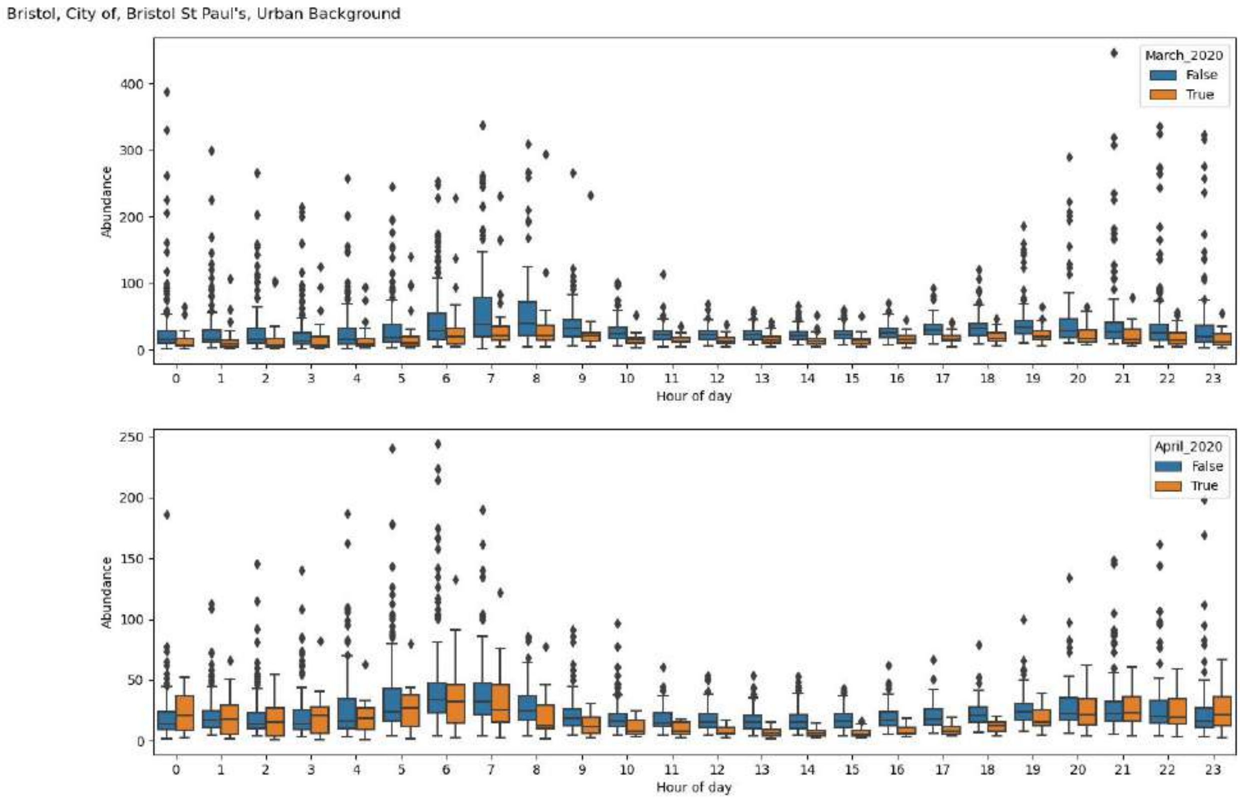
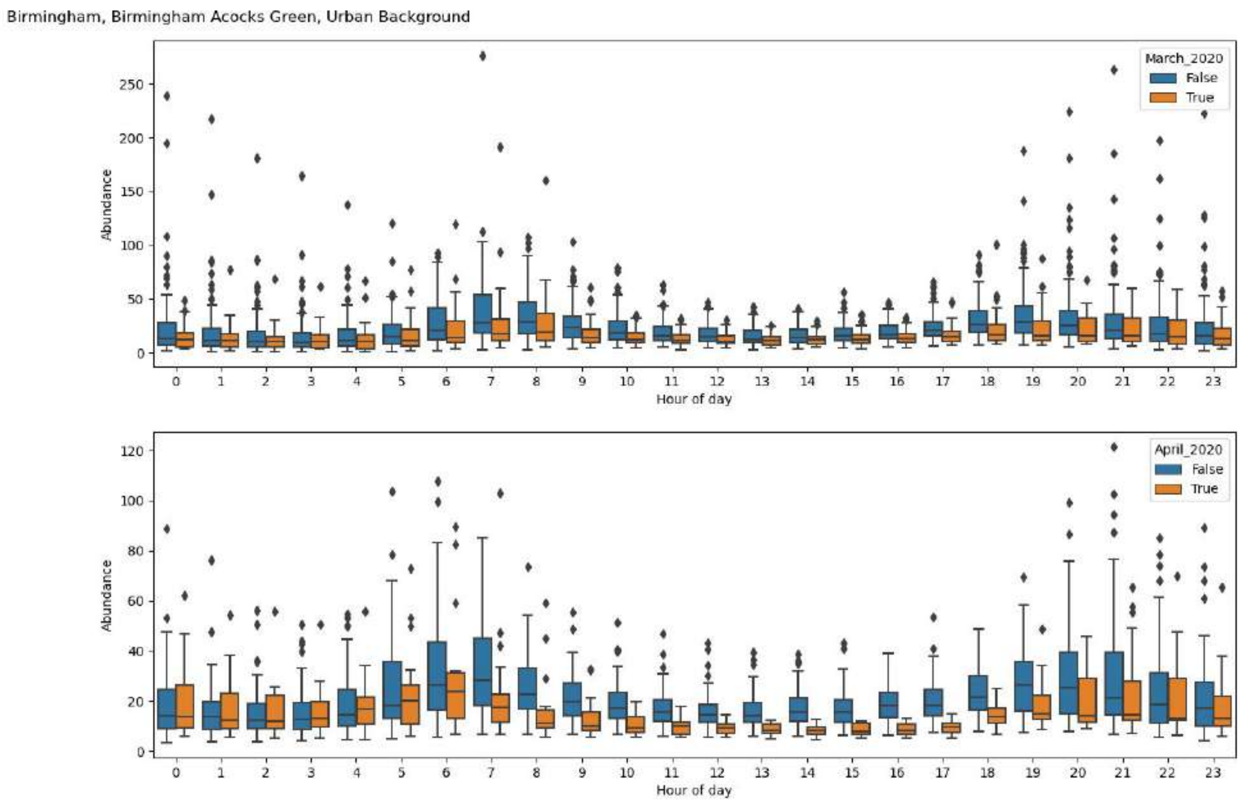
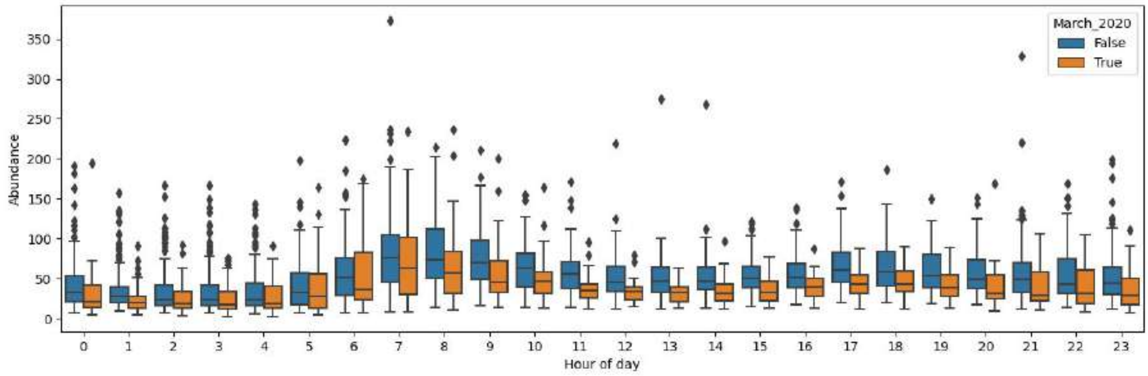


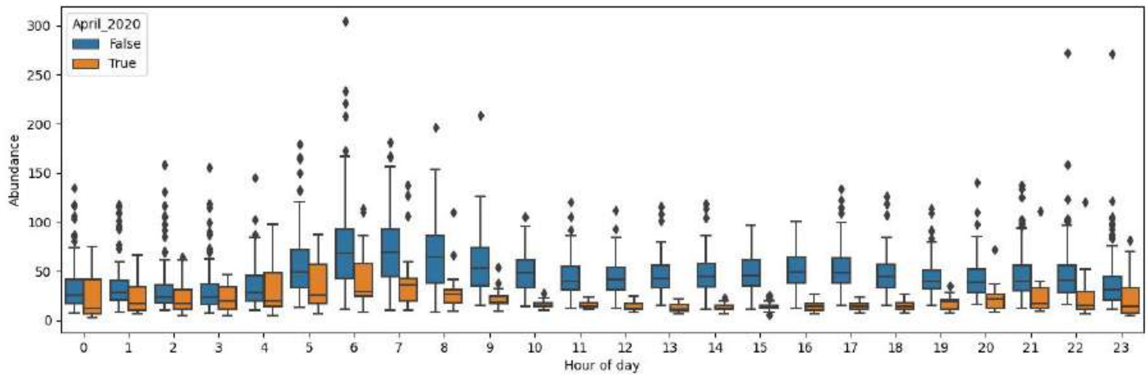
Figure 2.3 (panels a, b, c and d) Examples of mean monthly diurnal cycles of NO_x for March 2020 and April 2020 (True) compared to the average of the previous 5 years (False) for 4 urban background sites: Bristol St Pauls, Birmingham Acocks Green, Manchester Piccadilly, Nottingham Centre. Similar plots are available for all available AURN sites.



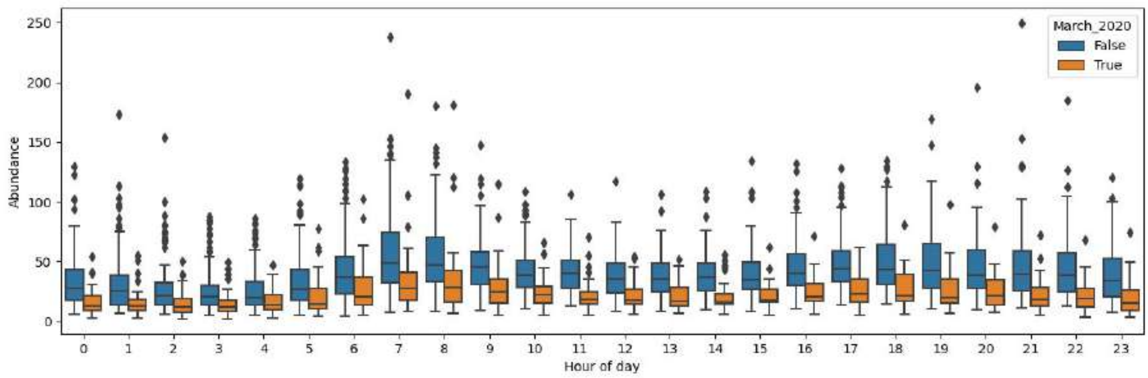


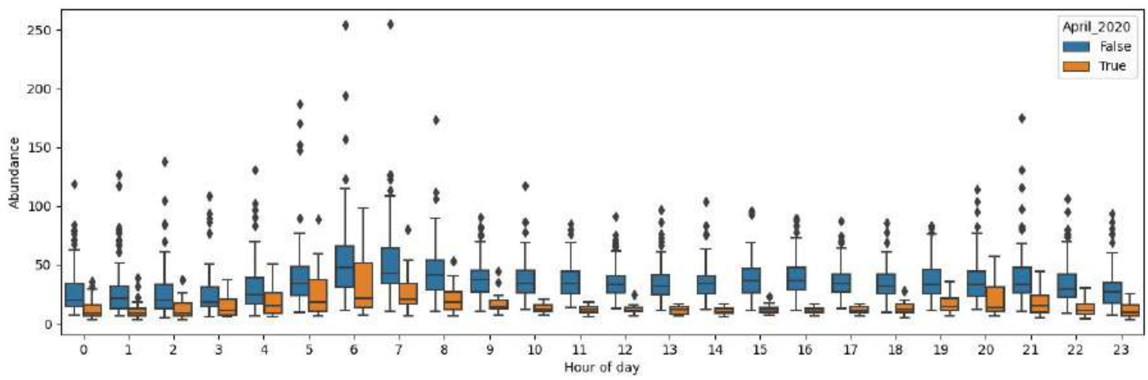
Manchester, Manchester Piccadilly, Urban Background





Nottingham, Nottingham Centre, Urban Background





Appendix 3: Summary of data from the Manchester Air Quality Supersite (MAQS) during the COVID lockdown

The University of Manchester Air Quality Supersite (MAQS) is located on the University Fallowfield campus (53º26'39''N, 2º12'52''W) and is an urban background site located approximately equidistant between the Manchester Piccadilly Gardens and Sharston AURN sites. It was supported by a capital award from NERC and receives ongoing funding from the UKRI CleanAir programme. It has been operational since mid 2019. Measurements have been made continuously before and since the lockdown period except for periods of instrument downtime for calibration, maintenance and failure.

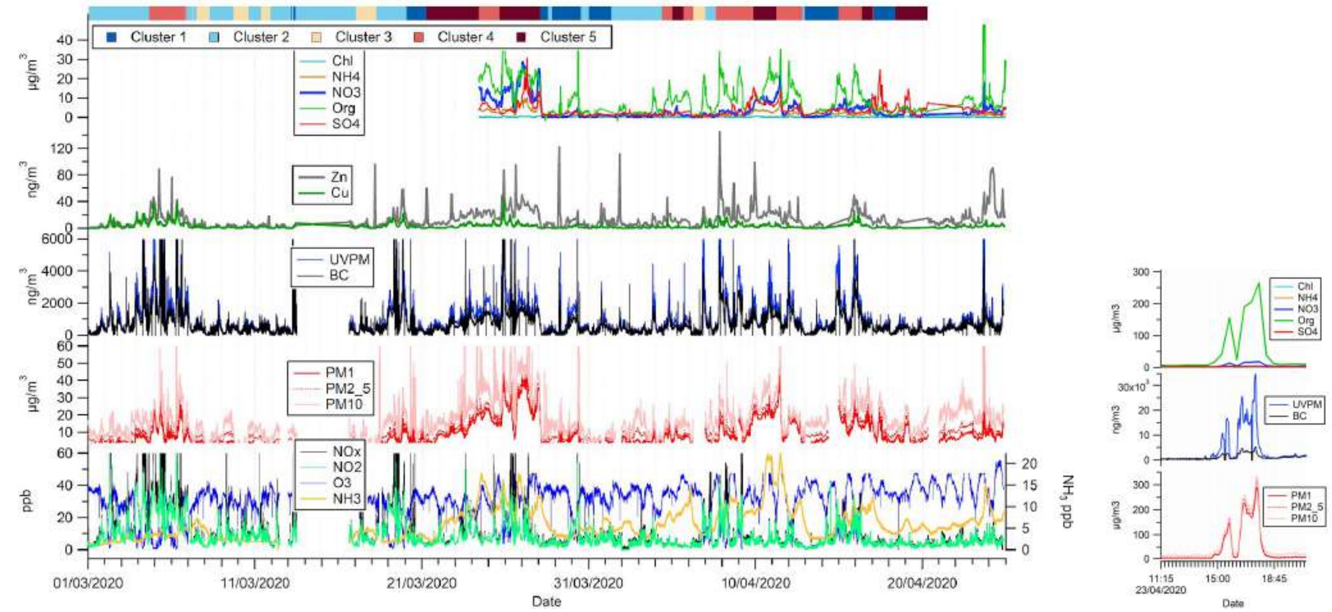


Figure 3.1 Time series of gas and particulate measurements taken at the Manchester Air Quality Supersite. The panels from the top downwards show: PM2.5 component mass loadings as measured by the ACSM; zinc and copper concentrations in PM2 as measured by the XACT; UVPM and BC measurements from the AE-33 Aethalometer; PM1, PM2.5 and PM10 measurements using the Palas Fidas; NO_2, NO_x, O_3 and NH_3 mixing ratios. The coloured top bar shows the 5 airmass clusters determined using the back trajectory analysis in appendix 1 using the same colour scheme. The inset figure shows the ACSM component mass loadings, the Aethalometer data and the PM measurements during the afternoon of 23rd April.

Figure 3.1 shows the time series of a wide range of particulate properties and gaseous pollutants before and during the lockdown at the MAQS. While the primary gas phase pollutants respond to the changes in traffic, with some enhancement during the periods of long range transport, ozone increases throughout the period. This is to be expected during the spring time and also as NO levels reduce in urban locations as a result of reduced emissions. There is a marked decrease in BC as well as NO_x since 23rd March though also clear variation with air mass history. The total PM loading and the main components of PM2.5 (organic matter, ammonium, nitrate and sulphate) all show big enhancements during the easterly and southerly air mass periods (clusters 4 and 5). This is mirrored in the time series of ammonia which is greatly enhanced in Manchester during the periods of regional pollution and reaches mixing ratios of more than 20 ppb. The Cu and Zn data show significant variability, there is some indication that Zn increases in concentration during the periods of easterly and southerly winds, whereas the Cu seems to be more locally generated. Source apportionment is discussed in more detail in Appendix 5.

There was a moorland fire at Rakes Moss on the A628 Crowden to Glossop on Thursday around 13:00. While the fire was controlled over the course of the day, it had a significant impact on air quality. The inset in figure 3.2 shows the very large enhancements in PM and key components through this period. A maximum concentration of PM close to 300 µgm^{-3}, nearly all of which was PM1 and over 85% was organic mass. We would caution include this period in any analysis of the effects of the lockdown but it serves to indicate that major air quality events can occur during periods of reduced activity.

The pre- and post-lockdown concentrations of pollutants from the MAQS have been compared using the cluster analysis in Appendix 1 to gain an estimate on the impact of changing emissions, while limiting the effect of differing meteorology. The pre and post lockdown concentrations of NO_x, O_3 and PM2.5 are shown for each of the back trajectory clusters identified in Appendix 1 in figure 3.2. The changes in both NO_x and O_3 are consistent between the different meteorological conditions represented by the clusters- post-lockdown levels of NO_x are 40 – 80% lower, and 10 – 110% higher, other than for the transported pollution in cluster 5 which showed a small increase in O_3. Some caution should be used here, as most clusters had a limited amount of data either before or after the lockdown (i.e. there was little overlap between the meteorology before and after the lockdown). Cluster 2 is the most robust and has several days' worth of data both before and after the lockdown, and shows a 60% reduction in NO_x and a 23% increase in O_3 for westerly winds. Cluster 4 shows the highest levels of NO_x and lowest O_3 pre-lockdown under stagnant winds, and under these conditions the decrease in local NO_x emissions has lowered NO_x and increased O_3 to levels similar to those under westerly winds.

The trends in the gas-phase pollutants are not mirrored by similar changes in particulates. $PM_{2.5}$ levels have remained fairly similar when accounting for the different meteorological conditions represented by the different clusters. The most robust cluster (cluster 2) had a 4% increase in mean concentrations between pre- and post-lockdown.

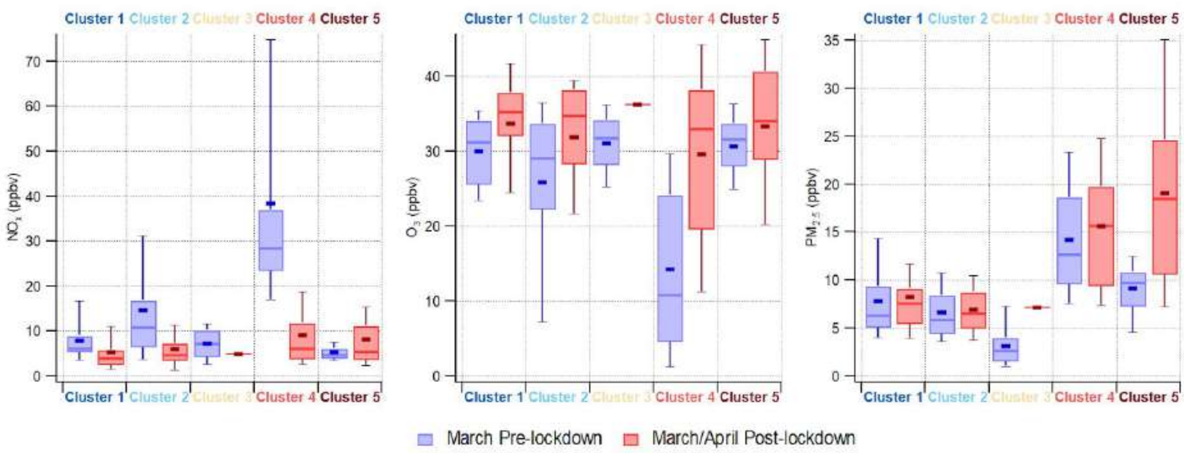


Figure 3.2 - Average concentrations of key pollutants before and after the 23rd March lockdown

Appendix 4 Regional Transport of Pollution and the Dominance of Secondary Particulate in PM2.5

The cluster analysis in Appendix 1 highlighted the high loadings of PM2.5 during clusters 4 and 5, the periods of stagnant UK air and European outflow respectively. The same cluster analysis was used to investigate the composition of PM2.5.

Figure 4.1 shows average composition of PM2.5 measured by the ACSM in each of the different clusters. These data were only available from the 24^{th} March (post-lockdown). All components increased in the polluted clusters 4 and 5, and organic aerosol was always the largest component, but large increases in nitrate and ammonium in clusters 4 and 5 suggest a significant secondary aerosol fraction associated with aged pollution during the periods with the highest aerosol loadings and are not dissimilar to those previously measured in Manchester during similar conditions (eg Martin et al., Atmos. Environ., 2011, 10.1016/j.atmosenv.2011.05.050).

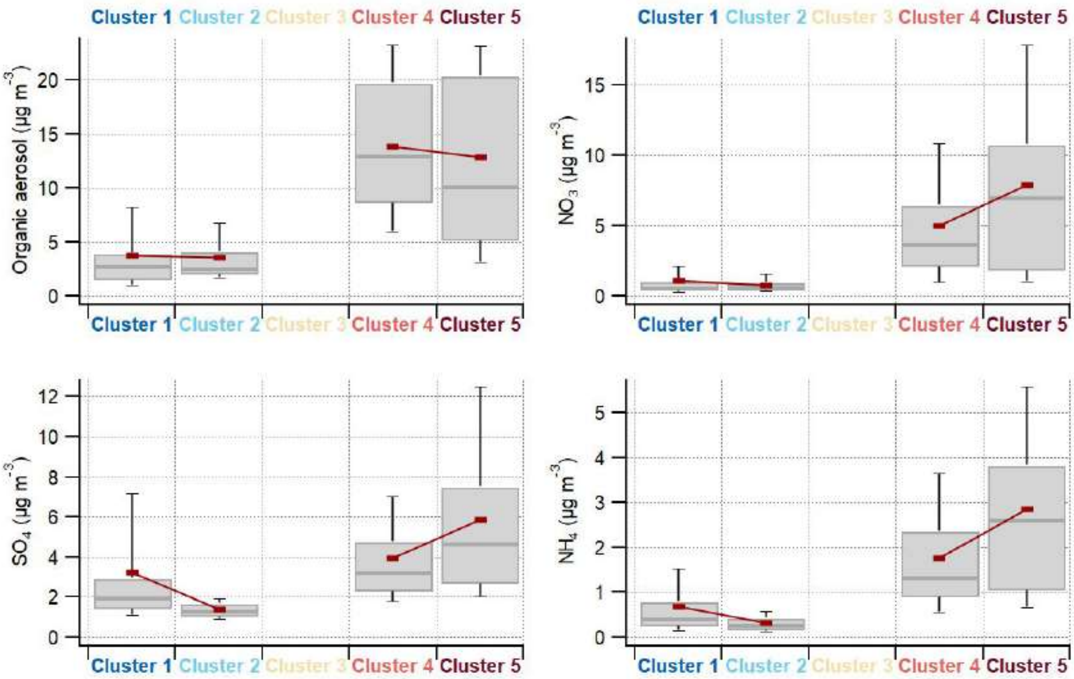


Figure 4.1- Average concentrations of aerosol chemical species measured by the ACSM, divided by cluster. Post-lockdown data only.

Pre-lockdown, the clean air masses travelling from the Atlantic mean that pollution levels were largely determined by local emissions. Post-lockdown, easterly and southerly air masses mean that Manchester is a receptor site for pollution transported from continental Europe and the south of the UK, and slack winds also allowed local pollution to build up. PM2.5 has a significant secondary fraction that is much less significant under the prevailing westerlies.

The ACSM data can be used to apportion sources of organic aerosol (OA) using the Multilinear Engine (ME-2) tool for receptor modelling. In this instance, data was available from around the time of the lockdown onwards; it was not available due earlier to a technical issue. While normally this technique can report on sources such as cooking and domestic burning, in this instance, the most satisfactory solution was obtained with only two factors, hydrocarbon-like (HOA), likely due to traffic emissions, and more oxidised (MO-OOA), which can be indicative of secondary aerosols. These can in turn be used to estimate the relative proportions of primary and secondary organics. However, it must be noted that this does not rule out the presence of domestic burning, as this could have been included within one or both of the identified factors. Similarly, cooking emissions may have been included in the HOA factor. More factors may well be extractable at a later date as more data becomes available, but the current two factor solution can be taken as the ‘best estimate’ obtainable at this stage. The results of the PMF are shown in figure 4.2.

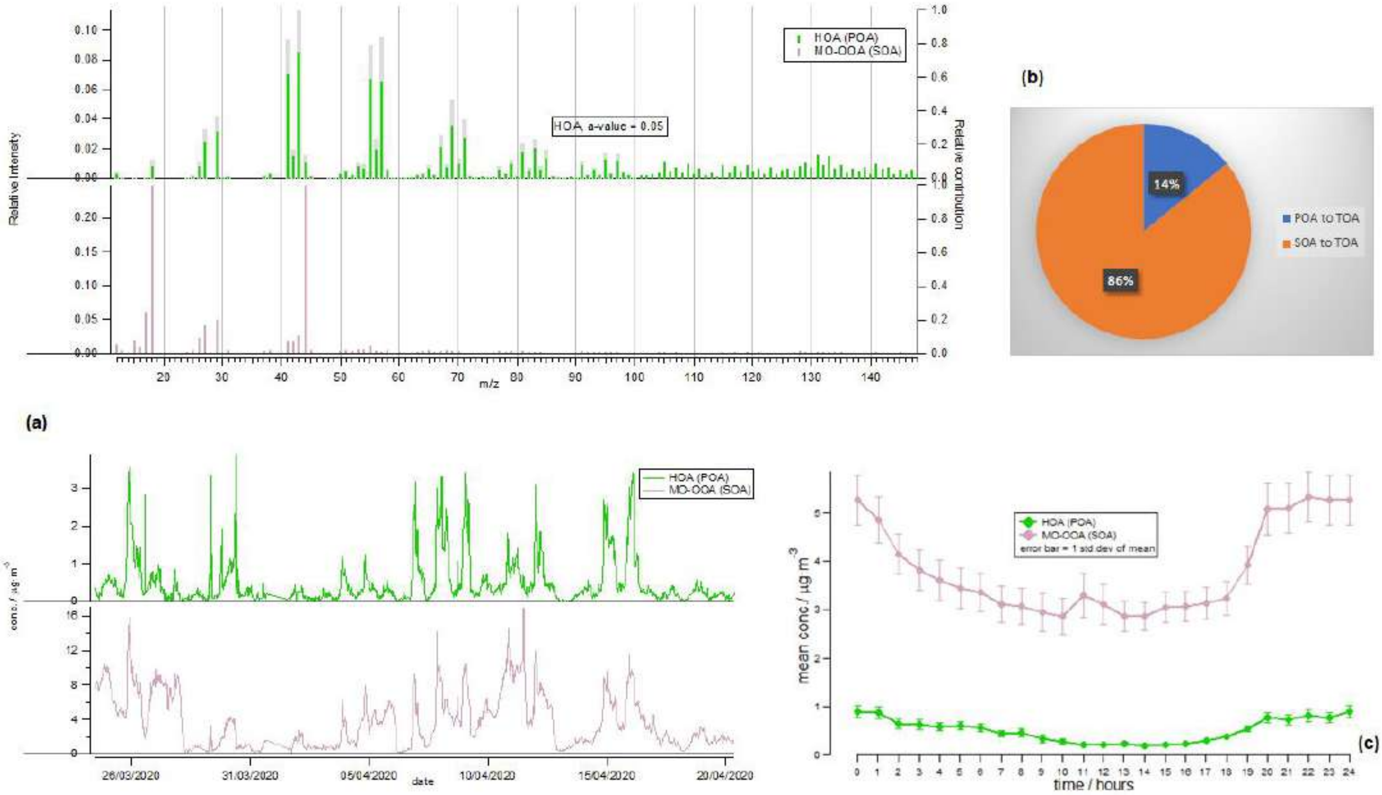


Figure 4.2 (a) The mass spectral fingerprint and time series of the two factors from the PMF of the organic mass spectral time series, Hydrocarbon-like Organic Aerosol (HOA) and More-Oxidised Oxygenated Organic Aerosol (MO-OOA); **(b)** The fractional contribution of Primary Organic Aerosol (POA) and Secondary Organic Aerosol (SOA) to the total organic aerosol. POA is represented by the HOA factor and SOA by the MO-OOA factor; **(c)** the average diurnal cycle of the two factors.

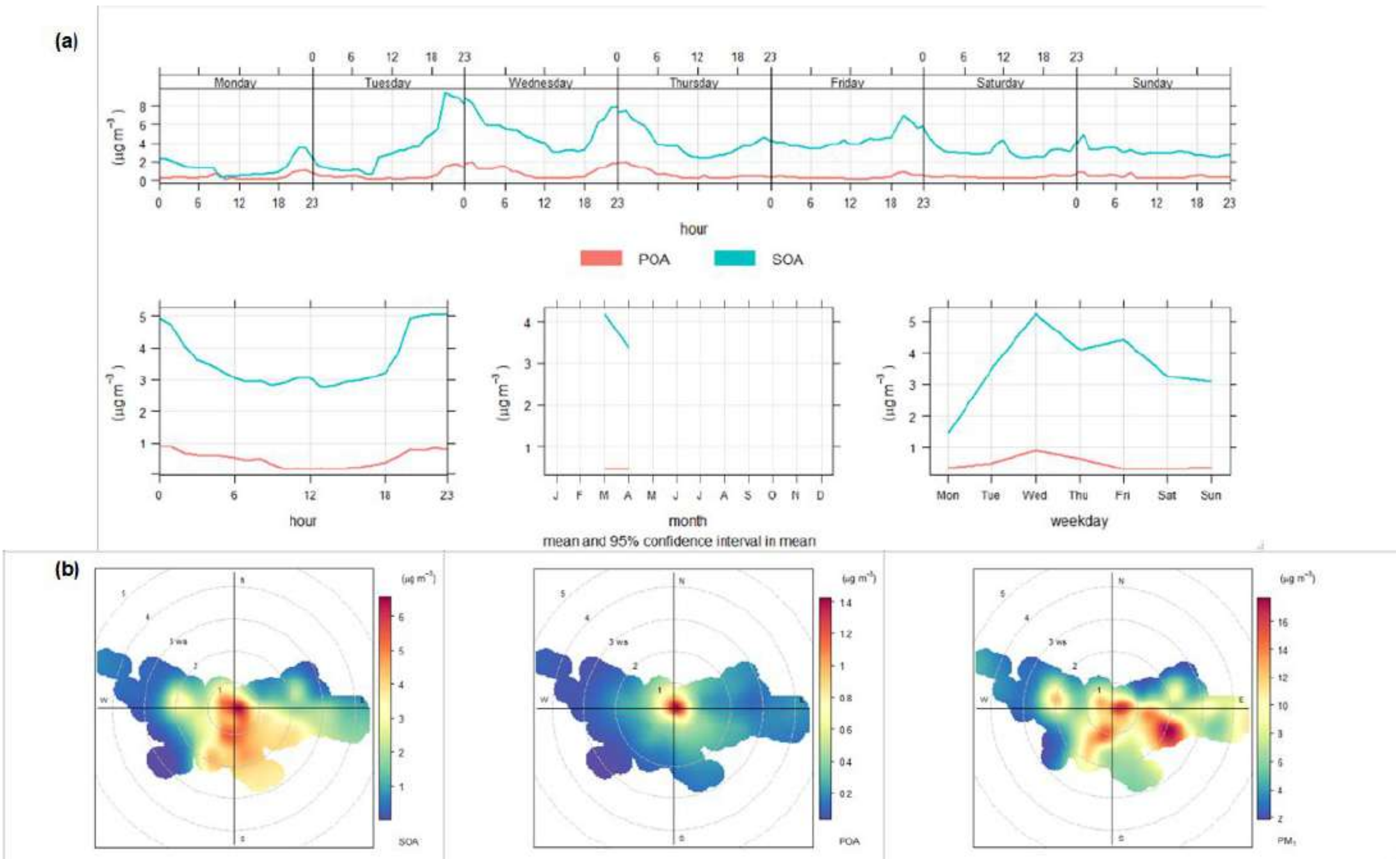


Figure 4.3 (a) The average weekly time variation, diurnal variability, monthly mass loading and average daily loading of the POA and SOA factors during April 2020; (b) polar plots showing the distribution of the concentration of the POA and SOA factors and the total with wind speed and direction during April 2020. This analysis clearly shows the dominance of SOA to the total organic aerosol measured, making up more than 80% of the total organic mass throughout the period. These results also clearly link the periods of elevated SOA to the periods when air masses were transported from the south and east and Manchester received substantial regional pollution from across the southern UK and Europe. Local sources only became a significant contribution when wind speeds were low and even under these circumstances they are not the main contribution.

We used the same clustering analysis to develop criteria to classify the expected pollution conditions based on local wind speed and direction, and applied these to $PM_{2.5}$ data from the Piccadilly Gardens measurement site. The criteria were broadly similar to those suggested by a previous analysis by Martin et al. (2011) investigating multi-year pollution trends in Manchester, and essentially separated conditions when the winds were either slack or from the south or east from all other wind conditions. These two criteria separate polluted conditions from clean conditions in the city. The spring season (March-May) AURN data from Piccadilly Gardens were selected for the years 2015 – 2020 until 22nd March 2020 as being representative of normal conditions and these were compared with post-lockdown 2020.

The results are shown in Figure 4.4. There is no evidence of significant reductions in $PM_{2.5}$ in Manchester due to the lockdown, compared to normal levels from this analysis in either the clean or polluted conditions. This suggests that there has been no detectable reduction in the regional contribution to secondary aerosol pollution during heavily polluted periods as a result of lockdown. Assessing such changes is best done with a chemical transport model tested against measured data. However, it should be noted that while some regional models carry a reasonable description of ammonia and nitrate, secondary organic aerosol remains a challenging atmospheric modelling problem. Hence, while the short data period may offer limited information to establish whether or not secondary aerosol production has changed or not from observations alone, it does provide at least a constraint on any lack of change of regional aerosol pollution.

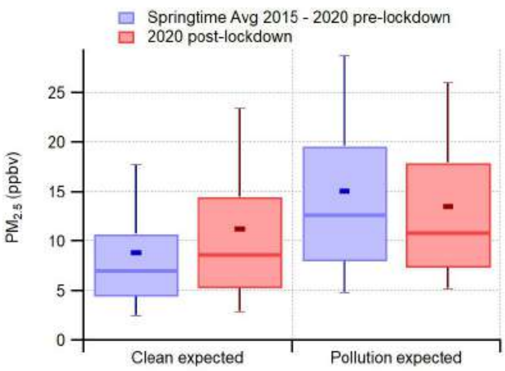


Figure 4.4 - Average concentration of PM2.5, comparing pre-lockdown spring measurements from 2015-2020 (blue) to post-lockdown 2020 (red) in wind conditions that generally produce clean or polluted air

Appendix 5: Source Apportionment

The Aethalometer (AE-33, Magee Scientific) measured BC and UVPM. The time series of these data are shown in figure 5.2 alongside the local meteorological parameters measured at MAQS and the planetary boundary layer height derived from ECMWF ERA5 (figure 5.1). The Aethalometer model (Sandradewi et al., 2008) was applied to identify the contribution of wood burning (wb) and fossil fuel (ff) to BC concentrations. The default absorption Angstrom exponent (AAE) values were applied AAE_{wb} = 2.0 and AAE_{ff} = 1.0. to estimate the absorption coefficient for wood burning at 470 nm (*babs_470wb* in Mm^{-1}) and absorption coefficient for fossil fuel at 950 nm and (*babs_950ff* in Mm^{-1}). It is worth mentioning that due to the improved sampling settings, the AE-33 instrument eliminates the need for filter loading correction.

As was shown in appendix 3, there was a moorland fire at Rakes Moss on the A628 Crowden to Glossop on Thursday around 13:00. While the fire was controlled over the course of the day, it had a significant impact on air quality. While the BC concentrations were low (5 μgm^{-3}) compared to the episode with high pollutant concentrations over week 1 where 20 μgm^{-3} of BC were observed (Figure 5.2), this moorland fire period provided a clear signature for wood burning as a source of BC source. *babs_470wb* values of 330 Mm^{-1} were observed during the moorland fire compared to 100 Mm^{-1} during Week 1.

Meteorology plays an important role in influencing the changes of concentrations. High BC during week 1 are related to low temperatures and low concentrations in week 2 with high wind speeds. As a result of the changes in meteorology, it is possible to see the changes in pollution resulting from changes in PBLH during the course of the day, with high PBLH being related to low aerosol concentrations and vice versa. This is similar for other pollutants, particularly those with emission sources within the city.

Similar to bulk PM loadings, there is no clear decrease of black carbon concentrations after the lockdown (Figure 5.3), however there is a change in the diurnal behaviour of wood burning (babs_470wb) and fossil fuel (babs_950ff, mainly related to traffic emissions) as shown in figure 5.4. Before the lockdown there is a typical diurnal fossil fuel trend characteristic of traffic emissions with high concentrations in the morning and in the evening. After the lockdown the morning peak decreases and there is a higher evening peak, with a higher contribution of wood burning compared to fossil fuel.

Figure 5.5 shows diurnal cycles for each week through the lockdown period from the start of March for several metals and NO_x, diurnal profiles of BC and UVPM from figure 5.4 are repeated for comparison. Potassium, a widely recognised marker for wood burning, shows increases in concentration during the period of lockdown, particularly during the late afternoon and evening period. This is consistent with the *babs_470wb* signature. Zinc shows similar week to week variability and diurnal cycle change as potassium and the *babs_470wb.* These data strongly indicate that a major source of Zn mass in particulate is wood burning. The variability in Cu follows the NO_x signature and the component of BC associated with traffic. Brake wear has previously been identified as a source of particulate Cu and the MAQS data are consistent with these earlier observations.

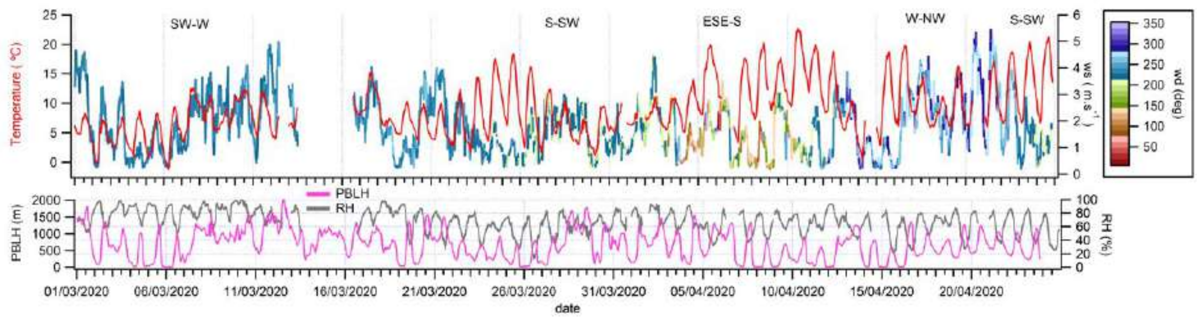


Figure 5.1. Meteorology parameters of the Firs site. PBLH data was downloaded from ECMWF ERA5 for the coordinates 53.5 N -2.2 W. PBLH is hourly data and the other variables are 30 min averages.

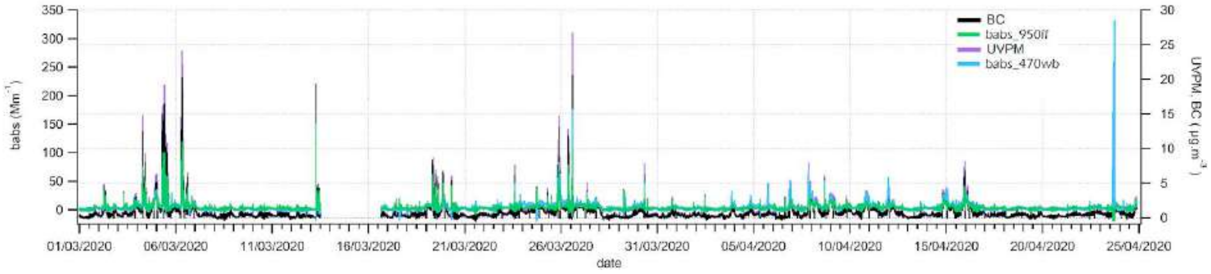


Figure 5.2. Time series of Aethalometer measurements (BC and UVPM) and Aethalometer model outputs (*babs_470wb* and *babs_950ff*).

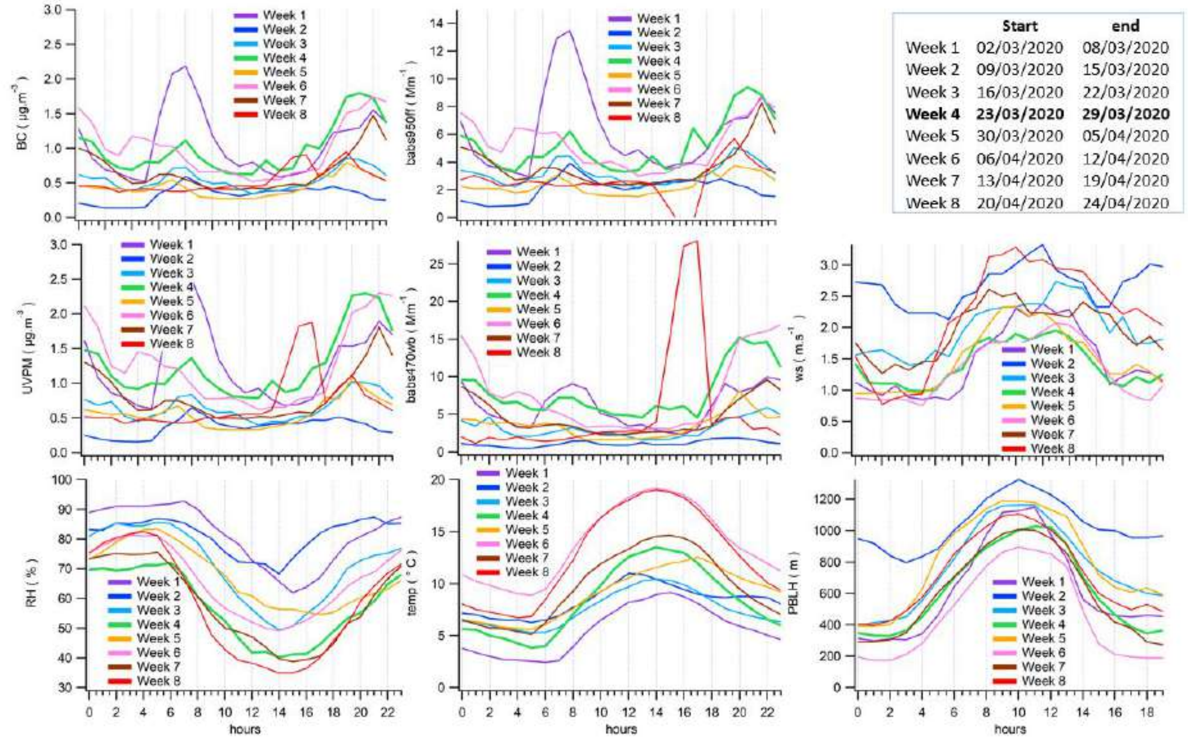


Figure 5.3. Diurnal plots of Aethalometer and meteorology data from the Firs site. Week 4 is when the lockdown started.

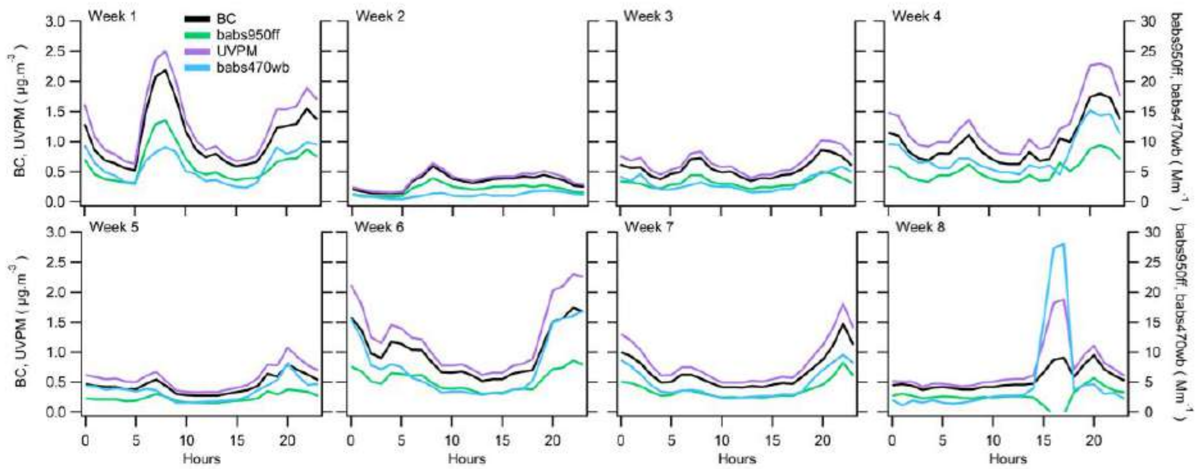


Figure 5.4. Diurnal plots of BC and UVPM with the Aethalometer model outputs.

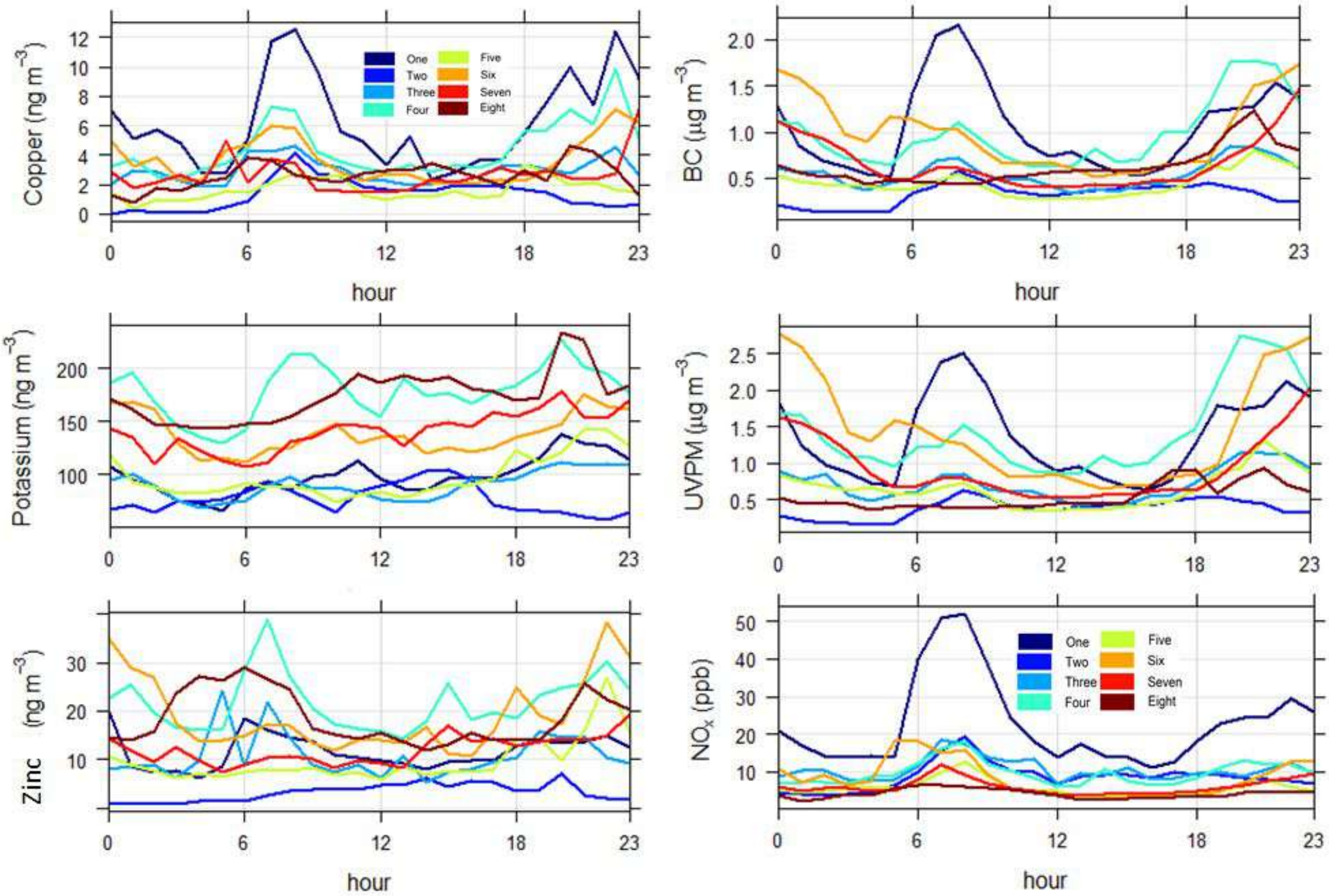


Figure 5.5 Average diurnal cycles of Cu, K, Zn, BC, UVPM and NO_x at MAQS for each week through the lockdown period. Week 1 begins on Monday 2nd March 2020, 7 subsequent weeks are shown. Week 4 (23rd March 2020) is the start of the lockdown period.

Sandradewi, J., Prévôt, A. S. H., Szidat, S., Perron, N., Alfarra, M. R., Lanz, V. A., Weingartner, E., and Baltensperger, U. R. S.: Using aerosol light absorption measurements for the quantitative determination of wood burning and traffic emission contribution to particulate matter, Environmental Science and Technology, 42, 3316-3323, 10.1021/es702253m, 2008.

Appendix 6 Separating weather influences and emissions reduction using time series modelling

As already shown it is challenging to separate the effects of weather and emissions changes on air quality from comparisons of observational time series alone. Whilst air quality models, constrained with observations, can be used to challenge emissions inventories, they often require substantial computer resource and domain expertise and cannot easily be run over extended time periods. Computational models that can forecast time series based on fitting of non-linear trends in data using an additive model are now being widely applied and may be very useful when applied to analysis of long term air pollution data, particularly when abrupt changes occur. Figure 6.1 shows a version of this type of model that has been applied to the NO_2 data from Manchester Piccadilly Gardens, whilst an equivalent model was run and is being continually evaluated for all AURN sites for the last 5 years as part of a fellowship held with, and associated project funded by, the Alan Turing Institute (e.g. Figure 6.2). Wind speed, wind direction, temperature and pressure data along with the NO_2 from January 2015 through to the end of February 2020 were used to build the model. In a second version of the model, the NO_2 data was normalized to the nearby traffic volume data from Transport for Greater Manchester [TfGM] and this, together with the volume data itself, was used to build the model. The black dots in the figure show the last week of data used to build the model. Once the model had been constructed using the meteorological and, in the second version traffic normalized NO_2 and traffic volume, data it was allowed to run through March 2020 to the present time without using further meteorological constraints.

The red line in Figure 6.1 shows the measured NO_2 data and this compares extremely well with the measured prediction based on meteorology only through the period from March 1st to March 27th. Differences in NO_2 are typically only a few μgm^{-3}. Since the model was constructed using data from a period when traffic flows were normal, the extrapolation of the model NO_2, which was built only using meteorological data into the period covering lockdown, would not be expected to match the observations. Rather it provides a counter view of the expected NO_2 assuming activities in Manchester were to have continued as normal. The difference between the model and the observations therefore provide a measure of the effect of the lockdown on the NO_2 concentration in central Manchester. The observed NO_2 is approximately 30% of the model prediction post lockdown. This is consistent with the reduction in the traffic volume in central Manchester obtained from traffic volume counters located inside the inner ring road (Figure 6.3).

In the second model in Figure 6.1 (green line), the NO_2 used to build the model was normalized by the traffic data and then traffic volume from the nearest traffic counter used as a scalar (shown in figure 6.3). In essence this version of the model assumes that all the NO_2 was local, that is, it was produced by nearby traffic. For large parts of the lockdown period this model provides a reasonably accurate representation of the observations and demonstrates that the majority of the observed NO_2 is produced locally. However, for the periods between 23rd and 27th March, 1st April and 4th April onwards, the second model does not capture the observations. These periods coincide with periods of wider outflow from the UK and from northern Europe.

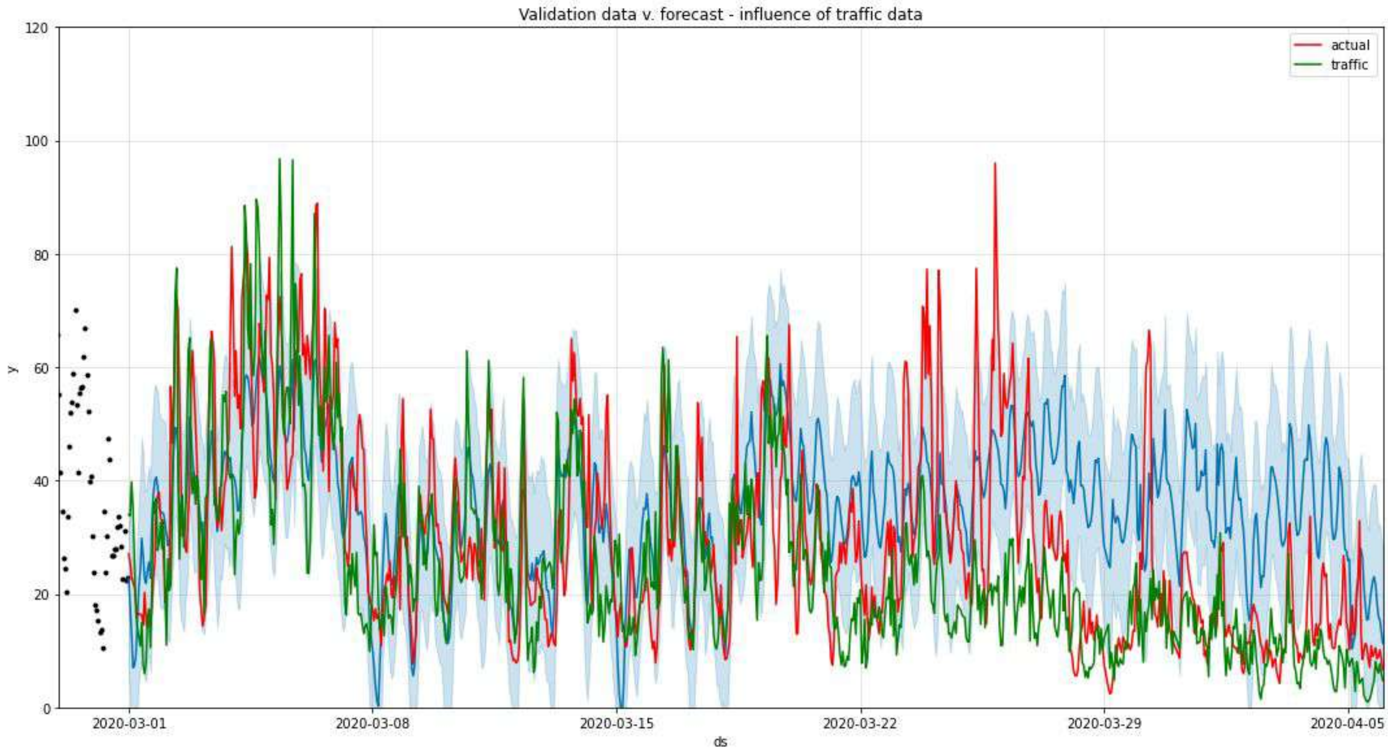


Figure 6.1: Time series models of NO_2 at Manchester Piccadilly Gardens compared to observational data (red line). The time series model was built using meteorological data (wind speed, wind direction, temperature and pressure) only (blue line) and also by normalizing the NO_2 data by the traffic flow data from nearby traffic flow counters and then scaling with the absolute traffic volumes (green line). The period used to build the model was from 1st January 2015 to the end of February 2020, the last week of the build data is shown as black dots. The model was run with no additional information from 1st March onwards.

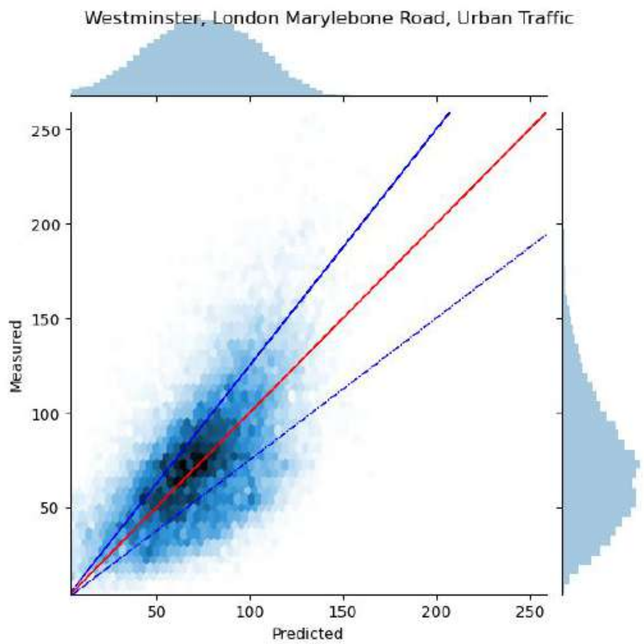


Figure 6.2: Comparing measured and predicted NO_2 levels at one hour resolution, over a rolling 1 month window from January 2018 to December 2019 using only data from that AURN site from the previous 3 years. The red line indicates the 1:1 ratio, and each blue line either side indicates a 25% boundary.

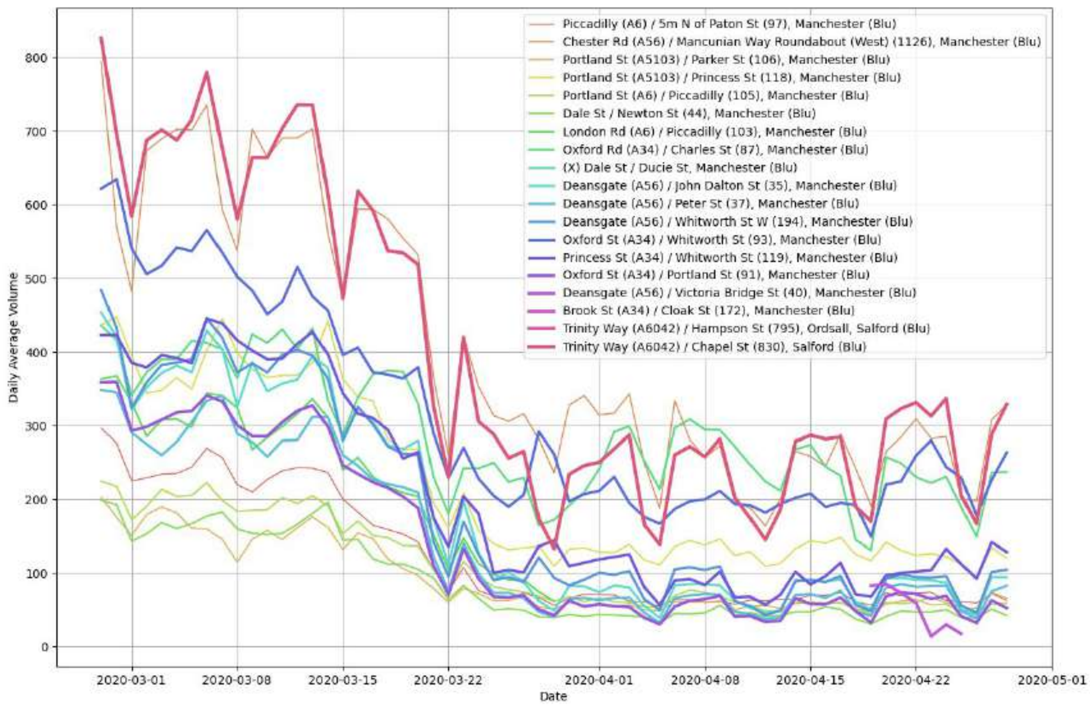


Figure 6.3: Traffic counter data, as average hourly volume, through March and April 2020 from the Manchester urban network inside the inner ring road. Reductions of traffic volume to around 25-35% of the pre-lockdown volumes occur across the network.

Appendix 7 Are there any insights that can be gained from aerosol science on possible viral transmission mechanisms?

There have recently been some high-profile contributions from the aerosol science community regarding the possibility of aerosol transmission of SARS-Cov-2, notably Asadi et al. (2020) and a submission in the USA made by the National Academy of Science, Engineering and Medicine (https://www.nap.edu/read/25769/chapter/1). News articles were also published in *Science* and *Nature* on this topic (https://www.nature.com/articles/d41586-020-00974-w and https://www.Sciencemag.org/news/2020/04/you-may-be-able-spread-coronavirus-just-breathing-new-report-finds). Previous evidence exists for the airborne transmission of SARS-Cov-1 (Yu et al., 2004), there is evidence that the SARS-Cov-2 virus can remain viable in the aerosol form (van Doremalen et al., 2020) and it is known that simple activities such as breathing can produce these aerosols, in addition to coughing and sneezing (Yan et al., 2018; Tang et al., 2013). The possibility of aerosol transmission is seemingly at odds with the official advice regarding social distancing, where 2 metres is supposed to be 'safe'. Furthermore, knowledge of $PM_{2.5}$ exposure would also suggest that inhalation of submicron particles could cause the virus to deposit directly into bronchial or alveolar regions of the lungs (Kappos, 2011), potentially causing a more severe infection.

Of particular note from the literature is a paper (published prior to the emergence of COVID-19) suggesting that the amount of aerosols produced by a person is increased if they are vocalising (in the paper, making an "aa" sound) and the amount produced can be linked with volume (Asadi et al., 2019). The implication is that in addition to coughing and sneezing, actions such as talking and singing could be considered risky. This paper also indicated that some individuals produce significantly more aerosols than others, creating the possibility that these could be 'superspreaders'. However, it must be stressed that this paper only measured the amount of aerosol produced and not whether viable coronavirus-containing particles were suspended in the particles. Another noteworthy paper measured a decrease but not elimination of aerosol coronaviruses (not SARS-Cov-2) being emitted from coughing patients through mask use, although the sample size was very small (Leung et al., 2020).

There is clearly the need for more evidence to support the notion of the significance of this as a transmission route in various settings, however experience of air quality research would suggest that enclosed areas would be of most concern, as aerosols will persist in these environments as opposed to outdoors, where they rapidly disperse through air movement. While there is rightly much attention being currently paid to aerosol transmission in care settings, interventions in other enclosed environments such as shops and public transport may need to be considered. These may include increasing ventilation, wearing of masks and limiting occupancy. Perhaps a good analogy is exposure to second hand smoke; this is known to be a much bigger problem in enclosed environments but it is still possible to catch the smell of a smoker outdoors if the wind is moving in the right direction. Because of the reported correlations between COVID-19 and air pollution, the hypothesis has been posed that outdoor pollution is somehow aiding transmission (e.g. https://dx.doi.org/10.1016%2Fj.jinf.2020.03.045). However, this is awaiting a definitive mechanistic process; while it has been suggested that ambient particles can accommodate viruses through atmospheric processing, it is not clear from a mechanical perspective how this would improve the virus' persistence. It is also possible that the relationship is indirect, due to established morbidities associated with air pollution (such as lung inflammation) making people more susceptible to infection or severe symptoms, or that the relationship is non-causative, such as pollution acting as a proxy for population density.

There is also the question of what scientific capability the UK atmospheric aerosol science community can contribute to assessing both the importance of the transmission route and effectiveness of any measures adopted. There are instruments within the community, in particular in the Atmospheric Measurement and Observation Facility (AMOF) pool that can be used to quantitatively measure aerosol in real time. The capabilities of the UK atmospheric aerosol science community can be used to characterise aerosolisation from breathing, coughing, sneezing and AGPs in the clinical setting and characterise aerosol extraction as a form of mitigation of viral transmission in such settings. Such work is being carried out in collaboration between the aerosol science groups in Manchester and Cambridge and Applied Fluid Dynamics expertise in Leeds, in consultation and collaboration with surgical colleagues from a number of NHS Trusts across the UK.

However, while these instruments could be of much use in many studies, these only measure the presence of particles, so will not unambiguously detect aerosolised viruses. While real-time bioaerosol sensors employing fluorescence (such as the WIBS) are in use by the atmospheric community, these have only previously been used for fungal spores, bacteria and pollen (Huffman et al., 2019). It is unlikely that these instruments will be able to detect airborne SARS-Cov-2 viruses, as these are much smaller and unlikely to contain fluorophores in the quantities needed to differentiate these from other particles.

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How have NO2, NOx and O3 changed since the COVID outbreak?

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29 April 2020

Summary
The aim of this work is to determine if emissions and ambient concentrations of NO2 and NOx and O3 have changed since the COVID-19 outbreak. Multiple linear regression modelling is used to predict what we might expect concentrations to be during the recent meteorological conditions with no reduction in emissions. These predictions are compared to measured concentrations at 110 AURN sites for the recent period. It is found that as a result of reduced emissions, on average, NO2 concentrations have reduced by 24%, NOx concentrations have reduced by 36% and O3 concentrations have increased by 20%. The largest changes in NOx are seen at Urban Traffic sites such as Oxford Centre Roadside (-63%) and London Marylebone Road (-48%). The smallest changes in NOx are seen at rural background sites such as High Muffles and Chilbolton Observatory (+8% and +2% respectively). All sites show an increase in O3 concentrations due to reduced emissions, ranging from the largest increase at London Marylebone Road site (+49%) to the smallest at High Muffles (+6%).

Data
Air quality data from the AURN network is used to produce a timeseries of hourly ambient concentrations of NOx, NO2 and O3. Meteorological data from the Met Office operational weather forecast (UKV) is extrapolated to the AURN site locations to provide hourly atmospheric pressure, temperature, wind speed, wind direction, precipitation and boundary layer depth data. Multiple linear regression is used to create statistical models for each AURN site and each pollutant using over 3 years of air quality and meteorological data (01/01/17 – 22/04/20).

Case study: Reading New Town

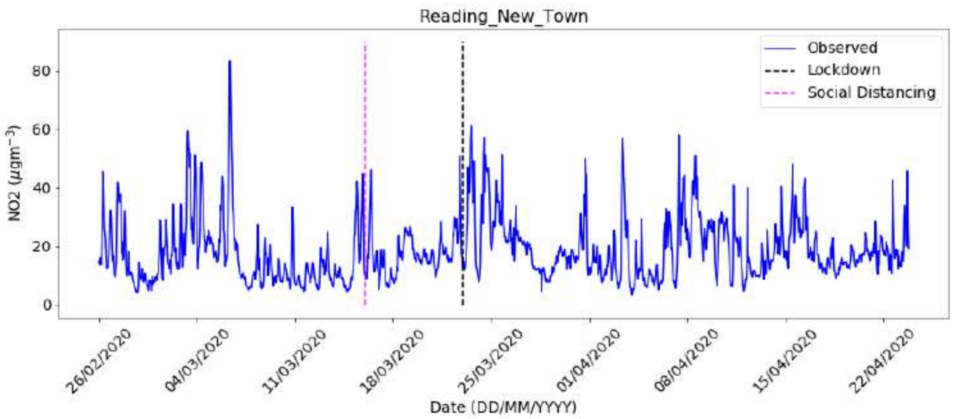

Figure 1. Hourly measured NO2 concentrations from 26 February - 22 April 2020. Date of social distancing implementation on 16 March (magenta dashed) and non-essential travel restrictions on 23 March, 'lockdown' (black dashed).

Figure 1 shows hourly AURN measurements of NO2 concentrations from 26 February – 22 April 2020 at the Reading New Town AURN site. The magenta dashed line shows the date on which social distancing was recommended (16 March) and the black dashed line shows the date on which non-essential travel was enforced, 'lockdown' (23 March). There's lots of variability in the data with no obvious decrease in NO2 following the COVID-19 travel restrictions. This is due to the fact that NO2 concentrations are influenced both by emissions and meteorology. NO2 concentrations tend to be higher when meteorological conditions are such that atmospheric dispersion is less efficient. These conditions are often associated with sunny high-pressure periods, such as those that occurred at the end of March and beginning of April.

To remove the variations that are due to changes in the weather a multiple linear regression model is used to predict the NO2 concentrations. Inputs to the model include hourly meteorological variables (wind speed, wind direction, temperature, pressure, precipitation and boundary layer depth) plus temporal data (time of day, day of the week, Julian day and the date). The model predicts what we might expect NO2 concentrations to be during the recent meteorological conditions with no reduction in NO2 emissions.

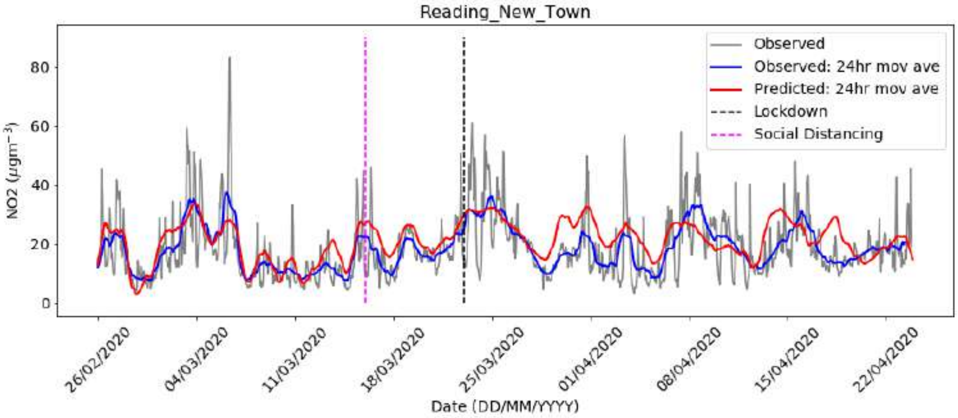

Figure 2: Hourly measured NO2 concentrations (grey), 24-hour moving averaged measured NO2 concentrations (blue) and predicted NO2 concentrations (red) from 26 February - 22 April 2020.

Figure 2 shows the observations, with a 24-hour moving average applied (blue) and model predictions for the same period (red). The model captures the NO2 variability in the 3-week period prior to the COVID-19 restrictions relatively well. After the COVID-19 restrictions the measured NO2 is generally lower than the model predictions suggesting that reduced NO2 emissions are leading to a reduction in NO2 concentrations at the Reading New Town site.

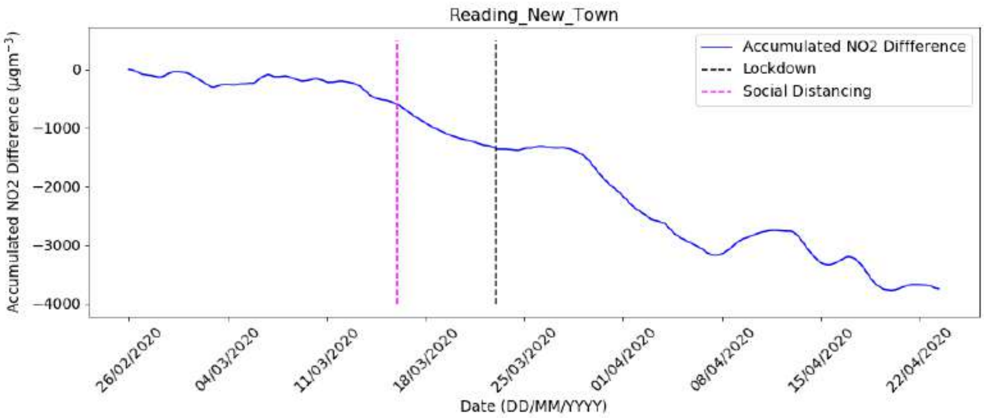

Figure 3: Accumulated difference between measured and predicted concentrations of NO2 from 26 February 2020.

The accumulated difference between the model prediction and the observed NO2 concentrations from the 26 February highlights the changes due to reduced emissions (figure 3). The accumulated difference is initially small since the over or underestimations predicted by the model cancel each other out. Following the social distancing recommendations, there's a significant increase in the difference indicating that measured NO2 is systematically below that predicted by the model.

This analysis suggests that at the Reading New Town AURN site a reduction in NO2 emissions has decreased NO2 concentrations by approximately 15% during the COVID-19 travel restriction period. Performing a similar analysis for NOx and O3 shows that due to a reduction in emissions, NOx concentrations have decreased by an average of 23% during the COVID-19 outbreak (figure 4) and as a result O3 concentrations have increased by approximately 16% (figure 5).

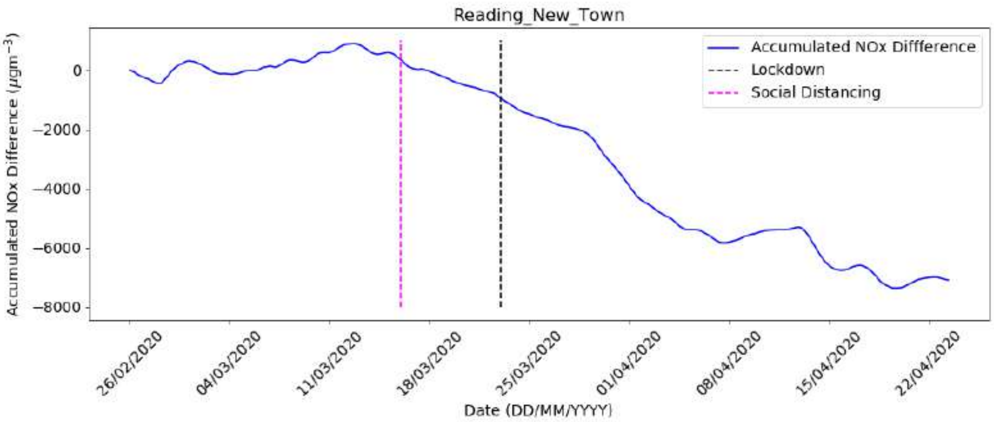

Figure 4: Accumulated difference between measured and predicted concentrations of NOx from 26 February 2020.

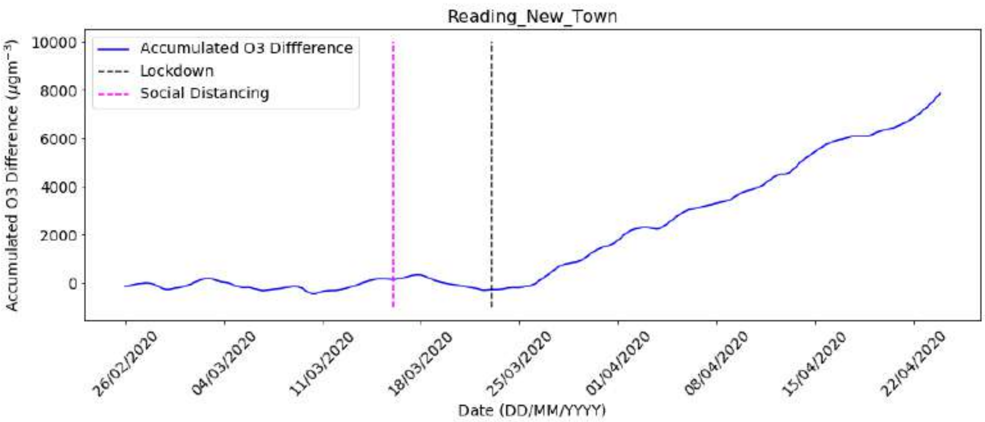

Figure 5: Accumulated difference between measured and predicted concentrations of O3 (right) from 26 February 2020.

Multi-site Analysis

Repeating the analysis for a further 110 AURN sites in England we find that not all sites exhibit the same changes in NO2, NOx and O3 concentrations due to reduced emissions as were observed at Reading New Town.

AURN site	Site classification	NO2	NOx	O3
Reading New Town	Urban Background	-15%	-23%	+16%
London Marylebone Road	Urban Traffic	-35%	-48%	+49%
High Muffles	Rural Background	+9%	+8%	+4%
Oxford Centre Roadside	Urban Traffic	-46%	-63%	No data
Lullington Heath	Rural Background	-2%	+12%	+9%
Chilbolton Observatory	Rural Background	+11%	+2%	+14%
Liverpool Speke	Urban Industrial	+22%	+30%	+20%

Table 1: Changes in NO2, NOx and O3 concentrations at specific AURN sites

Table 1 shows statistics for some example sites. The average change in NO2 concentrations due to reduced emissions since 16 March across all 110 sites is estimated to be -23% (stddev 13%), and the average change in NOx concentrations is estimated to be -36% (stddev 14%). The maximum decrease in concentration due to reduced emissions for both NO2 and NOx (-46% and -63% respectively) is found at Oxford Centre Roadside, which is an urban traffic site. Another urban traffic site, London Marylebone Road, also has large reductions in NO2 and NOx concentrations (-35% and -48% respectively). At some rural background sites there is an increase in NO2 above that predicted by the statistical models. For example, at the High Muffles site (+9%) and Chilbolton Observatory site (+11%). There is also an increase in NOx concentrations at some rural background sites, High Muffles (+8%) and Lullington Heath (+12%). Although it should be noted that concentrations of NO2 and NOx at these rural sites are low. All of the 51 AURN sites analysed for O3 show an increase in O3 concentrations above that predicted since 16 March, with an average increase of +20% (stddev 9%). The maximum O3 concentration increase is seen at London Marylebone Road (+49%) and the minimum O3 concentration increase is seen at High Muffles (6%).

The composite accumulated NOx difference between the statistical models and the measurements for all 110 sites is shown in figure 6. On average, the accumulated NOx difference decreases after 25 March, consistent with that found at the Reading New Town site. The composite accumulated O3 difference between the statistical models and the measurements for 52 AURN sites is shown in figure 7. On average, the accumulated O3 difference increases rapidly after 25 March (blue line, figure 7). The spread over the different sites is relatively small compared to the increase since most sites show a similar increase in slope.

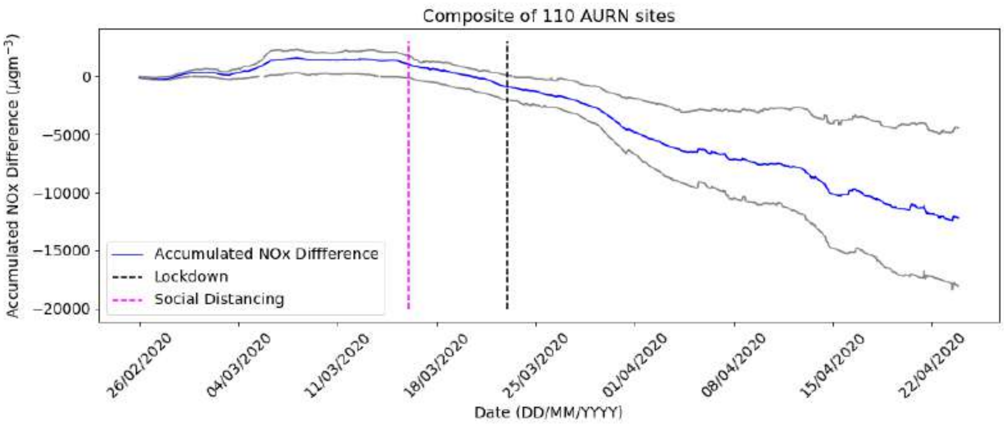

Figure 6: Composite accumulated difference between measured and predicted concentrations of NOx over 110 AURN sites. Mean NOx difference (blue) and 25th and 75th percentiles (grey).

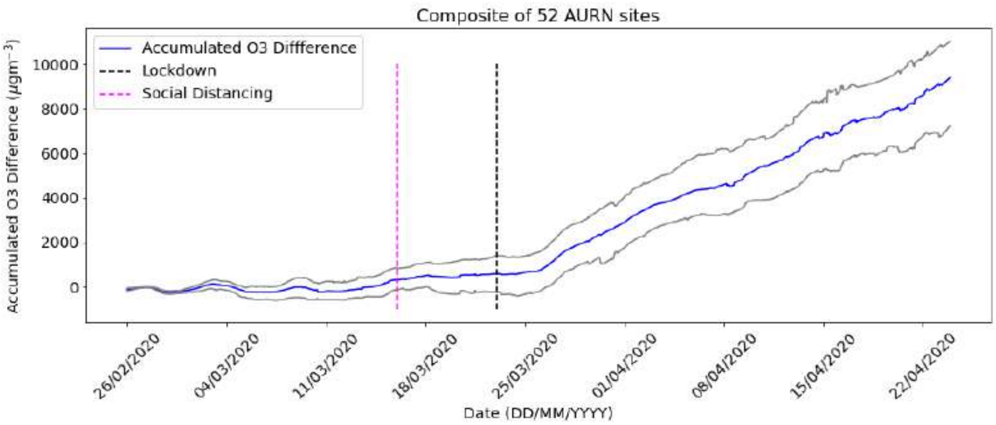

Figure 7: Composite accumulated difference between measured and predicted concentrations of O3 over 52 AURN sites. Mean O3 difference (blue) and 25th and 75th percentiles (grey).

AQEG Call for Evidence - Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK

Response from Ricardo to Key Questions

29th April 2020

The following is Ricardo's response to AQEG's Call for Evidence on the effects of the COVID-19 outbreak on air quality in the UK. Responses are provided to all questions, with the exception of the final two on health effects and viral transmission mechanisms where we defer to the expertise and insight of others.

What sectors or areas of socioeconomic activity do you anticipate will show a decrease in air pollution emissions, and by how much? Are there any emissions sources or sectors which might be anticipated to lead to an increase in emissions in the next three months?

Emissions will be affected in a variety of ways in response to changes in socioeconomic activities, with decreases expected in a number of key source sectors, and increases in others. There is likely to be changes in both the spatial distribution and temporal (hourly, daily) pattern of emissions. Emissions will have changed most sharply when the lockdown was introduced in mid- to late-March, but may not return to previous levels immediately after restrictions are lifted and the knock-on effect could be felt for several years as the economy gradually recovers. Eventually, emissions may stabilise at a 'new normal' reflecting changes in socioeconomic activity.

We have considered the effects on a number of source sectors captured in the NAEI (as well as some not captured) in terms of both direction and magnitude of change and the reasons behind them.

We start with considering those sources such as road traffic where changes in activities are most evident or most likely to have occurred and/or where sources make a significant contribution to UK emissions. We then consider some lesser sources where changes in activities and emissions have likely occurred, but the changes are more uncertain and/or the overall impact of those changes is smaller because it is a minor source. However, it should be noted that small changes in a number of related sources, whilst each on their own maybe relatively insignificant, may accumulate to more significant overall impacts e.g. VOC emissions from changes in the use of solvents.

Road traffic

The lockdown has had an obvious and noticeable effect on road traffic which has almost certainly led to **a reduction in exhaust emissions of all pollutants and non-exhaust sources of PM**. Reductions in traffic have been reported to be around

70% according to DfT. However, there are a number of nuances and competing effects that need to considered in estimating the overall impact on traffic emissions:

- Traffic flows are likely to have changed to a different extent for different vehicle and road types:
  - Large reductions in passenger car, taxis and bus traffic
  - Reductions in small van traffic, but possibly only small reductions in larger delivery van traffic where activity may have been less impacted because of increases in online shopping and the need to keep essential supplies moving
  - Little change in HGV traffic, especially on motorways and outside town centres
- Increases in average speeds in urban areas due to there being less congestion – this may lead to a reduction in emission factors
- Initially, over the first few months of lockdown, little change in fleet composition (Euro standard mix) because all vehicle ages are affected in a similar manner, except possibly few older HGVs which may tend to be used by smaller businesses currently in lockdown
- The exception to this will be in London where the congestion charge, LEZ and ULEZ charges have been suspended to help key workers travel. This will influence the Euro fleet mix with potentially greater proportions of older vehicles in the fleet than would normally be the case in the areas where the charges apply. A change in the activities of black cabs will also be expected.
- Beyond the first few months of the lockdown and into the coming years, the fleet composition could be significantly affected by the slowdown in new vehicle sales and hence turnover in the fleet. This slowdown in fleet turnover could offset the reduction in emissions that would otherwise occur due to lower traffic and could eventually, in future years, lead to an increase in emissions compared with current predictions if traffic returns to normal levels.
- There will be a change in the temporal as well as spatial distribution of traffic emissions, with less peak hour traffic on weekdays and weekday traffic more resembling weekend traffic. This will have an impact on pollutant concentration levels.

In summary, over the next few months we are likely to see an overall reduction in emissions, particularly in urban areas and a change in the hourly distribution of these emissions. However, in the longer term, it is uncertain what the trend might be and a possible increase in emissions due to lower fleet turnover cannot be ruled out. The NAEI has recently estimated that if there are significantly reduced new vehicle sales in 2020 and existing vehicles remain on the road in 2021, but traffic levels in 2021 are 5% lower than currently predicted levels, urban NO_x emissions will be 4% higher than currently predicted for 2021.

Aviation and airports

Emissions from aviation and activities at UK airports will have decreased significantly since the number of flights have been dramatically reduced. Flights from London Heathrow have reduced by around 70% since the 20th March according to the flight tracking website, flightradar24. These are scheduled departures and may include freight. The International Air Transport Association expects 2020 air traffic to fall by 38 per cent in total before embarking on a recovery later this year. As well as reductions in aircraft emissions, emissions from local traffic and airport support equipment and other activities in and around airports will have decreased.

Rail

Emissions from the rail sector are expected to have significantly decreased as many passenger rail services have been reduced. Data from DfT suggest a reduction of over 80% in national rail services since the imposition of social distancing rules. It is less clear what has been the impact on rail freight. According to Network Rail, freight services have continued to keep essential supplies moving such as food, medical supplies and fuel for power stations. In 2018, around 75% of all diesel fuel consumed by rail was consumed by passenger trains.

Domestic combustion

This sector may be expected to have shown an overall increase in emissions from gas used for heating and cooking as people are spending more time at home during the lockdown, although one report suggests that natural gas use is showing no clear surge in demand. Consumption may have been tempered by the fact that daytime temperatures have been above seasonal averages during the lockdown thus far. In spite of warmer weather, there **may have been an increase in PM emissions from wood burning** as people look for some comfort in trying circumstances and spend more evenings indoors.

As well as changes in total fuel consumed by domestic combustion, there may also be a change in the temporal patterns in fuel use which need to be taken into account in assessing overall air quality impacts. Hourly and daily consumption patterns during weekdays may now resemble weekends.

Power generation

Early indications from the National Grid are that overall demand for electricity has been reduced by 10-20% since the lockdown began compared with the average consumption rate for March/April. This is due to less demand from industry and commercial sectors, partially offset by a higher demand from the residential sector. This is likely have led to an **overall reduction in emissions from power generation**.

Commercial combustion

There is likely to have been a reduction in emissions from stationary combustion from most commercial and public buildings as businesses, offices, schools, leisure facilities and restaurants, pubs and bars remain closed. This is likely to have been offset by increases in emissions from hospitals and buildings used by key workers such as warehouses.

Industrial combustion

There is likely to have been an overall reduction in emissions from combustion in industry. The decreases will likely be most significant from combustion plant in smaller industries which have completely shut down, with smaller reductions in larger industrial plant which remain open but under reduced operation.

Industrial process emissions

There is likely to have been an overall decrease in fugitive emissions from many industrial processes, particularly from fuel distribution, mineral handling, cement, steel, brick and glass industries as well as solvents (however, see section below). There may be less change in VOC emissions from the food and drink industries as these remain in production.

Construction

Emissions from construction occur from fugitive sources (PM from dust release) and use of mobile machinery. A cessation in many construction activities will have significantly reduced emissions from both these sources. Various indicators on construction activities suggest a reduction of around 25% since the start of the lockdown.

Cooking

The NAEI does not cover emissions from the process of cooking, but there is evidence from ambient measurements of markers for cooking aerosols that this is a source of particulate matter. There is likely to have been a significant reduction in PM emissions from commercial cooking sources, such as restaurants, pubs and takeaway food establishments that have closed during the lockdown, especially in the centres of towns and cities. These may be offset by increased emissions from domestic cooking activity which will most likely be more widely dispersed.

Other sources

The NAEI has considered a range of other, fairly specific sources for which emissions MAY have changed since the lockdown began, but for which there is currently no firm evidence to support this. Current estimates of emissions of each of these individual sources may be relatively small and therefore of no great significance to air quality, but when considered together, their impact could become more significant, particularly in the case of VOC emissions affected from a range of different sources.

These sources and the pollutants emitted are summarised in the following table. The symbol ↑ indicates an increase in emissions may be expected, ↓ a decrease may be expected. The table also shows the percent contribution of the source to total UK emissions in 2018.

Emission source	Pollutants	Directional change	% UK totals in 2018	Reasoning
Burning of garden waste	NOx $PM_{2.5}$ BaP	↑	0.02% 1.3% 0.12%	An increase in bonfires due to reductions in garden waste collections and with people spending more time at home. Possible increase in emissions from outdoor cooking and barbeques
Composting/anaerobic digestion	NH_3	↓↑	2.3%	Lower emissions from less council waste collections offset by possibly more agricultural/commercial waste from food waste and domestic composting
Recycling/disposal of household waste, including wastewater treatment	NH_3 Hg	↑	0.66% 11.4%	Higher quantities of household waste collected as people spend more time at home
Clinical waste incineration	NO_x $PM_{2.5}$	↑	0.02% 0.01%	Increase in hospital admissions and activities, increased usage of PPE
Crematoria	NO_x $PM_{2.5}$ Hg	↑	0.05% 0.02% 15.8%	Increased hours of crematoria activity to cope with increased number of deaths
House & garden machinery	NO_x $PM_{2.5}$ VOCs	↑	0.10% 0.02% 0.16%	Increase due to people spending more time at home to do gardening / DIY
Shipping (domestic)	NO_x $PM_{2.5}$ SO_2	↓↑	10.6% 2.1% 7.1%	It is unclear how emissions have been affected. Vessels carrying foods and animal feedstocks and fishing may be unchanged or increased, those carrying petroleum, aggregates decreased.

				Decrease in passenger craft and cruise ships
Inland waterways	NO_x $PM_{2.5}$	↓	0.01% 0.01%	Decrease in recreational craft and tourist/pleasure boats on rivers and canals
Beer brewing	VOCs	↓	0.93%	Decrease in brewing to meet demands of pubs and restaurants unlikely to be offset by increase in domestic consumption at home
Production and use of sanitisers	VOCs	↑	Not estimated	Increased production, including at some distilleries, and consumption by general population
Aerosol and non-aerosol household cleaning and cosmetic products	VOCs	↑	7.7%	Increased by people spending more time at home. Possible increases in use of cosmetic products during periods of uncertainty
Non-aerosol products - Domestic adhesives, paint thinners	VOCs	↑	2.3%	Increased by people spending more time at home doing DIY
Car care products (screenwash)	VOCs	↓	4.4%	Less use of cars
Commercial cleaning products	VOCs	↓	1.6%	Less demand because of lower commercial activities
Commercial, industrial paints, coatings, adhesives, sealants	VOCs	↓	5.7%	Less industrial and commercial activities particularly among smaller businesses
Dry cleaning	VOCs	↓	0.08%	Lower demand
Refineries - fugitives	VOCs	↓	2.6%	Refineries may be operating at lower levels with less demand for fuels
Petrol distribution	VOCs	↓	2.1%	Less fuel being distributed as fuel demand decreased

Can you provide estimates for how emissions and ambient concentrations of NOx, NO_2, PM, O_3, VOC, NH_3 etc may have changed since the COVID outbreak? Where possible please provide data sets to support your response.

Emission changes

Ricardo is not yet in a position to estimate the changes in emissions that have occurred since the outbreak of COVID due to the lack of available data on a national scale. However, through the NAEI and PCM programmes we are in discussions with Defra on how early estimates could be made so that air quality in 2020 can be accurately modelled by the PCM for AQD compliance assessments.

The NAEI relies on various national statistics and annual data collected from regulators and industry. Quarterly statistics on energy trends and transport activities are published by BEIS and DfT, respectively, but with around a 3 month time lag from the end of a quarter, so data for the period April-June, the quarter most impacted by the current lockdown, may not be available until the Autumn. Ricardo will be investigating getting earlier sight of data. As indicated above, DfT has already estimated a 70% reduction in traffic since the lockdown begun.

Ambient Concentration changes - Analysis of NO_x and NO_2 data valid up until ~ 18th April

Selected AURN NO_x and NO_2 concentration measurements have been analysed using statistical models that control for meteorological variation. To quantify the changes due to Covid-19 it helps to make a comparison with the counterfactual i.e. what would have occurred in a business as usual case in the absence of Covid-19. Our approach has been to develop statistical models to predict the counterfactual at a range of AURN sites across the UK. These methods are described in a series of papers and previously applied to other situations where the quantification of the impact an intervention has been required (Carslaw and Taylor, 2009, Carslaw et al., 2012; Grange and Carslaw, 2019).

The approach adopted is summarised in the bullet points below.

- Models are developed using the *deweather* (https://github.com/davidcarslaw/deweather) R package. The models were based on hourly concentration data and meteorological data from the nearest surface measurement site over the period January 2018 to February 2020. The analysis was conducted at a total of 29 AURN sites (a mixture of roadside and background).
- Model variables include basic meteorological measurements such as wind speed, wind direction, ambient temperature and variables to account for the temporal variation in emissions such as (local) hour of the day and day of the week.
- The models are evaluated against randomly withheld data i.e. data not used for model development. Typically, an R^2 value of about 0.8 is typical of model performance for NO_x and NO_2.

- The models are then used to predict hourly concentrations from 1 March onwards at each site. These predictions provide the business as usual counterfactual against which the measurements can be compared.

A simple and effective approach to calculating whether concentrations deviate from expectations is to consider **cusum** charts. In a cusum chart, the deviation between two quantities is accumulated over time. This approach has the effect of amplifying changes over time and providing an indication of when one quantity begins to diverge from another. In the current context the two quantities being considered are the BAU and actual measured concentrations. If measured concentrations remain largely similar to predicted BAU concentrations, the cusum chart will remain close to zero. Cusum plots can help reveal the timing and magnitude of a change.

The models have been used to quantify the effect on NO_x and NO_2 concentrations from 16th March to 18th April i.e. from the Government advice on social distancing on the 16th March and including the period from 23rd March when the lockdown began. **At roadside sites the mean decrease in NO_2 concentration estimated was 37%, whereas at urban background sites the decrease was 25%. The corresponding NO_x reductions were 48 and 31%, respectively.**

An example of the application of the model to York Fishergate roadside site is shown in Figure 1.

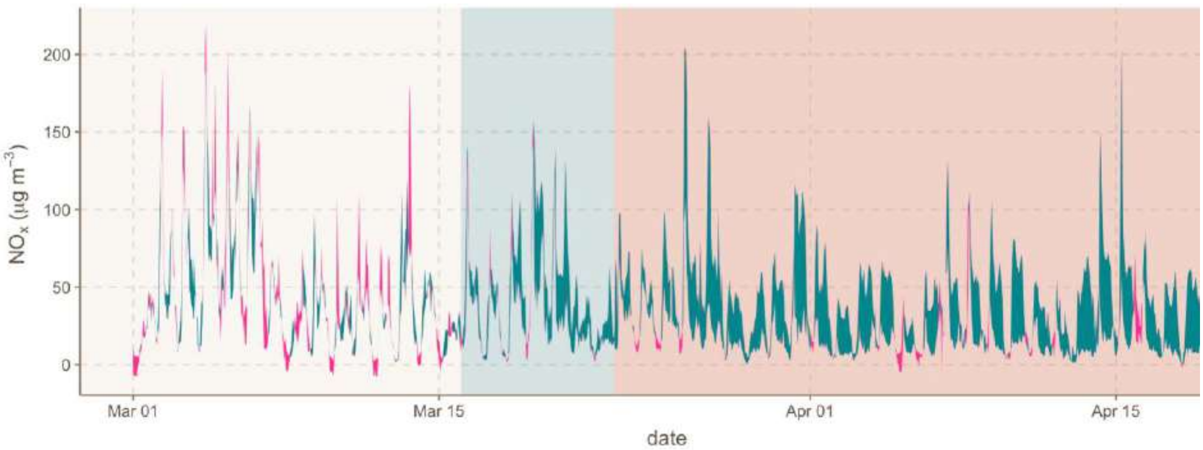

Figure 1 Concentrations of NO_x at the York Fishergate roadside site. Trend areas shaded green indicate that concentrations of NO_x were estimated to be lower than that expected through business as usual. The pink trend shading shows where concentrations of NOx were estimated to be higher than business as usual. The blue shaded rectangle shows the period recommended social distancing from 16th March, the pink shaded rectangle shows the period from the start of the lockdown starting 23rd March.

Analysis of concentrations at other sites are shown in the following figures.

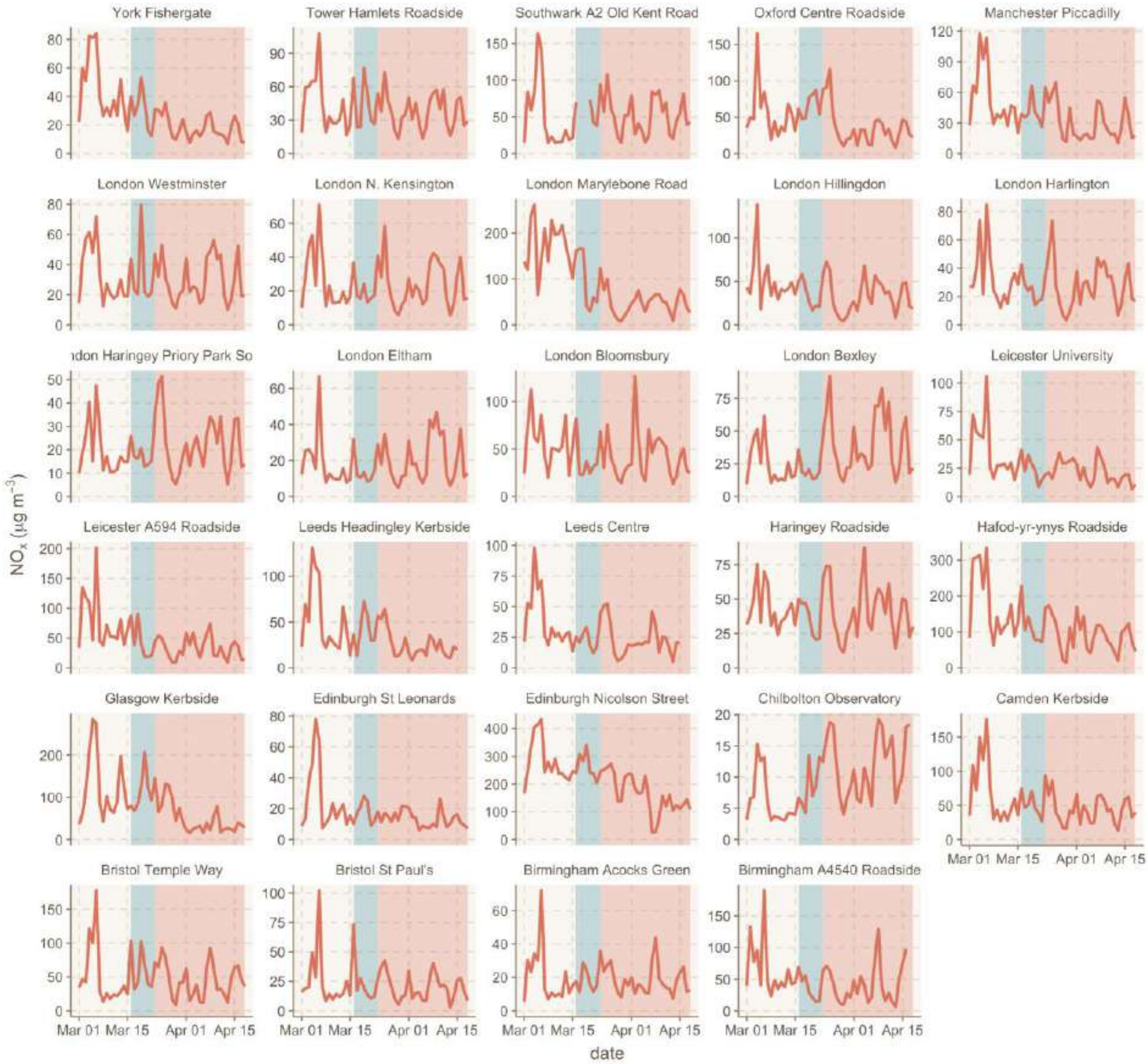

Figure 2 Daily mean concentrations of NO_x at a range of air pollution monitoring sites across the UK throughout March 2020. The blue shaded rectangle shows the period from 16th March, when social distancing was first recommended. The pink shaded rectangle shows the period from the start of the 'lockdown' that began on the 23rd March.

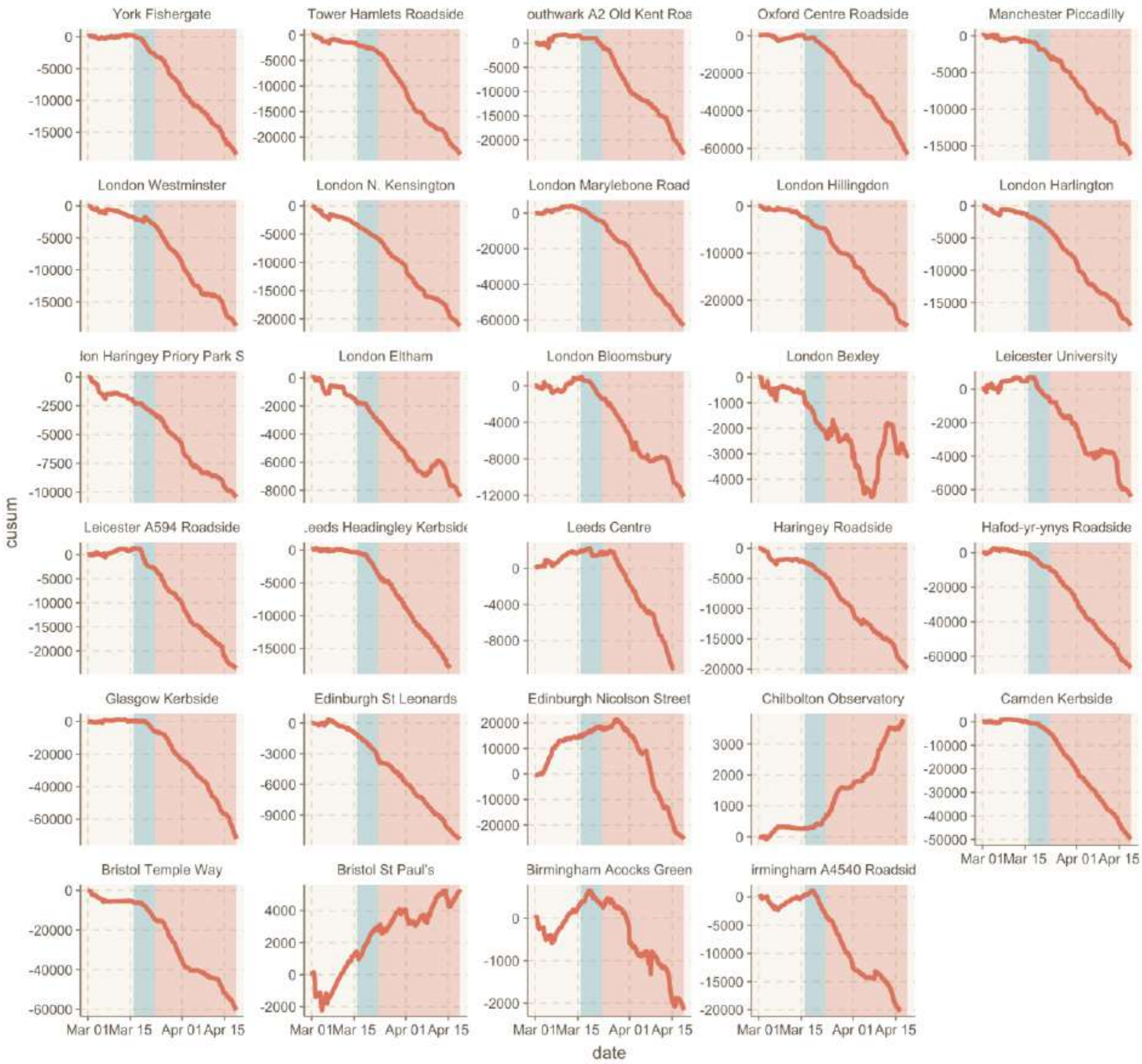

Figure 3 The cumulative sum (or cusum) of measured minus business as usual NO_x at a range of air pollution monitoring sites across the UK. The blue shaded rectangle shows the period recommended social distancing from 16th March, the pink shaded rectangle shows the period from the start of the lockdown starting 23rd March.

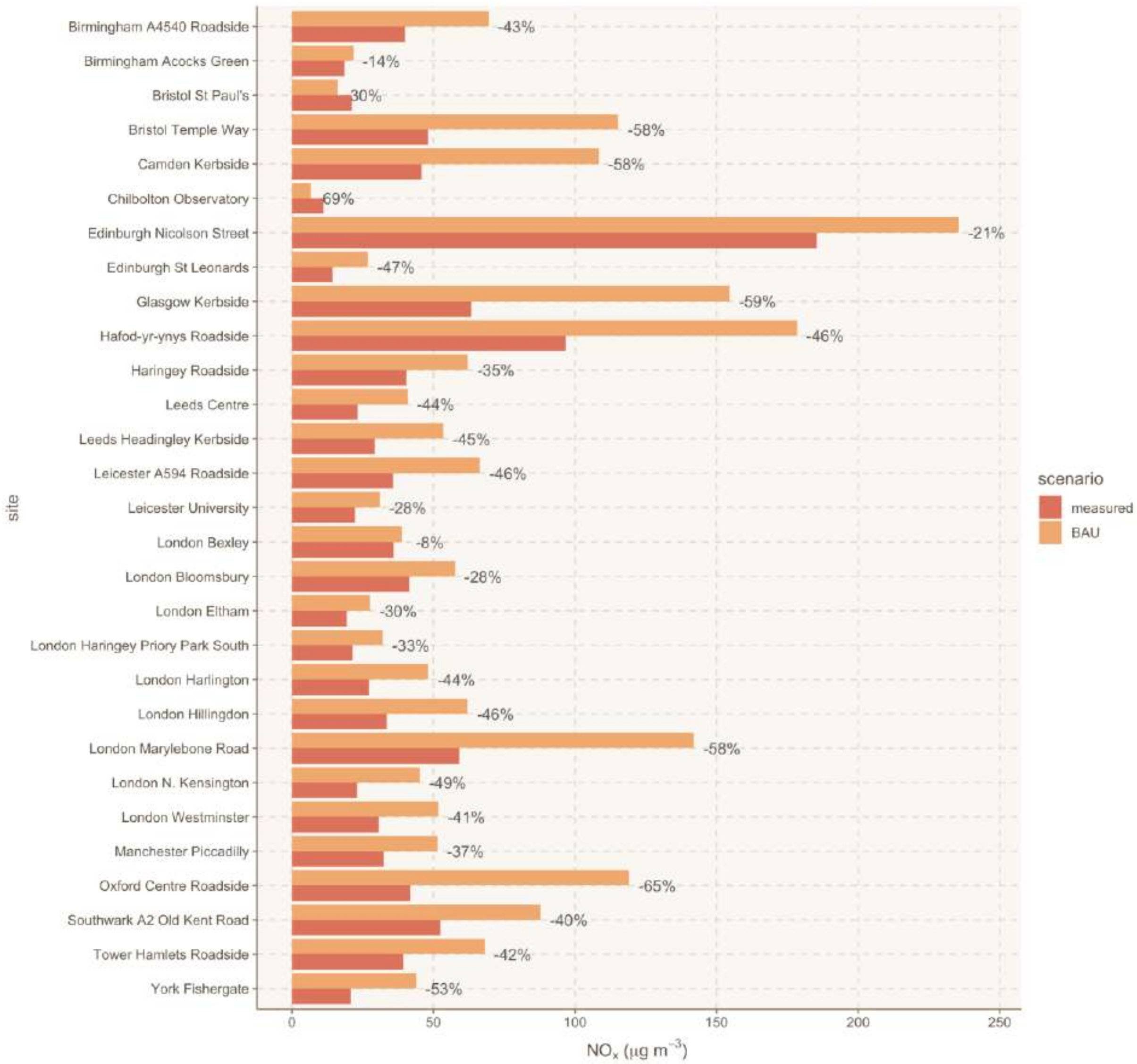

Figure 4 Measured and estimated business as usual NO$_x$ concentrations by site. The numbers show the percentage change in concentration relative to business as usual.

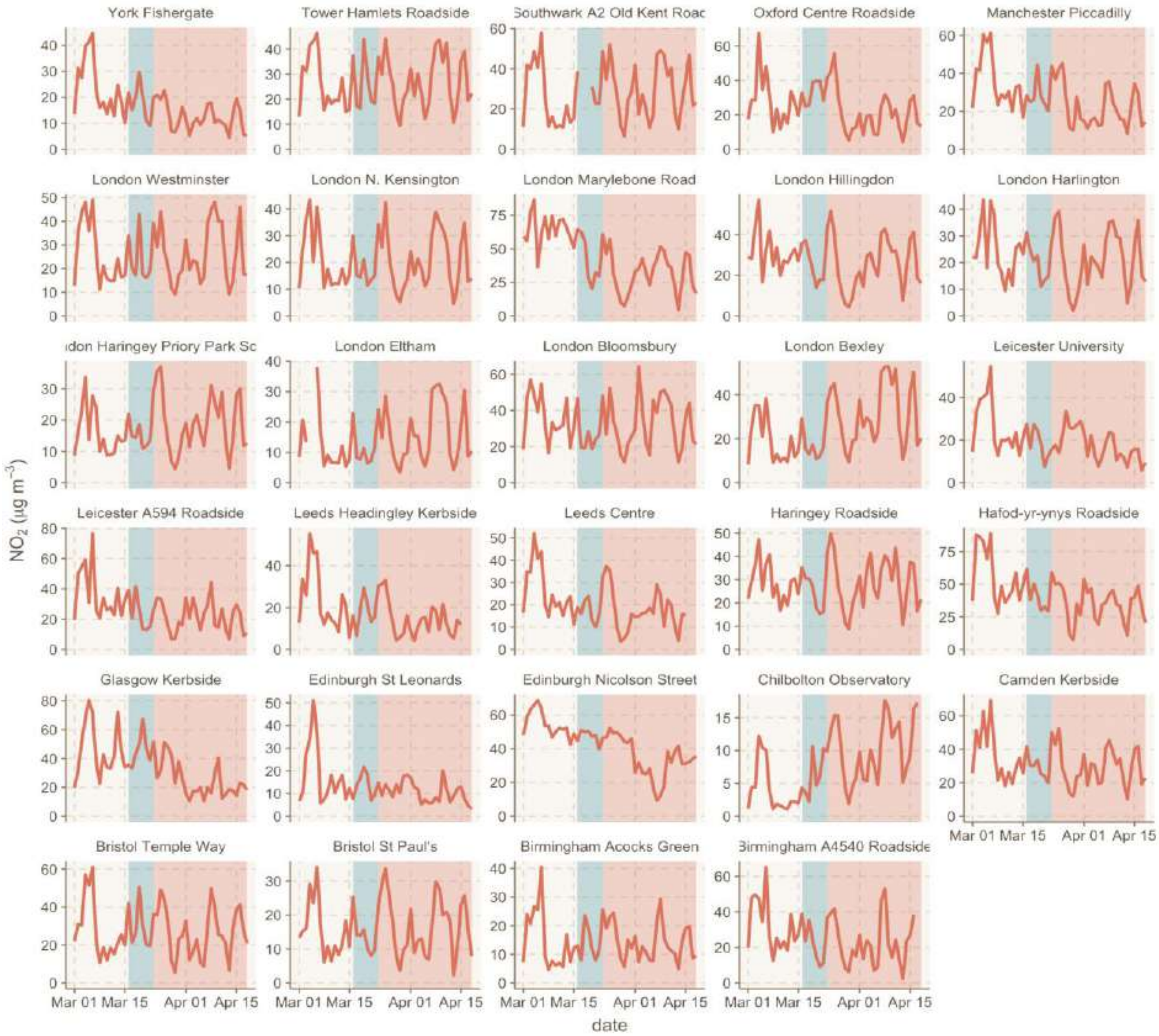

Figure 5 Daily mean concentrations of NO_2 at a range of air pollution monitoring sites across the UK throughout March 2020. The blue shaded rectangle shows the period from 16th March, when social distancing was first recommended. The pink shaded rectangle shows the period from the start of the 'lockdown' that began on the 23rd March.

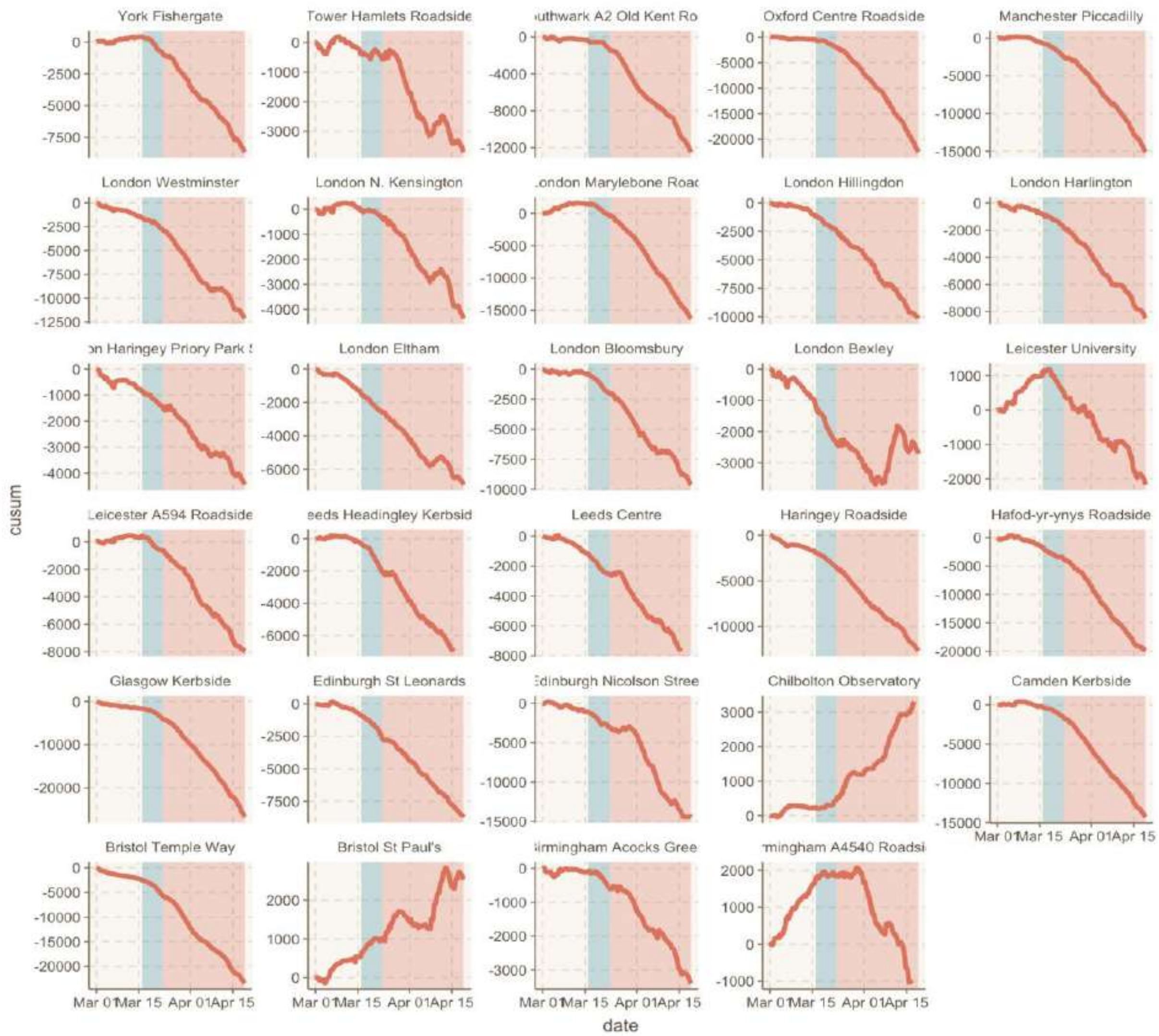

Figure 6 The cumulative sum (or cusum) of measured minus business as usual NO_2 at a range of air pollution monitoring sites across the UK. The blue shaded rectangle shows the period recommended social distancing from 16th March, the pink shaded rectangle shows the period from the start of the lockdown starting 23rd March.

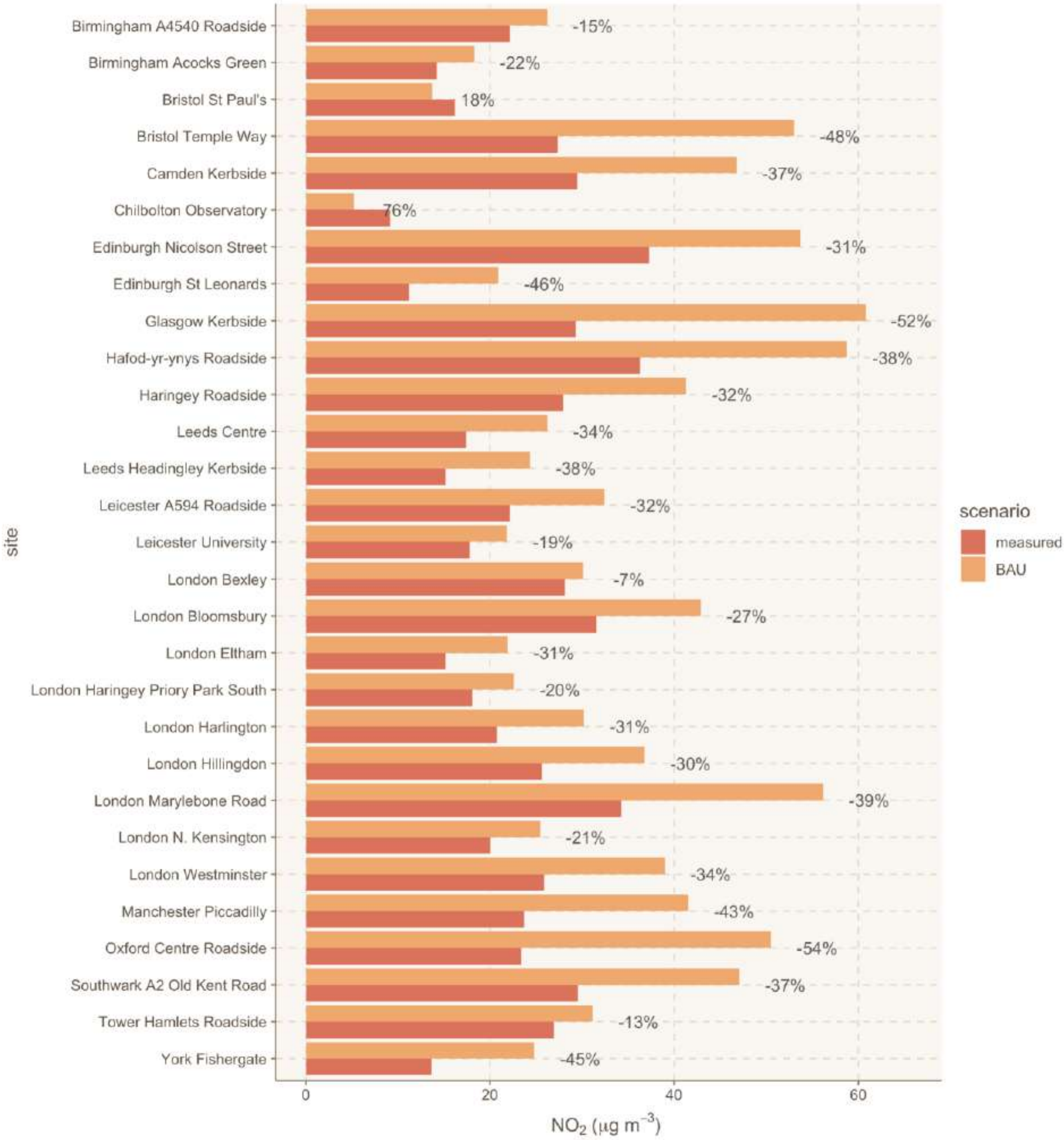

Figure 7 Measured and estimated business as usual NO_2 concentrations by site. The numbers show the percentage change in concentration relative to business as usual.

The models described above can also be used to estimate the impacts on exceedances of the NO_2 limit value of 40 µm^{-3} for 2020. There are several ways of approaching this issue. The approach adopted here, and demonstrated for the site at Marylebone Road, is to predict the rest of 2020 based on the emission levels experienced over the most recent weeks (last two weeks of the period analysed).

The actual 2020 annual mean NO_2 concentration will depend on the length of the lockdown, and activity levels and meteorology experienced for the rest of the year. Hourly predictions of NO_2 have been made for the rest of 2020 based on different meteorological years (2010 to 2019) to develop an understanding of the effect of meteorology on exceedance statistics. This analysis suggests a range of possible annual mean concentrations from 38.6 to 43.9 µm^{-3} at Marylebone Road depending on the meteorology.

The analysis can be refined in several ways e.g. considering different lockdown durations. The model developed also uses background ozone from North Kensington. It would be possible therefore to explore the effect of increased background ozone concentrations on NO_2 by adding (for example) a fixed increment of ozone to the measured values from other years (2010 to 2019).

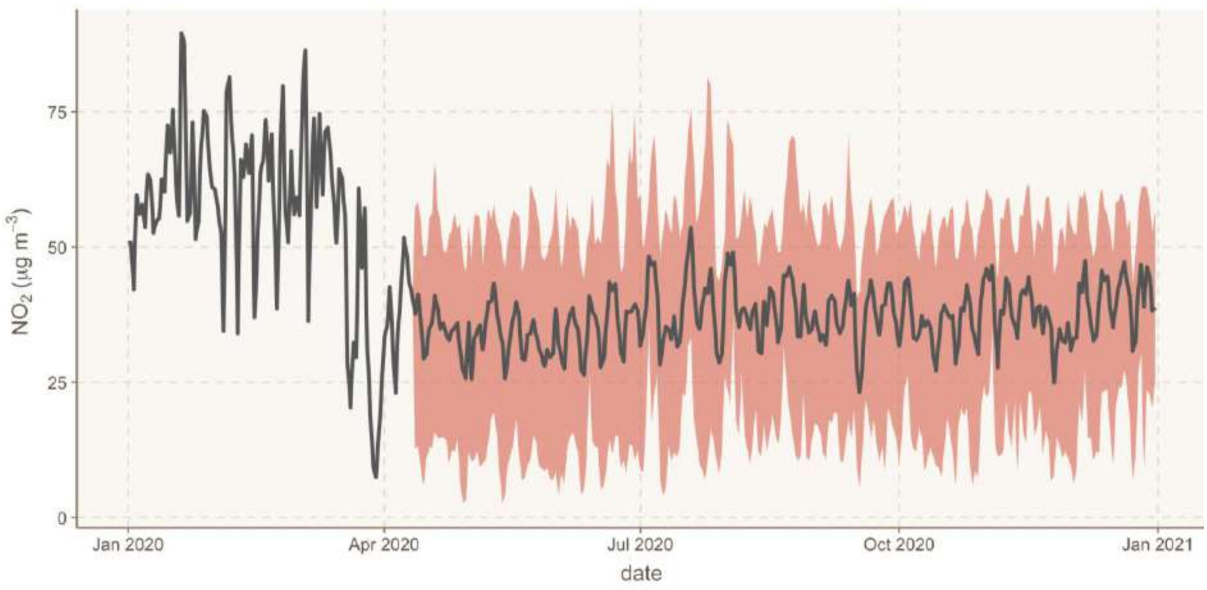

Figure 8 Trend in NO2 concentration at Marylebone Road. The initial part of the time series to early April is based on measured NO2. The remaining time series is based on simulations using different meteorology from 2010 to 2019.

Likely impact on the contribution of sources to background NO_x concentrations

The Pollution Climate Mapping (PCM) model has recently assessed the likely impact of COVID-19 measures on the contribution of sources to background NO_x concentrations in 2020. These are indicated qualitatively in the following table:

Sector	Average % contribution to background NOx	Likely impact of Covid-19 measures in 2020 (qualitative)
Urban background: Road transport	42%	Strongly down
Regional background: UK sources	14%	Slightly down
Regional background: EU sources	8%	Slightly down
Urban background: commercial combustion	6%	Down

Urban background: domestic combustion	5%	Up
Urban background: combustion industry	5%	Down
Urban background: other transport and mobile machinery: off road industry	5%	Down
Regional background: Shipping sources	4%	Slightly down
Urban background: Point sources (industry)	3%	Slightly down
Urban background: other transport and mobile machinery: rail	3%	Slightly down
Urban background: other transport and mobile machinery: ships	2%	Slightly down
Urban background: other transport and mobile machinery: off road other	1%	Likely no change, apart from air support down
Urban background: other transport and mobile machinery: other	1%	Likely no change
Urban background: energy production	0%	Likely no change
Urban background: other transport and mobile machinery: aircraft	0%	Down
Urban background: Agriculture	0%	Likely no change
Urban background: Waste	0%	Likely no change
Urban background: processes industry	0%	Down
Urban background: Solvents	0%	Down
Urban background: Natural and other sources	0%	Likely no change
Urban background: extraction of fossil fuels	0%	Likely no change

Impacts from Changes in Agriculture

It is less clear how emissions from agriculture have been affected by the lockdown. However, the wettest February on record across England followed by a uniformly dry period will have caused an initial delay in manure and slurry spreading and subsequently larger quantities being applied within a shorter period of time as ground conditions became more suitable for spreading. This will have had an impact on the timing of emissions of ammonia, with the later applications coinciding with the lockdown and therefore higher NH_3 emissions occurring when other pollutant emissions, particularly NO_x, are reduced thus impacting on secondary PM formation. The temporal variation in NH_3 emissions from these sources and the impacts this can have on sensitive habitats and secondary PM is poorly understood.

What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?

People are likely to be spending more time indoors and therefore a higher proportion of total exposure to indoor air pollution. The most significant change in indoor air quality is likely to come from increased emissions of VOCs from various domestic aerosol and non-aerosol household cleaning and personal care products, domestic adhesives, paint thinners and other products. There may also be an increase in home cooking and bread baking which are another source of VOC emissions indoors. These emissions may also lead to increases in levels of secondary organic aerosols and ozone indoors. The impact of these increases in emissions may have been tempered during the recent warm spell by the fact that people are keeping their houses well ventilated with opened doors and windows.

How might public exposure to air pollution have changed as a consequence of recent restrictions on movement?

Public exposure to air pollution will have shifted from less exposure to outdoor air pollution of NO_2 and PM to increased exposure to indoor air pollution, particularly to VOCs. This is partly due to people spending more time indoors and away from roadsides, combined with there being lower emissions from traffic, commercial and industrial sources and possibly higher emissions of VOCs from indoor activities.

Within an outdoor air pollution context, there may be a shift from there being less intensive emissions and concentrations of NO_2 and primary $PM_{2.5}$ in urban centres, but increases in emissions in more suburban regions due to domestic activities such as bonfires. People who do venture outdoors in urban areas may be exposed to higher levels of ozone as concentrations of this pollutant increase due to there being less removal of ozone by NO_x emissions from traffic sources.

How might altered emissions of air pollutants over the next three months affect UK summertime air quality?

The changes in emissions from UK sources outlined above are most likely to affect concentrations of NO_2, PM (possibly primary PM more than secondary PM) and ozone. In urban areas, NO_2 and PM may decrease, but O_3 increase. There are likely to be further influences on summertime air quality from changes in precursor emissions occurring across the northern hemisphere, particularly from sources in Europe and North America, which could lead to lower background concentrations of PM, ozone and NO_2. Changes in emissions of SO_2 and NH_3 maybe much smaller.

As a consequence, and in summary, we may see:

- Lower urban and background NO_2
- Little change in background PM, possibly a slight decrease

- Smaller urban NO_2 and PM increment
- Slightly lower background O_3
- Higher urban O_3

Prepared by Tim Murrells, David Carslaw, John Stedman, Hugh Martineau, Paul Willis

Ricardo Energy & Environment

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Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK
- Air Quality Expert Group -

Summary Response from BlockDox
www.blockdox.com

About BlockDox

BlockDox is an award winning urban digital solutions company headquartered in London, but working globally. Our patented technology uses the very latest innovations in the Internet of Things, as well as data science to help make spaces smarter.

We are currently focussed on two smart city verticals: smart buildings & public transportation.

We have previously been awarded innovation funding from InnovateUK and the European Commission, won several awards and participated in multiple smart city accelerator programs globally.

BlockDox and Indoor Air Quality

BlockDox have recently completed an extensive R&D project funded by InnovateUK combining indoor air quality data from a variety of retrofittable indoor air quality sensors, integrated with real time occupancy data and building management systems. This 12-month project finished on 31st March 2020, in the midst of the COVID-19 outbreak. Therefore, we have data from various pilot sites across the UK both immediately before the outbreak took hold and as it emerged in the UK.

The BlockDox technology can be deployed at other sites to support further research, policy making and/or practical interventions in response to COVID-19.

Response to Your Key Questions

1. **What sectors or areas of socioeconomic activity do you anticipate will show a decrease in air pollution emissions, and by how much?**

   Pollution emission is likely to have significantly decreased due to the following activities:
   - Reduced use of public and private transport due to the general public adherence to Social Distancing.
   - Significant reduction in workplace occupancy leading to reduced energy use, such as from Heating Ventilation and Air Conditioning (HVAC) systems, and consequently air pollution.

- The closing of the vast majority of shopping centers, restaurants, and cafés except for home delivery activities.

Quantifying the reduction in outdoor air pollution from empty workplaces is challenging because it depends on whether those buildings have actually intervened to reduce their regular HVAC system usage. HVAC is responsible for up to 50% of energy use in buildings and approximately 15% of all electricity consumption worldwide, with the consequent direct and indirect costs attached to producing and servicing this energy demand which also has an effect on air pollution.

2. **Are there any emissions sources or sectors which might be anticipated to lead to an increase in emissions in the next three months?**

Since public movement restriction is expected to continue over the next three months in various degrees of strictness, it is expected that domestic environments will have higher than normal energy consumption due to continuous occupancy. There may be an increase in pollutants from domestic gas appliances, an effect that is somewhat mitigated by the reduced need for heating as spring turns to summer. However the radical shift in working, travelling and shopping behaviour, combined with increased activity in some industrial sectors (such as grocery retail), but reduced activity in most, makes the overall local and national pollution trends hard to predict.

Wariness of overcrowding on public transport and the risk of infection may motivate people to travel by car instead, increasing emissions from road traffic. Transport system operators will need to provide customers with clear messaging about the availability of sufficient space in their vehicles and buildings. BlockDox PassengerCount™ is an integrated footfall and occupancy solution for rail and bus that provides decision makers with the tools they need to monitor people density in vehicles and at stations and terminals, and to provide the public with reassurance that social distancing can be maintained.

3. **Can you provide estimates for how emissions and ambient concentrations of NOx, NO2, PM, O3, VOC, NH3 etc may have changed since the COVID outbreak? Where possible please provide data sets to support your response.**

BlockDox are collecting occupancy and indoor air quality data from various buildings in Central London and South East England. Based on our data, we have observed a marked progressive reduction in both CO2 and VOCs in buildings since the COVID outbreak. Each parameter (VOC and CO2) responds slightly differently from each other to the reduction in occupancy since the COVID outbreak.

Interior CO2 concentration rises daily in buildings, and this rise is typically mostly caused by people breathing, but the daily rise in outdoor CO2, caused by pollution, can also be a contributory factor. Our data shows a gradual decrease in average concentration as well as a decrease in the size of daily peaks, estimated to be 20% of total CO2 concentration. By the end of the period, when very few people are in the buildings, average CO2 concentration is nearly the same as the global background concentration in the atmosphere. We also looked at the minimum CO2 in each 24 hour period, which occurs nightly when no-one is in the building and the air has all been replaced by outdoor air.The daily minimum CO2 concentration dropped by ~8%.This is likely to be largely caused by a similar drop in outdoor CO2 concentration, presumably as a result of the reduction of polluting activities in the lockdown period. The night time minimums are now approaching the global background concentration suggesting that night time CO2 pollution at Canary Wharf has been eliminated by the lockdown.

VOC pollution occurs in spikes, as VOCs tend to be caused by a localised source (e.g. off-gassing furniture) and diffuse rapidly. High concentrations of VOCs are very rare outdoors, and fluctuations in outdoor VOC levels are insignificant compared to the indoor spikes in concentration generated by localised sources. High concentrations of VOCs can be harmful over a short time period, so risk is defined by how high the spikes are rather than the averaged trend. We estimate the overall reduction to be 82%, which probably can be entirely explained by the sharp drop in occupancy. We note that in our data potentially harmful levels of VOCs are rarely seen.

The reduction in pollution indoors is likely to be largely driven by the reduction in occupancy, but outdoor air quality may also be a significant factor.

4. **How might public exposure to air pollution have changed as a consequence of recent restrictions on movement?**

Our data indicates that public exposure to outdoor air pollution will have reduced not just as a consequence of restrictions on movement,but also because the concentrations of pollutants are also significantly lower.

However, public exposure to indoor air pollution in a domestic environment may have increased. This will be exacerbated by those who live in areas where outdoor air pollution has previously shown to be worse. Even if outdoor air pollution has reduced due to movement restrictions, it has not been eliminated. As the public are spending more time at home in that location than they would otherwise, their exposure time is greater.

Further, as domestic environments rarely have sophisticated air ventilation or filtration systems compared to non-domestic environments, there are limited remedial measures that can be applied to reduce exposure. For example, opening a window may also create an entry point for harmful outdoor air pollution to make its way indoors. The installation of air quality sensors at residences, indoors and outdoors can provide occupants or building managers with the information they need about when to increase or decrease ventilation, to maintain the healthiest environment possible.

5. **How might altered emissions of air pollutants over the next three months affect UK summertime air quality?**

N/A

6. **Based on what is already known about air pollutants as respiratory irritants or inflammatory agents, can any insights be gained into the impact of air quality on viral infection?**

People usually spend an estimated 87% of their time indoors. Due to the recent restrictions, this is likely to have increased significantly. Indoor air can be more dangerous than outdoor air, because pollutants can become trapped inside buildings with inadequate ventilation. Indoor air quality is known to influence the propagation of airborne illnesses caused by viruses, bacteria and fungal spores.

Recent research has shown a significant correlation between the causes of death of COVID-19 patients and air pollution exposure (which can damage the heart and lungs making people more vulnerable). For example, an increase in particulate pollution (PM2.5) of just 1 microgram per cubic metre has been associated with a 15% increase in death rate (see Endnote for references). This suggests that interventions can be targeted towards areas where air pollution (and particularly PM2.5) is worse.

Other research from Wuhan University (see Endnote) found elevated levels of the COVID-19 virus in areas prone to crowding or poor ventilation. Their results indicated that room ventilation, open space, sanitisation, and proper use of disinfection effectively reduced the concentration of the virus in the air.

Therefore, the importance of real time monitoring of indoor air quality and occupancy, as BlockDox is capable of doing, is paramount in supporting appropriate interventions.

7. **Are there any insights that can be gained from aerosol science on possible viral transmission mechanisms?**

N/A

Endnote: Selected References

Conticini, Edoardo, et al. “**Can Atmospheric Pollution Be Considered a Co-Factor in Extremely High Level of SARS-CoV-2 Lethality in Northern Italy**?” *Environmental Pollution*, 2020, p. 114465., doi:10.1016/j.envpol.2020.114465.

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Report for The Air Quality Expert Group, on behalf of Defra: Analysis of air quality changes experienced in Sussex and Surrey since the COVID-19 outbreak

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1. Phlorum Ltd; 2. University of Brighton. With data contributions from Sussex Air Quality Partnership (Sussex-air) partners.

1. Overview

In response to Defra's call for focused and rapid scientific evidence to support decision-making on air quality management as a result of COVID-19, this analysis seeks to provide estimates for how emissions and ambient concentrations of NO_2, O_3, SO_2, HCHO and HONO have changed since the start of 2020. The report is a collaboration between the University of Brighton, Phlorum Ltd and Sussex-Air, and provides an overview of south east regional trends using automatic (continuous) monitoring measurements[1] from the AURN and Sussex network Air Quality Monitoring Stations (AQMSs), the University of Brighton's *Joaquin Advanced Air Quality reSearch* (JAAQS) laboratory and ESA's Sentinel-5P satellite.

2. Sussex Air Quality Monitoring Stations: De-weathered measurement data

OpenAir statistical modelling[2] software has been used to remove the influence of meteorology using the 'de-weather' package to identify whether changes in pollutant concentrations reflect changes in emissions as a result of current travel restrictions. The model runs have used multi-year historic air pollutant monitoring data and meteorological data from Shoreham Airport, Gatwick Airport, Herstmonceaux and Lydd.

3. Modelled changes in measured ozone (O_3) concentrations

De-weathered O_3 measurements from 3 background AQMSs in the Sussex Air Quality Network: Poles Lane, Crawley/Gatwick (RG3); Lullington Heath AURN (LL1); and Brighton Preston Park (BH0), have been analysed.

While O_3 is a complicated pollutant to analyse due to its secondary atmospheric formation, preliminary analysis shows minor increases in daily average background O_3, first in the pre-lockdown period from 15th March 2020 to 23rd March 2020, and then a step-up in de-weathered concentrations after the official lockdown start date (23rd March). Figure 3.1 presents the daily ozone concentrations at background sites in Sussex.

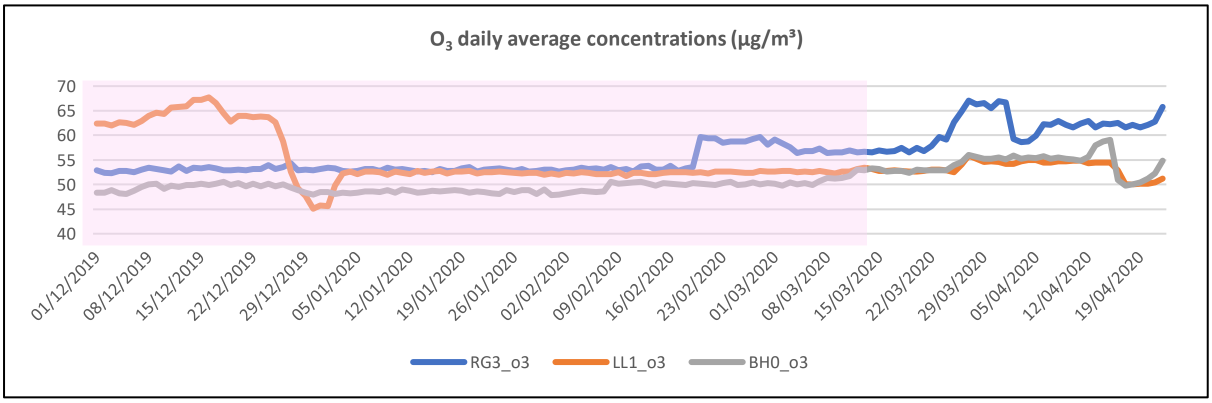

Figure 3.1: De-weathered daily ozone concentrations at background sites in Sussex.

Figures 3.2 and 3.3 show de-weathered O_3 concentrations when compared with NO_2 for the period 1st March to 20th April, which suggests a stepped increase in concentrations from around the 23rd March, whilst NO_2 concentrations remained low. O_3 concentrations may have also been partially influenced by a later short transboundary O_3 pollution episode from 11th to 13th April. A reduction in O_3 can be seen in the model results toward the end of April.

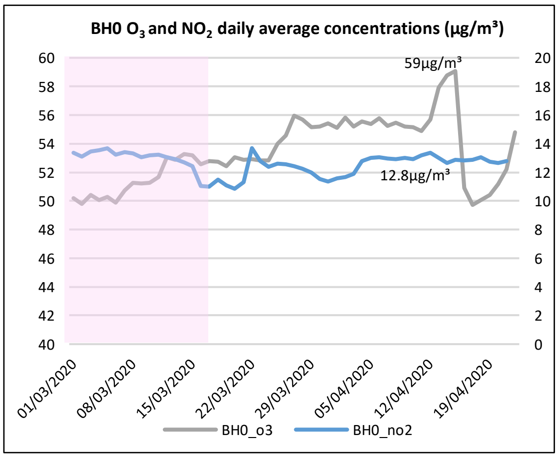

Figure 3.2: BH0 Daily O_3 and NO_2 concentrations

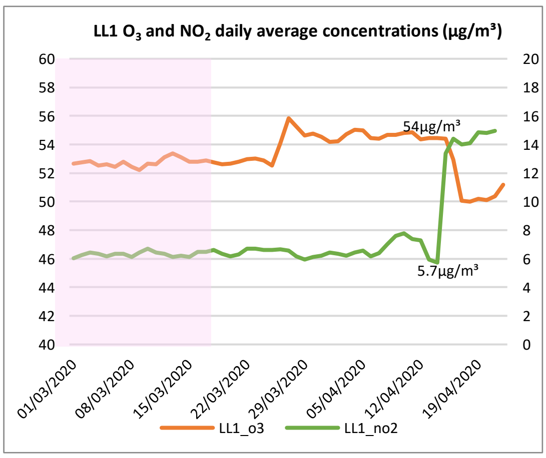

Figure 3.3: LL1 Daily O_3 and NO_2 concentrations

1 Observed measurements are taken directly from AQMS data on Sussex-air and are not currently ratified (24/04/20).
2 Carslaw, D.C. and Ropkins, K. (2012) 'openair - An R package for air quality data analysis', Environmental Modelling & Software, vol. 27-28, pp. 52-61.

4. Traffic reductions across East and West Sussex

Data provided by East Sussex County Council (ESCC) during the period 2nd March to 29th March has identified significantly reduced traffic flows at 50 traffic count sites. Total traffic flows from all these sites during each week are presented in Figure 4.1, with the percentage change. The weekly total traffic flows from week 1 to week 4 declined with reductions of 56% for all 7-day traffic, 51% for weekday traffic and 71% for week-end traffic. West Sussex count sites showed similar reductions of up to 59% for 7-day traffic during the same period.

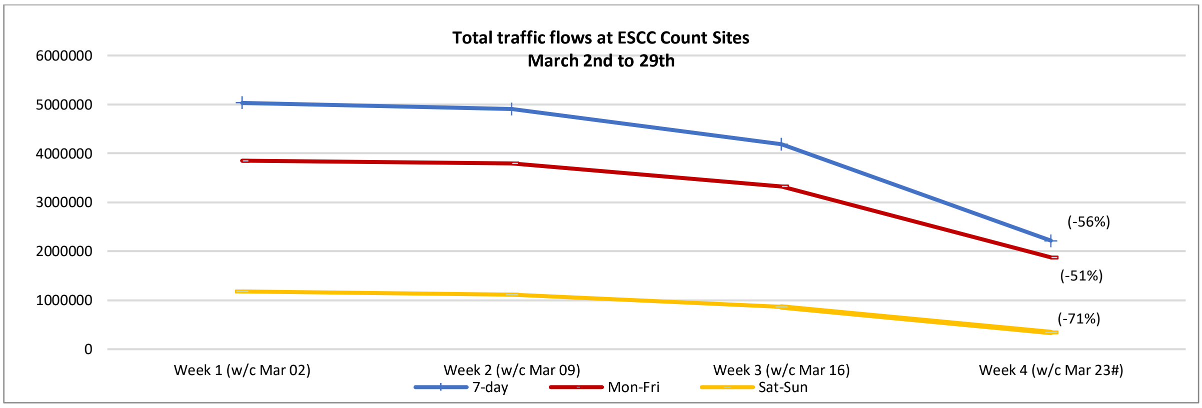

Figure 4.1: Weekly total traffic flows at ESCC count sites - March 2nd to 29th 2020.

This reduction in traffic flows, and hence traffic emissions, has the potential to reduce pollutants such as NO_2 but it could also explain the observed increase in background O_3 concentrations due to reduced availability of NO_x in the atmosphere which would otherwise scavenge O_3, reducing its concentration.

5. Modelled changes in measured nitrogen dioxide (NO_2) concentrations at automatic (continuous) sites.

Figure 5.1 shows the de-weathered daily mean average monitored NO_2 concentrations at 10 automatic (continuous) sites in Sussex and Surrey for the period 1st January to 24th April 2020, inclusive. The data includes both roadside and background sites. While some localised variability is apparent (e.g. LL1 showing increasing concentrations), the overall regional trend shows a decline, starting from approximately 10th March, and a marked step-change from approximately 23rd March following the enforced lockdown period.

The most recent data (from approximately 17th April), appears to show the return of a regional rising trend in NO_2 concentrations although it has not been possible to verify possible causes for this, at this stage.

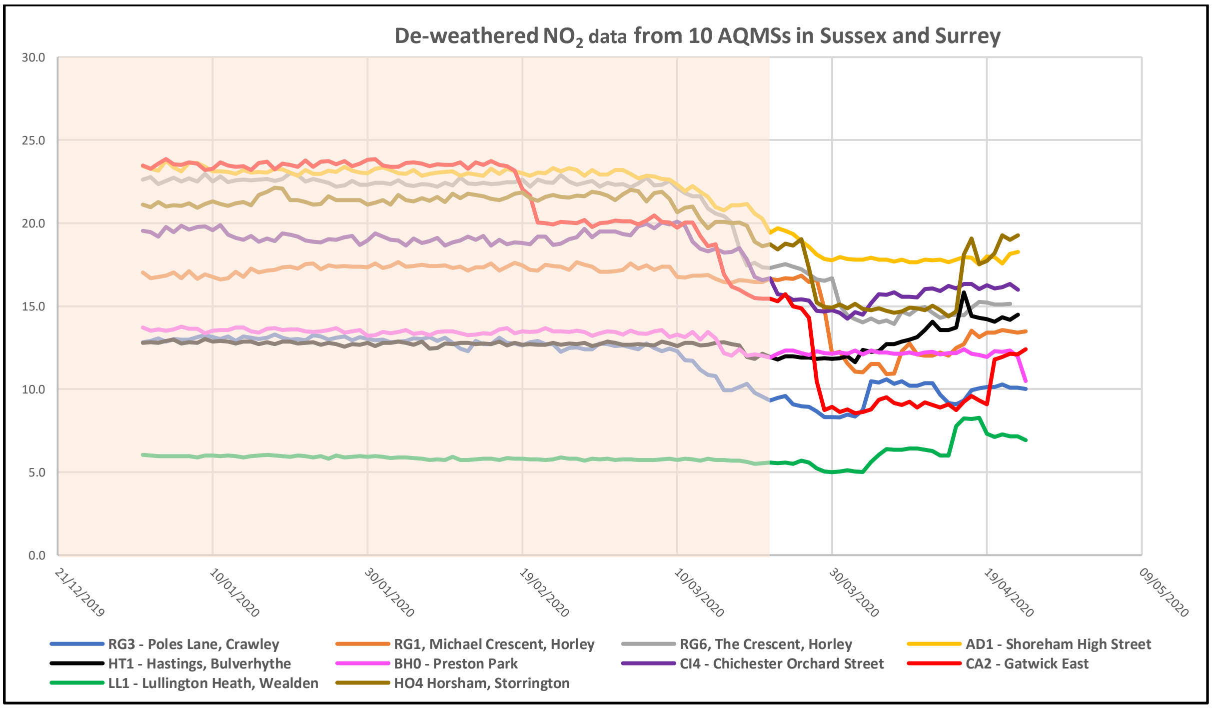

Figure 5.1: De-weathered NO_2 data from 10 AQMSs in Sussex and Surrey

6. Analysis of remote sensing data

Data has also been collected from the University of Brighton's *Joaquin Advanced Air Quality reSearch* (JAAQS) laboratory; a dedicated, permanent atmospheric observatory based on the outer suburbs of Brighton and Hove. JAAQS is equipped with long-path Differential Optical Absorption Spectroscopy (DOAS; Opsis AB; MCerts Certified) for *real-time* remote sensing of trace gas parameters (path length ~300 m), including NO_2, O_3, SO_2 formaldehyde (HCHO), nitrous acid (HONO) and benzene (C_6H_6). JAAQS is also equipped with particle counters (TSI 3031 and TSI 3783), a black carbon monitor (Thermo MAAP 5012), a $PM_{2.5}$ monitor (Met One ES-642) and a meteorology station (Campbell Scientific). Data is presented here only for NO_2, O_3, SO_2, HCHO and HONO; other data is available on request.

Figure 6.1 shows the monthly mean NO_2, O_3, SO_2 and HCHO concentrations between January 2019 and April 2020 in Brighton at the JAAQS suburban background site, with a linear trend estimate in each case (red lines); 95 % confidence intervals are also shown (dashed red lines). The data shows a clear decline in NO_2 concentrations over the period January – April 2020, with post lockdown March and April 2020 concentrations significantly lower than the previous year. In contrast, Figure 6.1 shows a concomitant rise in O_3 concentrations over the same period, to averages higher than the same time the previous year. SO_2 concentrations show a small increase over the year, which appears to accelerate during the lockdown period. HCHO concentrations were observed to decrease during the lockdown period.

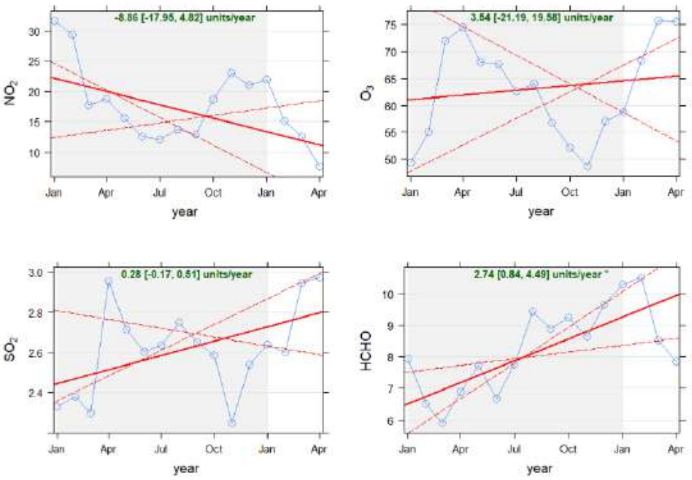

Figure 6.1: Monthly mean JAAQS DOAS NO_2, O_3, SO_2, HCHO concentrations with linear trend, Jan 2019 – April 2020; Brighton

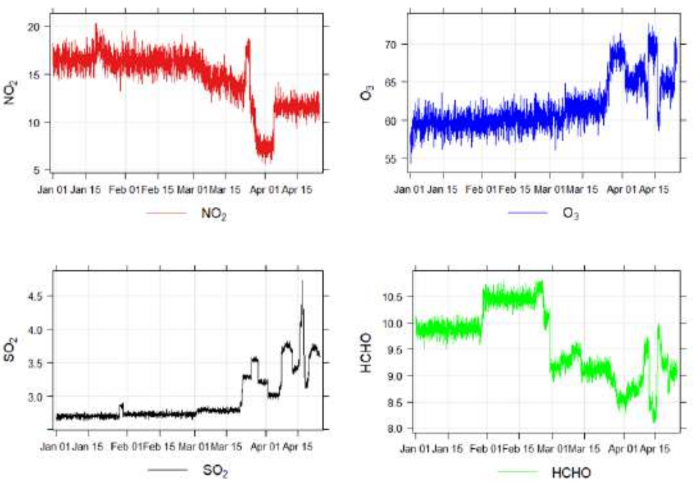

Figure 6.2: Hourly 'de-weathered' JAAQS DOAS NO_2, O_3, SO_2, HCHO concentrations Jan – April 2020; Brighton

Figure 6.2 focuses in on 2020, with 'de-weathered' hourly NO_2, O_3, SO_2 and HCHO data from JAAQS. Detrending the data in this manner extends the analysis presented in Figure 6.1 and demonstrates very clearly the atmospheric trace gas response to government restrictions imposed during the pandemic. With decreasing anthropogenic activity, including road transport in the region (see Section 5), NO_2 concentrations gradually declined at the beginning of March, before spiking between the 24th and 25th March, a day after the lockdown measures were announced. From 25th – 28th March, there was a rapid and continuous decline in NO_2 concentrations, suggesting that the general public were conforming with government guidance. NO_2 concentrations remained at a minimum until the 5th April when they rose again, but to a plateau lower than that observed pre-restrictions. O_3 concentrations were observed to anti-correlate with NO_2, peaking between 28th March and 1st April and then between 12th and 16th April (Figures 6.1 and 6.2). Following well known NO_x-O_3 chemistry, a relative lack of NO_x in the urban atmosphere and relatively cloud-free conditions during the lockdown period, an increase in tropospheric O_3 would be expected. In line with Figure 6.1, the de-weathered SO_2 data displays a small increase during the lockdown period and HCHO a small decrease relative to values at the beginning of the year.

Table 6.1: Monthly average ambient NO_2, O_3, SO_2, HCHO and HONO concentrations for January – April, 2016 – 2020, as measured by DOAS at the JAAQS suburban background site in Brighton

	Monthly Averages (January; February; March; April) / µg.m^{-3}					Average / µg.m^{-3}	% Change
	2016	2017	2018	2019	2020	(2016 – 2019)	
NO_2	23; 26; 20; N/A	44; 29; 23; 27	22; 23; 20; 19	32; 29; 18; 19	22; 15; 13; 11	30; 27; 20; 21	-27; -44; -38; -50
O_3	52; 56; 66; N/A	38; 48; 59; 61	50; 53; 62; 67	49; 55; 72; 75	59; 68; 76; 81	50; 56; 67; 71	+24; +28; +17; +19
SO_2	2; 2; 2; N/A	2; 2; 2; 2	2; 2; 3; 3	2; 2; 2; 3	3; 3; 3; 4	2; 2; 2; 3	+28; +19; +26; +42
HCHO	10; 10; 8; N/A	5; 4; 4; 5	5; 6; 7; 7	8; 7; 6; 7	10; 11; 9; 9	7; 7; 6; 6	+42; +57; +40; +38
HONO*	N/A	N/A	1; 4; 3; 3	1; 5; 3; 3	4; 4; 7; 6	1; 4; 3; 3	+100; +1; +130; + 112

*Data is provisional

Table 6.1 gives the absolute monthly averages for NO_2, O_3, SO_2, HCHO and HONO for the months January – April over the years 2016 – 2020 in Brighton at the JAAQS suburban background site. As demonstrated in Figures 6.1 and 6.2, monthly average NO_2 values during the lockdown period of March and April (*i.e.* 20 and 21 µg m^{-3}) were 38% and 50% lower than the 2016 – 2019 average, respectively; however it should be noted that pre-lockdown, January and February NO_2 values were also lower in 2020 than the average of the previous four years. Monthly average O_3 concentrations in March and April 2020 were 67 and 71 µg m^{-3}, respectively, 17% and 19% up on the 2016 – 2019 averages for these months; as with NO_2, pre-lockdown, January and February O_3 values were also higher in 2020 than the average of the previous four years. Absolute SO_2 values were up by 26% and 42% (*i.e.* 2 and 3 µg m^{-3}) during the lockdown months of March and April, respectively, and HCHO values by 40% and 38% (*i.e.* 6 and 6 µg m^{-3}); increased HCHO may be a result of increased atmospheric reactivity owing to higher O_3 concentrations and cloud-free conditions, and requires further investigation.

Figure 6.3 shows the average diurnal, monthly and weekly concentrations of ambient NO_2, O_3, SO_2 and HCHO before 1st March 2020 (red) and after 1st March 2020 (blue) at the JAAQS suburban background site in Brighton. Figure 6.3 clearly demonstrates the fall in ambient NO_2 concentrations during the March and April period of government-imposed restrictions and the increase in O_3, SO_2 and HCHO discussed previously. Generally, the diurnal profiles for each trace gas remained similar in terms of shape during both periods, with the exceptions being formaldehyde and NO_2, where for the latter, the evening peak was relatively smaller that its morning counterpart after 1st March 2020, suggesting the reduction in anthropogenic activity in the region is relatively greater in the evening than the morning. For NO_2 and O_3, weekly profiles were similar during both periods, with a clear 'weekend effect' of lower NO_2 and higher O_3 concentrations during the weekend than the working week. This weekend effect was comparable during both periods, suggesting a similar reduction in level of anthropogenic activity during the lockdown period compared to normal.

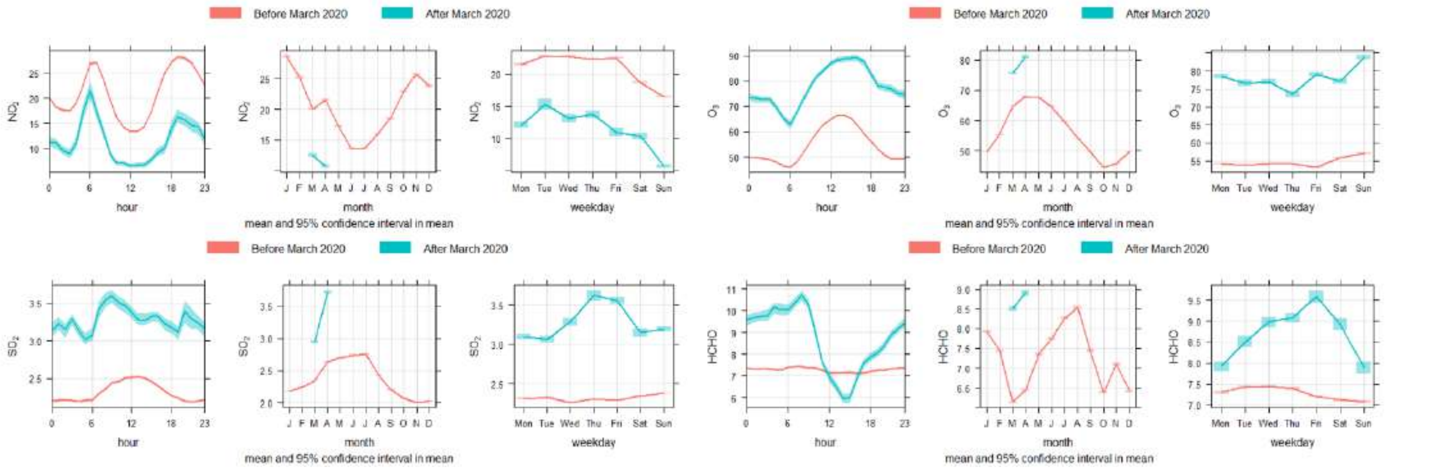

Figure 6.3: Average diurnal, monthly and weekly concentrations of ambient NO_2, O_3, SO_2 and HCHO before 1st March 2020 (red) and after 1st March 2020 (blue) as measured by DOAS at the JAAQS suburban background site in Brighton.

Figure 6.4 shows regional daily average NO_2 concentrations as recorded by TROPOMI onboard ESA's Sentinel-5P satellite over the period 25th March to 22nd April 2019 (a) and 23rd March to 20th April 2020 (b); the percentage change between the two periods is also shown (c), as are the regionally integrated values (d). To obtain the NO_2 averages, the L2 data products were filtered to remove problematic observations (*i.e.* errors, cloud cover), and scaled for clarity. The scaled, filtered products were totalled, with the sum for each pixel divided by the number of times that pixel was observed throughout the period.

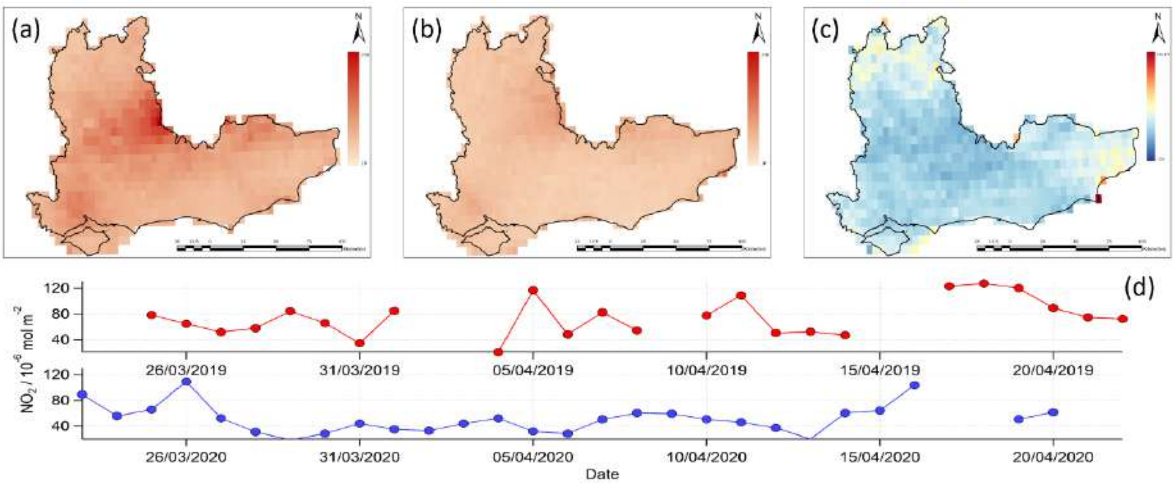

Figure 6.4: TROPOMI NO_2 data averaged over the period 25th March to 22nd April 2019 (a) and 23rd March to 20th April 2020 (b); percentage change between the two periods (c); regionally integrated NO_2 values (d). (Data source: Copernicus Sentinel-5P Pre-Operations Data Hub).

Figure 6.4 demonstrates the change in spatial distribution of ambient NO_2 concentrations over the entire south east region of the UK during the lockdown period with respect to the same period during 2019. In-line with other data presented within this report, it is clear that concentrations of NO_2 have decreased across the entire region during the pandemic, with the regional averages going from 74 x 10^{-6} to 51 x 10^{-6} mol m^{-2}, *i.e* a decrease of 31%. As shown in Figure 6.4 (c), the largest changes were observed in the centre of the region in the areas surrounding London and in certain coastal locations. Data integrated over Brighton and Hove shows local city values have dropped from 66 x 10^{-6} to 45 x 10^{-6} mol m^{-2}, *i.e.* a decrease of 32%, which is in-line with measurements made by DOAS over the same period within the city (*i.e.* a decrease of 43%).

7. Summary and Conclusions

This report provides estimates for how emissions and ambient concentrations of NO_x/NO_2, O_3, SO_2, HCHO and HONO have changed across Sussex and Surrey as a result of the COVID-19 outbreak. Data presented indicates a regional inverse relationship between ambient NO_2 and tropospheric O_3 concentrations in the south east, which is not a-typical, and is apparent in all de-weathered model results. However, as a likely consequence of reduced primary NO_x emissions due to COVID-19 travel restrictions, initial findings suggest that baseline O_3 concentrations are above average.

Further research is needed to investigate the impact of increased atmospheric reactivity owing to higher O_3 concentrations, which are likely due to established NO_x-O_3 chemistry. Increased summertime biospheric VOC emissions are also likely to be a contributing factor. Work on formaldehyde and atmospheric reactivity will continue and be reported at a later date.

Air Quality in the UK during the COVID-19 pandemic – evidence from national monitoring stations

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Introduction

The current Coronavirus (COVID-19) outbreak was first identified in Wuhan, China, in December 2019, and was recognized as a pandemic by the World Health Organization (WHO) on 11 March 2020. As of 17 April 2020, more than 2.2 million cases had been reported in 190 countries, resulting in approximately 150,000 deaths. Efforts to prevent the virus spreading have included severe travel restrictions and the closure of workplaces, inevitably leading to a significant drop in emissions of air pollutants. In the UK, significant events of the pandemic have been:

- 30th January: first cases appear in the UK.
- 28th February: first transmission within the UK documented.
- 6th March: first death in the UK reported.
- 11th March: WHO declare a pandemic.
- 16th March: Government statement to avoid all non-essential travel and contact with others, avoid crowds, and work from home if possible.
- 20th March: Schools, restaurants, pubs, clubs, and indoor leisure facilities closed.
- 23rd March Full lockdown imposed.

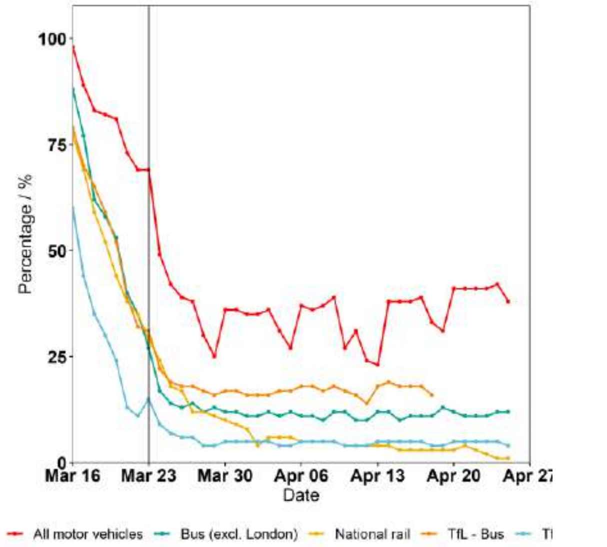

Figure 1: *Change in transport use in the UK.*

An immediate effect of the restrictions was a drop in transport use. Figure 1 shows UK transport use since late February and demonstrates the large reduction in motor vehicles, starting on around 17th March and continuing throughout the lockdown period. This report will examine how these reductions (and other changes) have affected air pollution in the UK.

The DEFRA Automatic Urban and Rural (AURN) network consists of 150 currently active sites across the UK (see map in Figure 2) and is the main network used for compliance reporting against the Ambient Air Quality Directives. It includes automatic air quality monitoring stations measuring oxides of nitrogen (NO_x), sulphur dioxide (SO_2), ozone (O_3), carbon monoxide (CO) and particles (PM_{10}, $PM_{2.5}$). Online measurements of volatile organic compounds (VOCs) are available at a small amount of sites. These sites provide high resolution hourly information which is communicated rapidly to the public, using a wide range of electronic, media and web platforms. More detail, including the techniques used for monitoring within the AURN can be found at https://uk-air.defra.gov.uk. AURN data was accessed using the **importAURN**() function from the **openair** R package. Data from the past 6 months has not yet been ratified and may change during the QA/QC process

Figure 2: *Map showing the location of all the AURN sites. The labelled sites identify those with online hydrocarbon measurements used in this analysis.*

NO_2 and $PM_{2.5}$ data

At present the main pollutants of concern in the UK in urban centres are nitrogen dioxide (NO_2) and particles less than 2.5µm in diameter, measured by their mass ($PM_{2.5}$), alongside ozone (O_3) in suburban and rural environments. As these are routinely measured at most of the AURN sites we have concentrated on examining their response during the COVID-19 lockdown period. Data is examined in terms of bulk averages of sites across the UK and in some selected cities.

All sites

Initially data was analysed based on the site type with Urban Traffic and Urban Background sites chosen for the analysis. Urban Traffic sites are defined as being in continuously built-up urban areas, with pollution levels predominantly influenced by emissions from nearby traffic. Urban background sites are located such that pollution level is not influenced significantly by any single source or street, but rather by the integrated contribution from all sources upwind of the stations. These can be considered more representative of residential areas.

Figure 3 presents an 'Air Quality Stripes' time series aggregated from all 97 Urban Traffic sites in the AURN network from 2015 to 27th April 2020. It can be seen, especially for NO_2, that concentrations have been lower since early March, compared to earlier in the year and to the previous years. The situation is less clear for $PM_{2.5}$ although it does appear that levels in 2020 are lower than previous years. While it is not easy to quantitatively assess air pollution levels using this type of plot, it does show the general picture, and confirms that the average of the previous 5 years (2015-2019) is a reasonable comparator to use for assessing the effect of the COVID-19 lockdown, despite the general trend of reducing NO_2 and $PM_{2.5}$ in the UK.

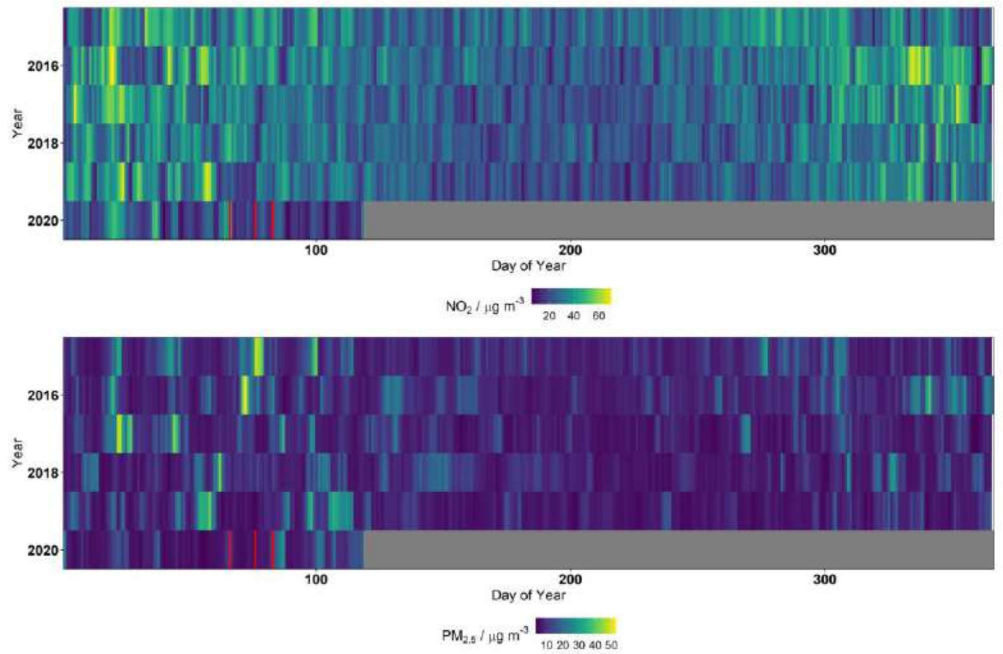

***Figure 3**: Daily median NO_2 (top) and $PM_{2.5}$ concentrations (bottom – both mg m^{-3}) averaged for all urban traffic sites in the AURN network. The red vertical lines indicate 6th, 16th and 23rd March.*

In order to better assess the magnitude of changes in NO_2 and $PM_{2.5}$, data has been averaged into two periods. The pre lockdown period is from 10th Jan – 5th March, chosen as it is the day before the first UK deaths were announced. The post-lockdown period is from 16th March, when the advice to avoid all non-essential travel and contact with others was given. Figure 4 compares average concentrations for these periods across all Urban Background and Urban Traffic sites. Comparing the pre and post-lockdown period, there is a reduction in NO_2 concentrations at both types of site, although the error bars are large. For $PM_{2.5}$ there is actually a slight increase post-lockdown. Comparing the two periods must be taken in context of other factors that affect air pollution levels, most notably the meteorology. The start of the year was influenced by strong westerlies with a lot of rain and wind. This reduces particulate a lot and disperses NO_2 compared to previous years. This lasted until early March. Since lockdown the UK has had almost constant high pressure and easterly or south easterly winds which have varied in strength. The UK has become a receptor site for wider European pollution. This metrological effect is particularly important for PM, which has a wider variety of sources than NO_2 and can be transported to the UK from Europe on Easterly winds. There are mathematical techniques that can take out the effect of meteorology in order to provide a measure of the change in air pollutants compared to what would be expected for given conditions, however this analysis is outside the scope of this report. For this reason, we chose to base the remainder of our analysis on comparing the lockdown period to the average of the equivalent period for the previous 5 years.

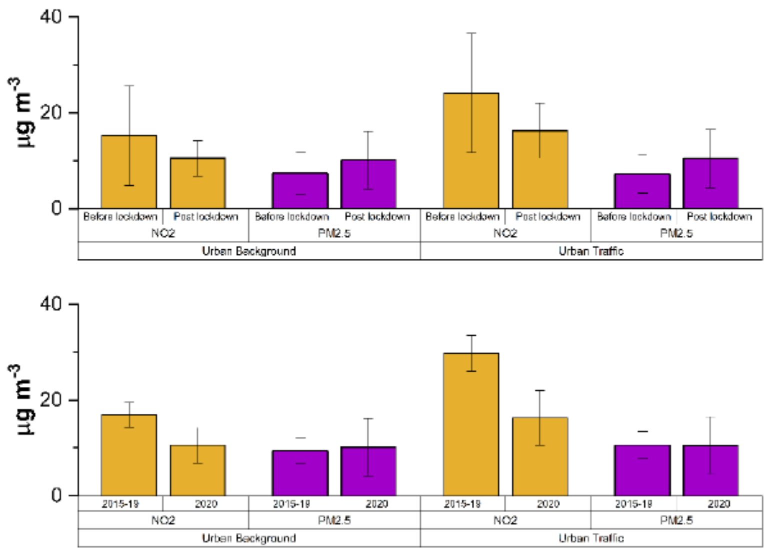

***Figure 4**: Mean of the daily median NO_2 and $PM_{2.5}$ concentrations taken across all Urban Background and Urban Traffic sites for the pre (10th January – 10th March) and post (17th March – 2th^{d} April) lockdown period (top panel) and comparing the post-lockdown period from 2020 to the equivalent period over the previous 5 years. Error bars indicate the standard deviation of the mean data.*

On average, NO_2 has decreased by 38 and 45% at Urban Background and Urban Traffic sites respectively during the lockdown period, compared to the previous 5 years. $PM_{2.5}$ is virtually unchanged at Urban Traffic sites and there is a slight (8%) increase for Urban Background. The reduction in traffic flow (as shown in Figure 2), has clearly had an effect on ambient NO_2 levels. The majority of NO_2 in cities (~60-80% according to the National Atmospheric Emissions Inventory) comes from vehicle exhausts, so reductions are expected when traffic levels decrease. The situation for $PM_{2.5}$ is clearly very different. Traffic emissions contribute a smaller proportion of $PM_{2.5}$ in UK cities than for NO_2 (~20-40%), with other major sources being agriculture, other industry and solid fuel burning. As $PM_{2.5}$ is generally longer liver than NO_2, it can also be transported from further afield, therefore the easterly flows we have been experiencing during the lockdown period will have brought $PM_{2.5}$ from Europe to the UK. Figure 5 shows average diurnal cycles for NO_2 and $PM_{2.5}$ across all Urban Background and Urban Traffic AURN sites for the pre and post lockdown periods and the lockdown period averaged for 2015-2019. We see that, for both species at Urban Traffic sites the cycle is very similar with rush hour peaks in the morning and evening. There is some evidence that at Urban Background sites post-lockdown, the morning rush hour is slightly repressed, especially for $PM_{2.5}$. Further analysis using data from individual sites is required to properly assess the difference in diurnal cycles and how these link to traffic flow and other behavioural changes during the lockdown.

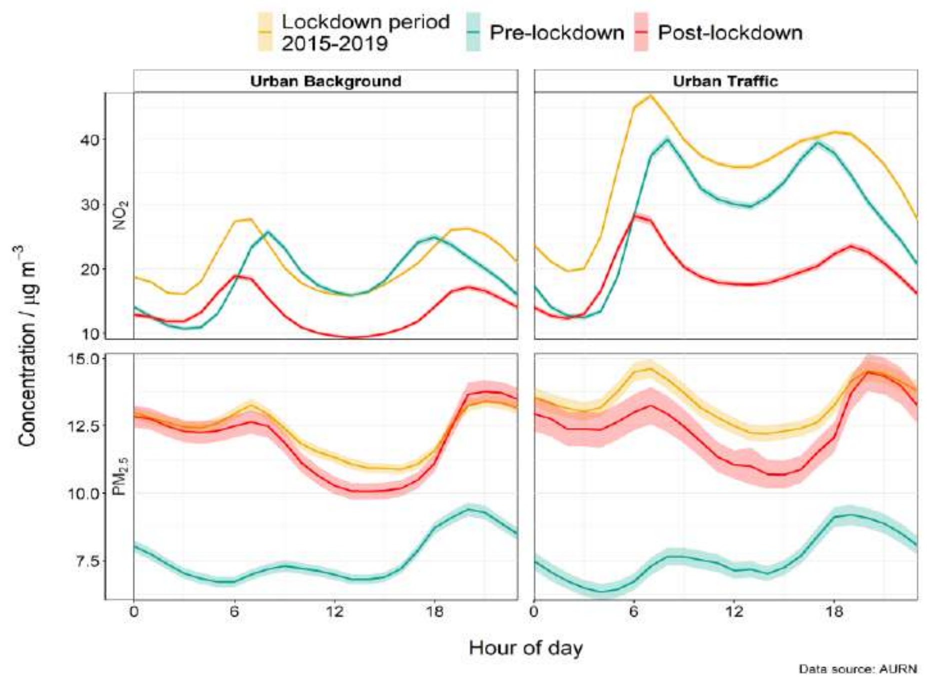

***Figure 5:** Average diurnal cycles for NO2 (top) and PM2.5 (bottom) for all data at Urban Background (left) and Urban Traffic (right) sites. Data is calculated for 2015-2019 (green) and 2020 (blue) for the lockdown period (17th March – 2th^{d} April).*

Individual cities

We have also examined the NO_2 and $PM_{2.5}$ changes in different cities across the UK. We aggregated data from Urban Background sites in nine cities and the time series are shown in figure 6. Graphs presenting the averaged data for the lockdown period this year and the previous 5 years, along with percentage changes are shown in figure 7.

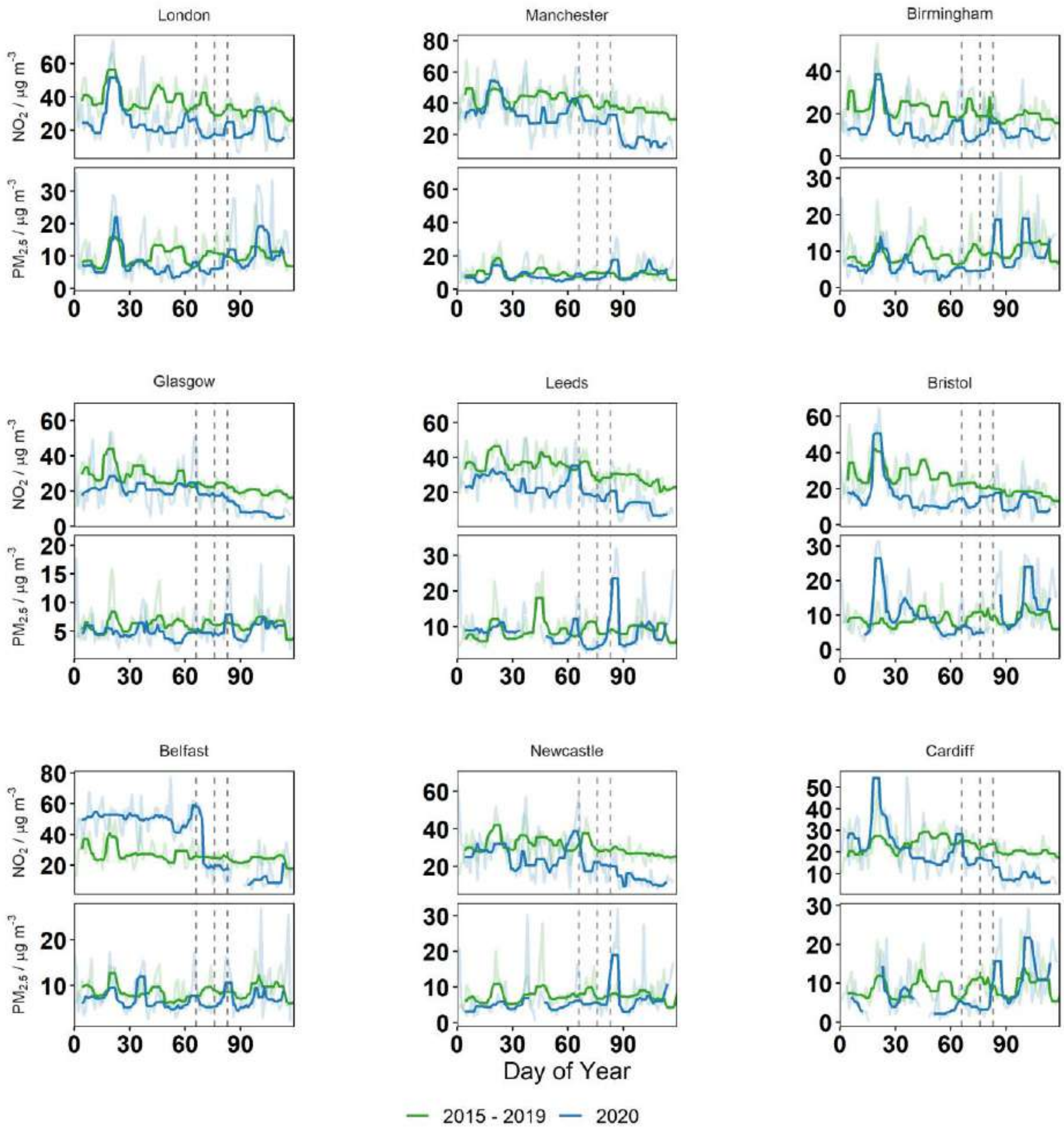

Figure 6: *Daily median values of NO_2 and $PM_{2.5}$ across all urban background AURN sites in a given cities. The thick line is the 7 day rolling median. Vertical lines indicate 6th, 16th and 23rd March.*

The time series data in Figure 6 shows some contrasting behaviour across cities. For NO_2, cities in the south (London, Bristol and Birmingham) all show a large peak in January but otherwise concentrations are significantly less during 2020 than previous years throughout the whole of the year (not just since lockdown) and cities in the NE (Newcastle and Leeds) show significantly less NO_2 for the whole year. In the west and north (Manchester, Glasgow and Cardiff) the trends show agreement with previous years until lockdown. It is hard to see much of a trend for $PM_{2.5}$ for any of the average city plots. Taking averages (Figure 7), shows that for the pre and post lockdown period, Belfast and Cardiff show the greatest decrease (62 and 57%), with less reductions for cities in the South of England (Bristol, Birmingham and London). Looking at the change in the lockdown period in 2020 compared to 2015-2019, the picture is actually broadly similar for all cities. The greatest NO_2 reduction is seen in Leeds (49%) and Cardiff (48%) with lower reductions in Belfast and Bristol (25 and 31% respectively). Again the situation for $PM_{2.5}$ is very different, with increases of 10 %

in Manchester and 10% in Bristol. The exact numbers in this analysis should be treated with caution due to the relatively short time series meaning percentage reductions can changes quite significantly on a day by day basis as new data is added. However, the general picture is clear and as some restrictions are likely to continue in some form for many weeks. Continued analysis will allow us to quantify the effect on NO_2 and $PM_{2.5}$ of this unprecedented reduction in traffic and other activity. Future work should look at Urban Traffic and local authority sites in the individual cities to see if a pattern to the changes in pollution can be ascertained.

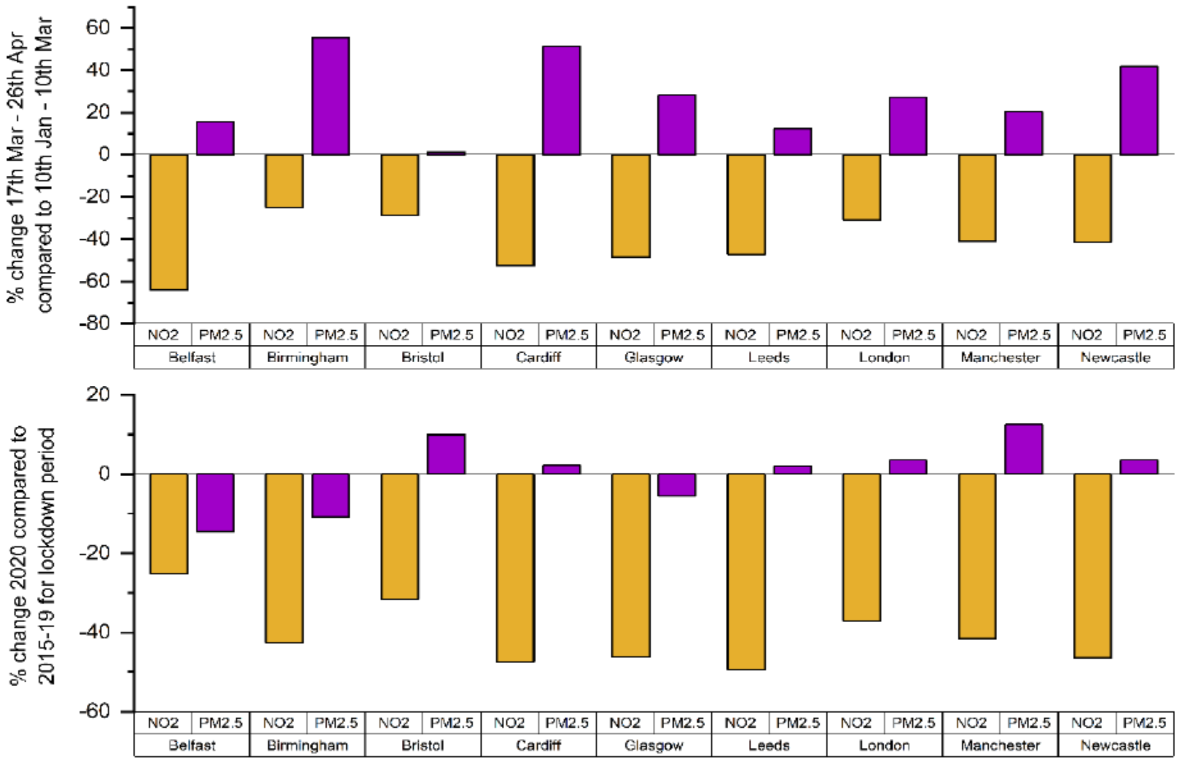

Figure 7: *Percentage change in NO2 (orange and PM2.5 (purple) for 9 UK cities for pre and post lockdown period (16th March) (top) and for the lockdown period in 2020 and compared to 2015-2019 (bottom).*

Effect on AQ Exceedances

There are a series of air quality objectives and target values set out by the UK government, the WHO and European directives specifically for the protection of human health. We have analysed data from All AURN sites to assess whether there has been a change in the number of exceedances during the COVID19 outbreak and associated lockdown. We compared data from March and April 2020 with the same period in 2019. The exceedances we looked at are:

- 1 hour mean NO_2 of 200µg m^{-3} (EU directive - not to be exceeded more than 18 times in a year)
- 24 hour mean $PM_{2.5}$ of 25µg m^{-3} (WHO recommendation)

We analysed data separately for London (using data from 380 sites in the London Air Quality Network), and the rest of the UK (using 70 AURN Urban Traffic sites). Results are shown in Figure 8; exceedances presented are the top 10 sites from 2019. It is clear that the number of exceedances at these sites has decreased for 2020 compared to 2019. For NO_2 in London there has only been one exceedance at one of these sites this year, compared to 61 in 2019. For the rest of the UK there have only been 5 exceedances (at the Hafod-yr-ynys Roadside site in South Wales), although the data is more limited due to the small number of NO_2 exceedances at AURN sites across the UK in 2019. There are also drops in the WHO 24 hour mean $PM_{2.5}$ number of exceedances in London and across the UK. This is despite the fact that the earlier analysis showed only a small decrease in $PM_{2.5}$ at Urban Background site on average across the country. This potentially shows that there is an effect of the lockdown on $PM_{2.5}$ at the most polluted Roadside sites, greatly reducing the exceedances and thus having a positive health effect. Further analysis is required on this, especially once we can analyse the full year of ratified data.

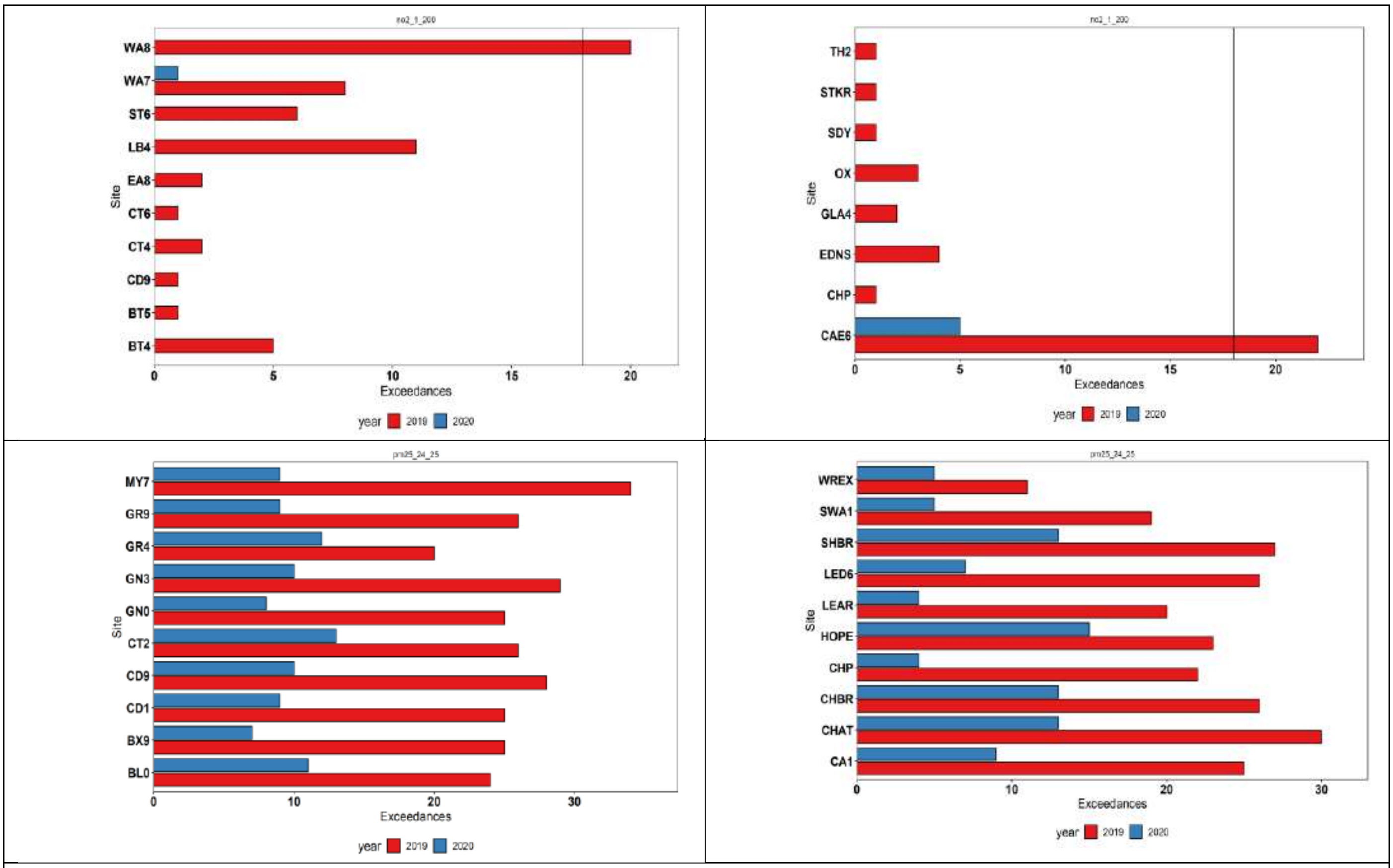

***Figure 8**: Air quality guideline exceedances by site in March and April 2019 and 2020. Left; London Air Quality Network Sites. Right; Automatic Urban and Rural Network sites, excluding those in London; top = NO_2 bottom = $PM_{2.5}$ / µg m^{-3}. Vertical bars represent number of "allowed" exceedances per year where applicable. Up to 10 sites in each category are shown*

<u>Volatile Organic Compounds</u>

VOCs have both biogenic and anthropogenic sources although it is estimated that less than 5% of VOCs are emitted from vegetation in the UK. VOCs are important precursors in the formation of tropospheric O_3 due to the reaction with the OH radical in the presence of NO_x and sunlight. The wide range of lifetimes of VOCs means they can be transported large distances before reacting, often leading to impacts on air quality downwind of the emission source. Additionally, some compounds are directly linked to human health concerns due to their carcinogenic nature. The most important of these are benzene and 1,3-butadiene and hence target exposure limits exists in the UK to control emissions of these species. VOC measurements in the UK are limited to a small number of sites. Here we analyse data from London Eltham, a Suburban Background site and London Marylebone Road, an Urban Traffic site.

Diurnal variation

The diurnal variation at London Eltham is driven by changes in the boundary layer (BL) depth throughout day. This leads to a build-up of concentrations overnight when temperatures decrease and the BL is shallow and a decrease in concentrations throughout the day when surface heating results in a deeper, more diluted BL. In contrast at Marylebone Road, concentrations are heavily influenced by traffic sources, leading to a distinctive diurnal cycle of some species, where concentrations peak during the morning and evening rush hour. The diurnal cycle of selected VOCs, shown in Figure 9, reveals a suppressed cycle for pollutants at Marylebone Road. Benzene and n-Butane show a distinctive cycle associated with traffic emissions, which is much flatter post-lockdown due to the reduction in traffic flow. No such change is observed at London Eltham, where the diurnal cycle of all species remains very similar during the pre and post-lockdown periods. This shows that local emission of VOCs associated with traffic sources are likely to have decreased along with NO_x emissions. However regional VOC concentrations appear to be largely unaffected by

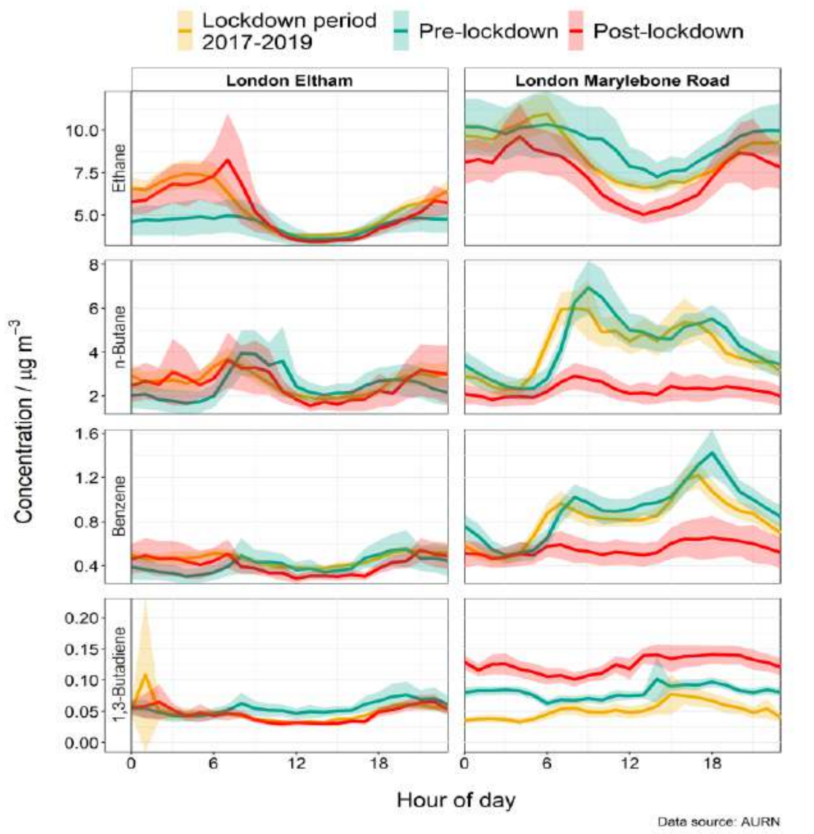

***Figure 9:** Average diurnal cycles for selected VOCs at during the pre and post-lockdown periods in 2020 as well as the average of the lockdown period between 2017-2019. Shaded areas show the upper and lower 95% confidence intervals.*

lockdown measures up to this point and are primarily controlled by weather conditions. 1,3 – Butadiene shows contrasting behaviour in that it is higher during the lockdown period at the Marylebone Road site, both compared to pre lockdown and previous years. The diurnal cycle and lack of a morning rush hour peak suggests different sources other than traffic which will need further investigation.

Total VOC concentrations

Total daily VOC concentrations show an overall reduction at Marylebone Road when compared to the average of the total daily concentrations between 2017-2019. Figure 10 shows that most days showed a decrease after restrictions began. The opposite is seen at London Eltham where there is no clear difference in total VOC loading. Total VOC between the pre and post-lockdown periods increased by an average of 4% at Eltham, whereas a decrease of 38% was observed at Marylebone Road. This again reflects the effect of reduced traffic evident in local VOC concentrations but not regional concentrations.

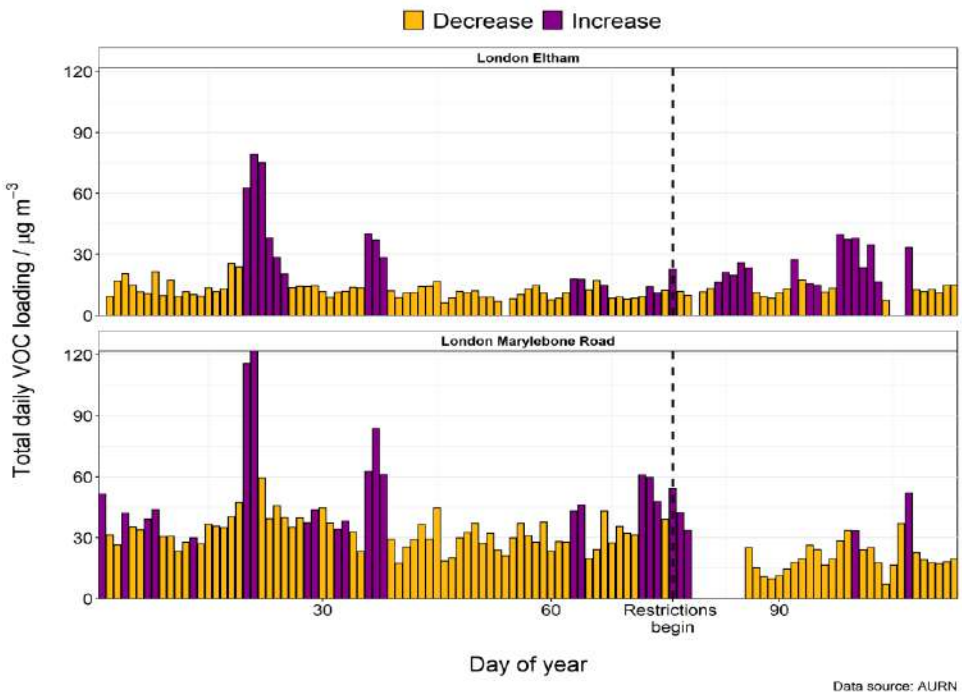

***Figure 10:** Daily total VOC loading for both sites calculated from daily averaged data in 2020 compared to the average of the total daily loading during 2017-2019.*

Ozone

O_3 is formed through a series of chemical reactions from NO_x and VOCs in the presence of sunlight. It its therefore dependant not only on emissions but the balance of different VOCs, NO_x and radiation. We have analysed daytime O_3 concentrations in the same way as for NO_2 and $PM_{2.5}$ and there is an increase for the 2020 lockdown period compared to the previous 5 years. Figure 11 shows daytime concentrations of O_3, NO_2 and O_x (NO_2 + O_3) for the lockdown period for 2015-2019 and 2020, plotted for London and Glasgow as example cities. Throughout the year we observe an increase in O_3 that broadly anti-correlates with NO_2. This O_3 increase appears largely to be caused directly by the reduction in NO_x, with net oxidants (O_x) being conserved. There is a period in early April in London where O_x is increased compared to the previous years, but this could be due to the warm, sunny weather at this time. O_3 will continue to increase during spring and summer due to its seasonal cycle and we will continue to monitor data to assess if the increase becomes more significant. It has been noted in China that reductions in PM have contributed to increased O_3 due to reduction in radical loss to particles and we will attempt to assess if this is important for UK O_3.

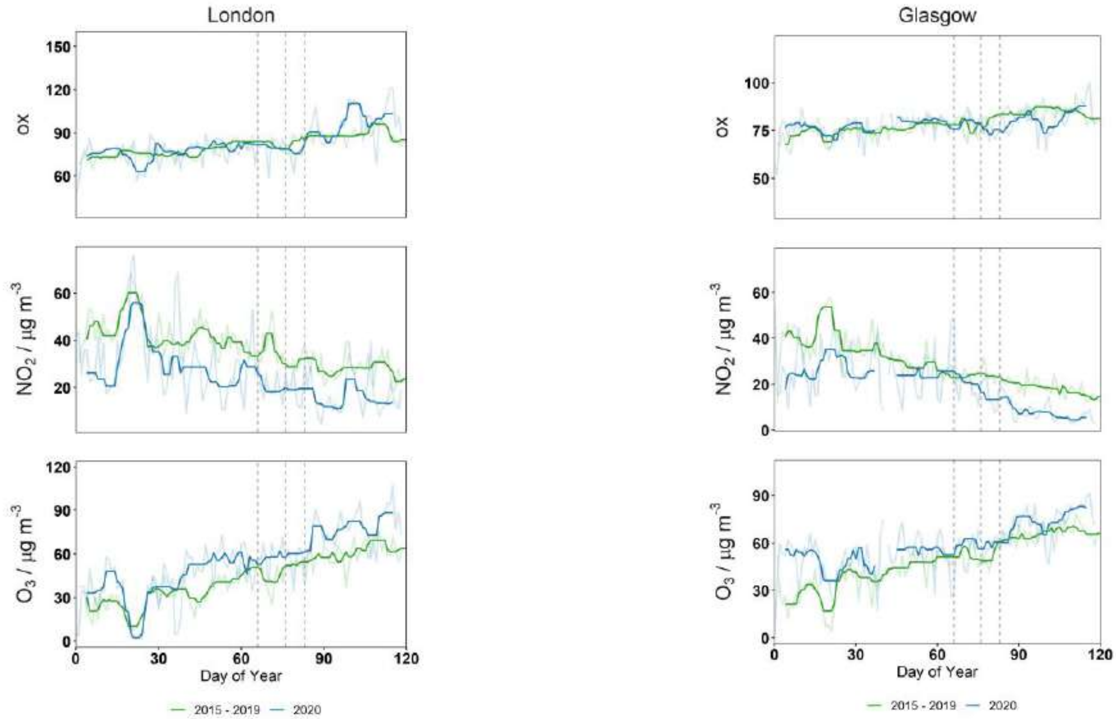

Figure 11: *Daytime median values of NO_2 and O_3 across all urban background AURN sites in London and Glasgow. Thick line is the 7 day rolling median. Vertical lines indicate 6th, 16th and 23rd March.*

Summary

We have analysed NO_2 and $PM_{2.5}$ (and VOC and O_3) data from a series of AURN and LAQN sites since the lockdown period of the COVID19 outbreak. In order to attempt to account for the effect of meteorology we have compared levels in the lockdown period with the average of the same period over the previous 5 years. There are significant reductions in NO_2 across the country, with the largest reductions being at sites close to roads (Urban Traffic – average of 45%). Urban background sites see a reduction of 38% on average. The situation for $PM_{2.5}$ is more complicated due to the greater number of different sources and longer range transport on the Easterly winds experience during the lockdown period. We see a virtually no change in $PM_{2.5}$ on average at Urban Traffic sites and a slight (8%) increase in the Urban Background. Examining the number of exceedances at sites across the country for March and April 2020 shows a decrease for NO_2 EU limit exceedances across the country compared to the same period in 2019. We also see a reduction in the exceedances of the WHO recommended $PM_{2.5}$, showing that the lockdown is having an effect on $PM_{2.5}$ at the most polluted Roadside sites, despite the lack of a significant reduction on average across the country.

This analysis provides us with a unique window into what air pollution might be like in cities with a largely electrified transport system (as may happen in 10-15 years), and should help us to answer such questions as will cities be able to meet the WHO recommended limit for $PM_{2.5}$ of 10µg m^{-3} and how will NO_2 respond to changes in the vehicle fleet?

Appendix

Table A1: List of Urban Traffic sites used in analysis.

site	code	latitude	longitude
Aberdeen Union Street Roadside	ABD7	57.144555	-2.106472
Aberdeen Wellington Road	ABD8	57.133888	-2.094198
Armagh Roadside	ARM6	54.353728	-6.654558
Ballymena Antrim Road	BAAR	54.851491	-6.274961
Barnstaple A39	BPLE	51.074793	-4.041924
Bath A4 Roadside	BHA4	51.390922	-2.35503
Bath Roadside	BATH	51.391127	-2.354155
Belfast Stockman's Lane	BEL1	54.572586	-5.974944
Birkenhead Borough Road	BBRD	53.388511	-3.025014
Birmingham A4540 Roadside	BIRR	52.47609	-1.875024
Birmingham Kerbside	BHAM	52.328057	-1.907512
Birmingham Tyburn Roadside	BIRT	52.512194	-1.830861
Blackburn Accrington Road	BLAR	53.747751	-2.452724
Blackburn Darwen Roadside	BLCB	53.715504	-2.483815
Bradford Mayo Avenue	BDMA	53.771245	-1.759774
Brentford Roadside	BRN	51.489448	-0.310121
Brighton Roadside	BRIT	50.82354	-0.137281
Bristol Old Market	BRS2	51.45603	-2.583519
Bristol Temple Way	BR11	51.457968	-2.583975
Bromley Roadside	BY1	51.4071	0.020128
Bury Roadside	BURY	53.53911	-2.289611
Bury Whitefield Roadside	BURW	53.559029	-2.293772
Cambridge	CAMB	51.995804	0.037756
Cambridge Roadside	CAM	52.20237	0.124456
Camden Kerbside	CA1	51.54421	-0.175269
Cannock A5190 Roadside	CANK	52.687298	-1.980821
Cardiff Kerbside	CAR	51.482431	-3.17794
Cardiff Newport Road	CNPR	51.49096	-3.152305
Carlisle Roadside	CARL	54.894834	-2.945307
Chatham Roadside	CHAT	51.374264	0.54797
Chepstow A48	CHP	51.638094	-2.678731
Chesterfield Roadside	CHS7	53.231722	-1.456944
Christchurch Barrack Road	CHBR	50.735454	-1.780888
Coventry Binley Road	COBR	52.407708	-1.490082
Derby St Alkmund's Way	DESA	52.922983	-1.469507
Doncaster A630 Cleveland Street	DCST	53.51824	-1.138057
Dumbarton Roadside	DUMB	55.943197	-4.55973
Dumfries	DUMF	55.070033	-3.614233
Ealing Horn Lane	EA8	51.51895	-0.265617
Edinburgh Nicolson Street	EDNS	55.94476	-3.183991
Exeter Roadside	EX	50.725083	-3.532465
Glasgow Great Western Road	GGWR	55.872038	-4.270936
Glasgow High Street	GHSR	55.860936	-4.238214

Glasgow Hope St	GLAS	55.858317	-4.259062
Glasgow Kerbside	GLA4	55.85917	-4.258889
Greenock A8 Roadside	GKA8	55.944079	-4.734421
Hafod-yr-ynys Roadside	CAE6	51.680579	-3.133508
Haringey Roadside	HG1	51.5993	-0.068218
Hounslow Roadside	HS1	51.48965	-0.308975
Hove Roadside	HOVE	50.82778	-0.170294
Hull Holderness Road	HULR	53.758971	-0.305749
Inverness	INV2	57.481308	-4.241451
Leamington Spa Rugby Road	LEAR	52.294884	-1.542911
Leeds Headingley Kerbside	LED6	53.819972	-1.576361
Leicester A594 Roadside	LEIR	52.638677	-1.124228
Lincoln Canwick Road	LIN3	53.221373	-0.534189
Lincoln Roadside	LINC	53.22889	-0.537895
Liverpool Queen's Drive Roadside	LV6	53.446944	-2.9625
London A3 Roadside	A3	51.37348	-0.291853
London Bromley	BY2	51.40555	0.018869
London Cromwell Road	CRD	51.49492	-0.180564
London Cromwell Road 2	CRD2	51.495483	-0.178709
London Marylebone Road	MY1	51.52253	-0.154611
Luton A505 Roadside	LUTR	51.892293	-0.46211
Newcastle Cradlewell Roadside	NCA3	54.986405	-1.595362
Norwich Forum Roadside	NO10	52.62817	1.291714
Norwich Roadside	NOR1	52.622	1.299064
Nottingham Western Boulevard	NWBV	52.969377	-1.188851
Oldbury Birmingham Road	BOLD	52.502436	-2.003497
Oxford Centre Roadside	OX	51.751745	-1.257463
Plymouth Tavistock Road	PLYR	50.411058	-4.130288
Portsmouth Anglesea Road	POAR	50.798339	-1.095558
Reading London Road	REA5	51.454896	-0.940382
Saltash Callington Road	SASH	50.411463	-4.227678
Saltash Roadside	SALT	50.4131	-4.2303
Sandy Roadside	SDY	52.132417	-0.300306
Shaw Crompton Way	CW	53.579283	-2.093786
Sheffield Barnsley Road	SHBR	53.40495	-1.455815
Southampton A33	SA33	50.920265	-1.463484
Southwark A2 Old Kent Road	SK5	51.480499	-0.05955
Southwark Roadside	SK2	51.48199	-0.0623
St Helens Linkway	SHLW	53.451826	-2.742134
Stanford-le-Hope Roadside	HOPE	51.518167	0.439548
Stockton-on-Tees A1305 Roadside	SOTR	54.565819	-1.3159
Stockton-on-Tees Eaglescliffe	EAGL	54.516667	-1.358547
Stockton-on-Tees Yarm	YARM	54.50918	-1.354319
Stoke-on-Trent A50 Roadside	STKR	52.980436	-2.111898
Storrington Roadside	STOR	50.916932	-0.449548
Sunderland Wessington Way	SUNR	54.91839	-1.408391
Sutton Roadside	SUT1	51.36636	-0.182789
Swansea Roadside	SWA1	51.632696	-3.947374

Tower Hamlets Roadside	TH2	51.52253	-0.042155
Widnes Milton Road	WSMR	53.365391	-2.73168
Worthing A27 Roadside	WTHG	50.832947	-0.379916
Wrexham	WREX	53.04222	-3.002778
York Fishergate	YK11	53.951889	-1.075861

Table A2: List of Urban Background sites used in analysis.

site	code	latitude	longitude
Aberdeen	ABD	57.15736	-2.094278
Ballymena Ballykeel	BALM	54.861595	-6.250873
Barnsley	BARN	53.580045	-1.475865
Barnsley 12	BAR2	53.55593	-1.485153
Barnsley Gawber	BAR3	53.56292	-1.510436
Belfast Centre	BEL2	54.59965	-5.928833
Belfast East	BEL	54.59653	-5.901667
Belfast South	BEL3	54.59918	-5.912416
Bircotes	BIR	53.422855	-1.054946
Birmingham Acocks Green	AGRN	52.437165	-1.829999
Birmingham Centre	BIRM	52.479724	-1.908078
Birmingham East	BIR2	52.49763	-1.831498
Birmingham Ladywood	BMLD	52.481346	-1.918235
Birmingham Tyburn	BIR1	52.511722	-1.830583
Blackpool	BLAC	53.79046	-3.029283
Blackpool Marton	BLC2	53.80489	-3.007175
Bolton	BOLT	53.57232	-2.439583
Borehamwood Meadow Park	BDMP	51.661229	-0.270671
Bournemouth	BORN	50.73957	-1.826744
Bradford Centre	BRAD	53.79339	-1.748694
Brighton Preston Park	BRT3	50.840836	-0.147572
Bristol Centre	BRIS	51.45718	-2.585622
Bristol East	BRS	51.45365	-2.578496
Bristol St Paul's	BRS8	51.462839	-2.584482
Burton-on-Trent Horninglow	BOTR	52.82105	-1.635718
Canterbury	CANT	51.27399	1.098061
Cardiff Centre	CARD	51.48178	-3.17625
Cardiff East	CAR2	51.48887	-3.163711
Central London	CLL	51.494722	-0.138333
Chesterfield	CHS6	53.230583	-1.433611
Chesterfield Loundsley Green	CHLG	53.244131	-1.454946
Coventry Allesley	COAL	52.411563	-1.560228
Coventry Centre	COV2	52.41345	-1.522133
Crewe Coppenhall	COPP	53.115941	-2.453492
Cwmbran	CWMB	51.6538	-3.006953
Derry	DERY	55.001225	-7.329115
Derry Rosemount	DERR	55.002818	-7.331179
Dewsbury Ashworth Grove	DYAG	53.693104	-1.637111
Dundee Mains Loan	DCC1	56.475434	-2.959861

Eastbourne	EB	50.805778	0.271611
Edinburgh Centre	ED	55.95197	-3.195775
Edinburgh Med. Sch.	ED2	55.94427	-3.191183
Edinburgh St Leonards	ED3	55.945589	-3.182186
Featherstone	FEA	53.670222	-1.352146
Glasgow Centre	GLA3	55.85773	-4.255161
Glasgow City Chambers	GLA	55.860414	-4.245959
Glasgow Townhead	GLKP	55.865782	-4.243631
Hartlepool St Abbs Walk	HSAW	54.683242	-1.203838
Honiton	HONI	50.792287	-3.196702
Hull Centre	HULL	53.74479	-0.338322
Hull Freetown	HUL2	53.74878	-0.341222
Immingham Woodlands Avenue	IMGM	53.619241	-0.213324
Leamington Spa	LEAM	52.28881	-1.533119
Leeds Centre	LEED	53.80378	-1.546472
Leeds Potternewton	LDS	53.82568	-1.535098
Leicester Centre	LEIC	52.631348	-1.133006
Leicester University	LECU	52.619823	-1.127311
Liverpool Centre	LIVR	53.40845	-2.980249
London Bloomsbury	CLL2	51.52229	-0.125889
London Brent	BREN	51.589769	-0.276223
London Bridge Place	BRI	51.49521	-0.141655
London Hackney	HK4	51.55877	-0.056592
London Haringey	HG2	51.58603	-0.126486
London Haringey Priory Park South	HG4	51.584128	-0.125254
London Harrow Stanmore	HR3	51.617333	-0.298777
London Hillingdon	HIL	51.49633	-0.460861
London Honor Oak Park	HP1	51.449674	-0.037418
London Islington	ISL	51.531362	-0.096954
London Lewisham	LW1	51.44541	-0.020139
London N. Kensington	KC1	51.52105	-0.213492
London Southwark	SK1	51.49055	-0.096667
London Teddington	TED	51.42099	-0.339647
London Teddington Bushy Park	TED2	51.425286	-0.345606
London UCL	LON5	51.52378	-0.128958
London Wandsworth	WA2	51.45696	-0.191164
London Westminster	HORS	51.49467	-0.131931
Manchester Piccadilly	MAN3	53.48152	-2.237881
Manchester Town Hall	MAN	53.4785	-2.2448
Newcastle Centre	NEWC	54.97825	-1.610528
Newport	NPT3	51.601203	-2.977281
Northampton	NTON	52.27349	-0.885933
Northampton Kingsthorpe	NTN3	52.271886	-0.879898
Northampton Spring Park	NTN4	52.272257	-0.916605
Norwich Centre	NOR2	52.63203	1.295019
Norwich Lakenfields	NO12	52.614193	1.301976
Nottingham Centre	NOTT	52.95473	-1.146447
Oxford St Ebbes	OX8	51.744806	-1.260278

Peebles	PEEB	55.657472	-3.196527
Plymouth Centre	PLYM	50.37167	-4.142361
Portsmouth	PMTH	50.82881	-1.068583
Preston	PRES	53.76559	-2.680353
Reading	READ	51.45352	-0.95518
Reading New Town	REA1	51.45309	-0.944067
Rotherham Centre	ROTH	53.43186	-1.354444
Rugeley	RUGE	52.753297	-1.93773
Salford Eccles	ECCL	53.48481	-2.334139
Sandwell Oldbury	OLDB	52.50431	-2.017629
Sandwell West Bromwich	WBRO	52.52062	-1.995556
Sheffield Centre	SHE2	53.37772	-1.473306
Sheffield Devonshire Green	SHDG	53.378622	-1.478096
Sheffield Tinsley	SHE	53.41058	-1.396139
Southampton Centre	SOUT	50.90814	-1.395778
Southend-on-Sea	SEND	51.544206	0.678408
Stockport	STOC	53.40994	-2.1582
Stockport Shaw Heath	STK4	53.40306	-2.161111
Stoke-on-Trent Centre	STOK	53.02821	-2.175133
Sunderland	SUND	54.906106	-1.380081
Sunderland Silksworth	SUN2	54.88361	-1.406878
Swansea	SWAN	51.62114	-3.943329
Swindon Walcot	SWHO	51.558061	-1.765678
Telford Hollinswood	TDHD	52.673471	-2.436692
Thurrock	THUR	51.47707	0.317969
Walsall Alumwell	WAL	52.58167	-2.010483
Walsall Willenhall	WAL2	52.60821	-2.033144
Walsall Woodlands	WAL4	52.605621	-2.030523
West Bromwich Kenrick Park	WBKP	52.508337	-1.986008
West London	WL	51.4938	-0.200361
Wigan Centre	WIG5	53.54914	-2.638139
Wigan Leigh	WIG3	53.49422	-2.506899
Wirral Tranmere	TRAN	53.37287	-3.022722
Wolverhampton Centre	WOLV	52.58818	-2.129008
York Bootham	YK10	53.967513	-1.086514

Table A3: List of Urban Background sites used for individual city analysis.

Site	**AURN Code**	**Latitutde**	**Longitude**
Belfast Centre	BEL2	54.59965	-5.928833
Belfast East	BEL	54.59653	-5.901667
Belfast South	BEL3	54.59918	-5.912416
Birmingham Acocks Green	AGRN	52.437165	-1.829999
Birmingham Centre	BIRM	52.479724	-1.908078
Birmingham East	BIR2	52.49763	-1.831498
Birmingham Ladywood	BMLD	52.481346	-1.918235
Birmingham Tyburn	BIR1	52.511722	-1.830583

Bristol Centre	BRIS	51.45718	-2.585622
Bristol East	BRS	51.45365	-2.578496
Bristol St Paul's	BRS8	51.462839	-2.584482
Cardiff Centre	CARD	51.48178	-3.17625
Cardiff East	CAR2	51.48887	-3.163711
Glasgow Centre	GLA3	55.85773	-4.255161
Glasgow City Chambers	GLA	55.860414	-4.245959
Glasgow Townhead	GLKP	55.865782	-4.243631
Leeds Centre	LEED	53.80378	-1.546472
Leeds Potternewton	LDS	53.82568	-1.535098
London Bloomsbury	CLL2	51.52229	-0.125889
London Brent	BREN	51.589769	-0.276223
London Bridge Place	BRI	51.49521	-0.141655
London Hackney	HK4	51.55877	-0.056592
London Haringey	HG2	51.58603	-0.126486
London Haringey Priory Park South	HG4	51.584128	-0.125254
London Harrow Stanmore	HR3	51.617333	-0.298777
London Hillingdon	HIL	51.49633	-0.460861
London Honor Oak Park	HP1	51.449674	-0.037418
London Islington	ISL	51.531362	-0.096954
London Lewisham	LW1	51.44541	-0.020139
London N. Kensington	KC1	51.52105	-0.213492
London Southwark	SK1	51.49055	-0.096667
London Teddington	TED	51.42099	-0.339647
London Teddington Bushy Park	TED2	51.425286	-0.345606
London UCL	LON5	51.52378	-0.128958
London Wandsworth	WA2	51.45696	-0.191164
London Westminster	HORS	51.49467	-0.131931
Manchester Piccadilly	MAN3	53.48152	-2.237881
Manchester Town Hall	MAN	53.4785	-2.2448
Newcastle Centre	NEWC	54.97825	-1.610528

Table A4: Site names used in London exceedances calculations

Site	**AURN Code**	**Latitutde**	**Longitude**
Brent - Ikea	BT4	51.55248	-0.25809
Lambeth - Brixton Road	LB4	51.46411	-0.11458
Westminster - Marylebone Road FDMS	MY7	51.52254	-0.15459
Brent - Neasden Lane	BT5	51.55266	-0.24877
Camden - Bloomsbury	BL0	51.52229	-0.12585
Bexley - Slade Green FDMS	BX9	51.46598	0.184877
Sutton - Worcester Park	ST6	51.37792	-0.24041
City of London - Farringdon Street	CT2	51.51453	-0.10452
Wandsworth - Putney High Street	WA7	51.46343	-0.21587
Ealing - Horn Lane	EA8	51.51895	-0.26562
Camden - Swiss Cottage	CD1	51.54422	-0.17528
Camden - Euston Road	CD9	51.52798	-0.12877
Wandsworth - Putney High Street Fa-cade	WA8	51.46372	-0.21589

Table A5: Site names used in AURN outside London exceedances calculations

Site	AURN Code	Latitutde	Longitude
Chatham Roadside	CHAT	51.37426	0.54797
Chepstow A48	CHP	51.63809	-2.67873
Christchurch Barrack Road	CHBR	50.73545	-1.78089
Edinburgh Nicolson Street	EDNS	55.94476	-3.18399
Glasgow Kerbside	GLA4	55.85917	-4.25889
Hafod-yr-ynys Roadside	CAE6	51.68058	-3.13351
Leamington Spa Rugby Road	LEAR	52.29488	-1.54291
Leeds Headingley Kerbside	LED6	53.81997	-1.57636
Oxford Centre Roadside	OX	51.75175	-1.25746
Sandy Roadside	SDY	52.13242	-0.30031
Sheffield Barnsley Road	SHBR	53.40495	-1.45582
Stanford-le-Hope Roadside	HOPE	51.51817	0.439548
Stoke-on-Trent A50 Roadside	STKR	52.98044	-2.1119
Swansea Roadside	SWA1	51.63270	-3.94737
Wrexham	WREX	53.04222	-3.00278

DEFRA Key questions- Dr. Adobi Okam

What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?

The indoors is basically an enclosed space and it includes: Homes, offices, restaurants, classrooms, hospitals, in vehicles e.t.c.

People spend around 85 -93 % of their time indoors (Lazaridis, 2011, Okam, 2017, Höppe and Martinac, 1998). This percentage will increase at this time of lockdown especially for the vulnerable and those not only on lockdown but are also shielding. Based on the amount of time people spend indoors regardless of lockdown or not, significant proportion of personal exposure to pollutants occur in the indoors. However, due to the lockdown most people are spending even more than that 93 % in their homes and doing more activities than usual. All the time they spend at work, children at school are all eliminated and instead that chunk of time in addition to the evening and the entire night is now spent at home. This means that for most people, the indoors that they are spending their time in is the home.

Like previously mentioned more activities are going on indoors as people are using these activities to occupy themselves and fill up the time, plus they are not allowed to go out unnecessarily. For instance cooking generates particles diameter size in the range of 22 – 940 nm (He et al., 2004) this depends on cooking style, cooking fuel/energy, food condiments and ventilation system. A study showed that particle concentration were significantly higher during cooking activities, and could remain that high for up to 14 hours (Morawska et al., 2003). Other activities such as: house chores, resuspension of particles as result of walking, exercise, sweeping, cleaning products, aerosol sprays, candle burning, wood fire e.t.c. all generate particles and other pollutants that can build-up. In addition to this we also have pollutants such bio-aerosols e.g. mould and pollution as a result of penetration from the outdoors. Exposure to these pollutants especially as a result of build-up can result in both acute and chronic health problems.

The major problem for the indoors is the build-up of pollutants as a result of these activities and poor ventilation. This will then result in Poor Indoor Air Quality, which could lead to acute symptoms such as headache, dizziness, watery eyes, sneezing, shortness of breath, throat irritation e.t.c. However, proper ventilation can help reduce pollutant build-up.

How might public exposure to air pollution have changed as a consequence of recent restrictions on movement?

Normally the public is exposed to air pollution in the indoors and the outdoors. The outdoors may have high levels of pollutant as a result of vehicular emissions. In fact as a result of build-up, lack or type of ventilation (mechanical or natural), type of indoor activity and duration of the activity and pollutant produced the indoors can be as catastrophic as the outdoors, if not even more. A study suggests that build-up of pollutants indoors can be up 3 to 5 times more than the outdoors.

Averagely people spend about 7 hours at work which is mostly an indoor, about 14 hours at home and the rest of the 3 hours between commuting and the outdoors. According to research people spend 85 – 93 % of their time indoors but with this lockdown, this has drastically changed for most people. Most people are outdoors only for their one hour daily exercise or shopping for supplies. In the case of those shielding in addition to the lockdown, they don't even go outdoors. Therefore such people spend 100 % of their time indoors while the others may be spending ≤ 1% outdoors.

The game has changed. Exposure for the public will be the indoor environment.

Are there any insights that can be gained from aerosol science on possible viral transmission mechanisms?

During tidal breathing, it is believed that the airway surface (epithelial lining) fluid is aerosolised. This is as a result of the turbulent airflow i.e. the reopening of closed bronchioles and alveoli. These aerosolised airway fluid are droplets that contain different compounds (about 3500) that will reflect composition of the brochoaveolar lavage fluid (BALF). Formation of these aerosolised droplets depends on the velocity of flow of exhaled air and surface tension, for example during talking, breathing, coughing and sneezing.

Using collection of Exhaled Breath Condensate (EBC) for example; EBC is a non-invasive technique used for analysing inflammatory mediators (biomarkers) present in the airway lining especially the lower airways secretion (Horvath et al., 2005). This technique involves the cooling down of breath in order to get respiratory air in liquid form. Collection of EBC is done through mouth breathing. This method although has some standardisation issues is still widely used.

It has been used to analyse biomarkers as a result of exposure to air pollution. Biomarkers such as nitrate, nitrite, cytokines, isoprostane, airway pH and other metabolites (Okam, 2017, Nel, 2005, Horvath et al., 2005, Montuschi, 2007). It has also been used for virus analysis in various researches (Houspie et al., 2011, Zakharkina et al., 2011, Yan et al., 2018).

Therefore the ability to detect virus in the EBC, confirms that the virus in an infected person aerosolises during turbulent flow. Therefore when aerosolised, can then be travel to different distances during exhalation. Distance the virus will travel then depends if the infected individual is breathing normally, fast breathing, coughing and sneezing.

Then how long the exhaled viral droplet stays viable in the air, how long it remains suspended in the air, when and if another person inhales and what happens when inhaled by another person, it depends on different physical and chemical factors.

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Response to AQEG Request for Rapid Evidence on COVID-19 & UK Air Quality

30 April 2020

Experts in air quality management & assessment

Ben Marner, Duncan Laxen, Ricky Gellatly, and Tomas Liska

1 Introduction

1.1 This note summarises some of the analysis carried out by Air Quality Consultants ltd. into the effects of the COVID-19 social and travel restrictions on UK air quality. It has been prepared in response to the Air Quality Expert Group call for evidence on: *"estimates for how emissions and ambient concentrations of NOx, NO_2, PM, O_3, VOC, NH_3 etc may have changed since the COVID outbreak"*.

1.2 The analysis has used Boosted Regression Trees (BRT)[1] to construct models of the dependence on meteorology of NOx, NO_2, and O_3 concentrations at 205 UK monitoring sites[2]. The models have been built using openair[3] from >5 years of hourly-mean measurements[4] covering the period 1st Jan 2015 to 9th Apr 2020. These models have then been used to nominally remove the effects of meteorological and temporal factors from the measurements[5]. The residual variation in concentrations at each site is that which cannot be explained by the BRT models solely in terms of predictable responses to weather and routine cyclical patterns. Recent measurements are unratified and so still subject to change; however, confidence is added to the overall findings of this note by the inclusion of a relatively large number of monitoring sites.

1.3 This note focuses on observed changes in ambient concentrations. Lockdown is taken to run from 23rd March, although more limited restrictions on travel started from around 11th March. It has not, at this time, been possible to link the changes in concentrations with robust activity data or emissions estimates.

2 NOx and NO_2

2.1 Figure 1 shows an example of the observed and BRT-adjusted results; focusing on daily mean NOx at four sites. While the raw measured concentrations (dashed lines in Figure 1) vary considerably over time, the models are able to account for almost all of this variability up until mid-march, resulting in little apparent variation in the BRT-adjusted concentrations (bold lines in Figure 1). The step change seen just before the 'lockdown' at these four sites is typical of that at many of the road-influenced sites. The precise timing of the main 'step' varies between sites, but is often followed by a second, smaller, step immediately after the lockdown. Results for all 205 sites are shown in a more concise format in Appendix 1, with time-series averaged across all sites, grouped by site type, in Appendix 2.

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1 Carslaw, DC & Taylor, PJ 2009, 'Analysis of air pollution data at a mixed source location using boosted regression trees', *Atmos Env* 43, no. 22-23, pp. 3563-3570.

2 All sites on the UK Automatic Urban and Rural Network (AURN), Scottish Air Quality Network (SAQN), Welsh Air Quality Network (WAQN) and Air Quality England (AQE) network, which achieved a data capture rate of at least 80% between 1st Jan 2015 and 29th Feb 2020 and at least 90% between 1st Mar and 9th Apr 2020.

3 Carslaw, D.C. and Ropkins, K. (2012) 'openair - An R package for air quality data analysis', *Environmental Modelling & Software,* vol. 27-28, pp. 52-61.

4 Pairing each air quality monitor with the nearest suitable meteorological monitoring site. The same data capture thresholds were applied to meteorological data as to air quality data.

5 The parameters which have been controlled for are: wind speed, wind direction, air temperature, relative humidity, hour of day, day of week, and week of year.

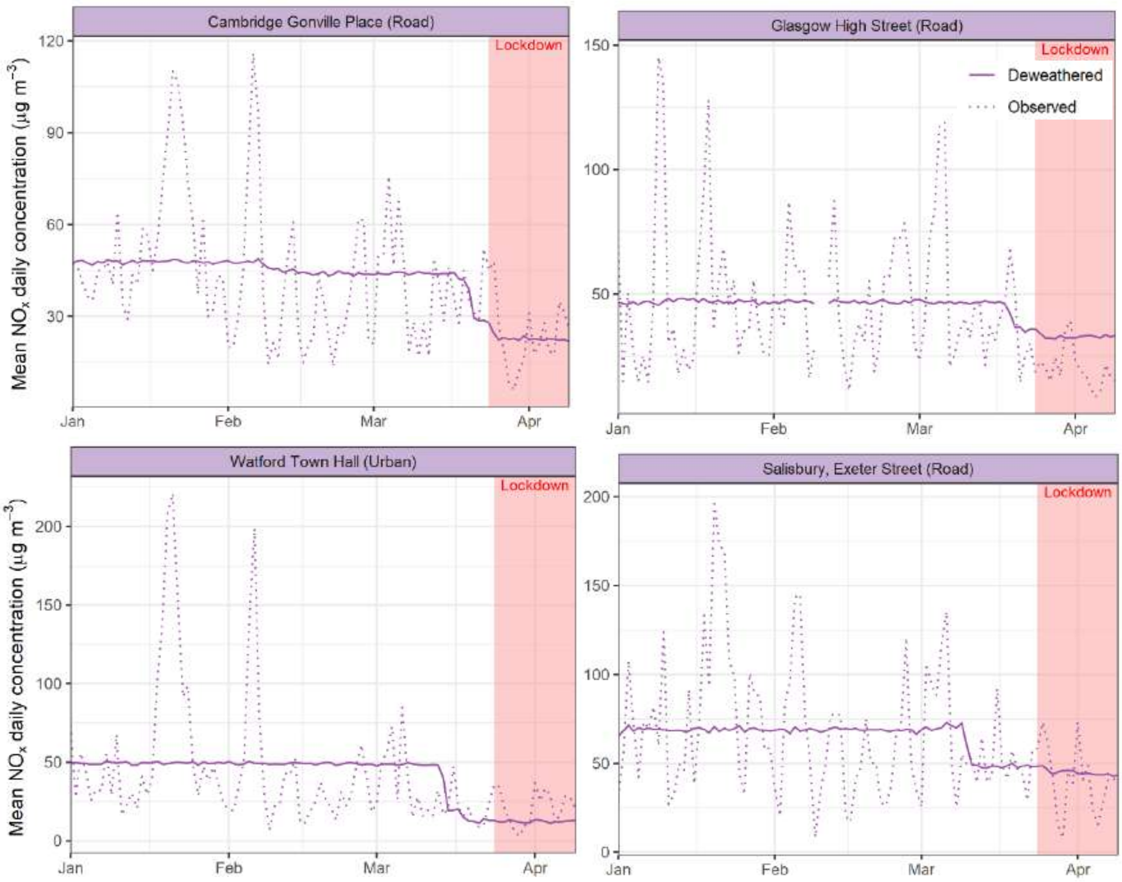


Figure 1: Measured and BRT-adjusted Daily-mean NOx at Four Sites 1st Jan to 9th April 2020

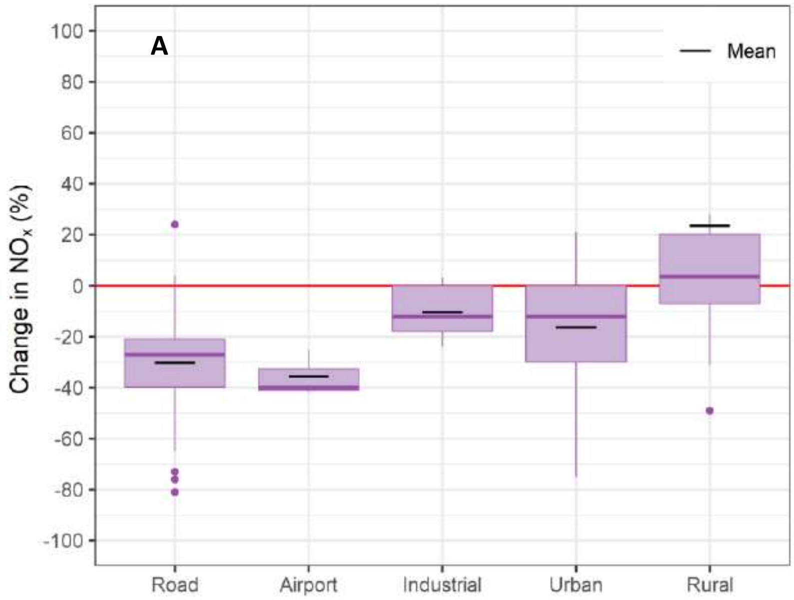
2.2 The relative changes in BRT-adjusted NOx concentrations, comparing a period before the travel disruptions (1st Jan to 14th Mar 2020) with the period between the lockdown and Easter (24th Mar to 9th Apr 2020) are summarised in Figure 2, and shown for each site in Appendix 3. More than half of the roadside sites (68 out of 122) recorded BRT-adjusted reductions in NOx of between 20 and 40%. The average reduction in NOx across 122 roadside sites was 30%, although there was considerable variation from site to site (see Appendix 3). Appendix 4 shows the geographical distribution of these changes, revealing no obvious spatial patterns.

2.3 NOx concentrations at the airport and industrial sites shown in Figure 2 are thought to be heavily influenced by road traffic and no attempt has been made to disentangle different drivers, although the significant reduction at the airport sites is to be expected given the known reductions in flight movements[6]. BRT-adjusted NOx and NO_2 concentrations at many rural sites have been higher since the lockdown; with these sites spread across much of the UK[7]. It is considered unlikely that this regional episode was caused by the COVID-19 restrictions, but it will have offset the improvements seen at roadside sites (noting that large relative changes at rural sites are typically small in absolute

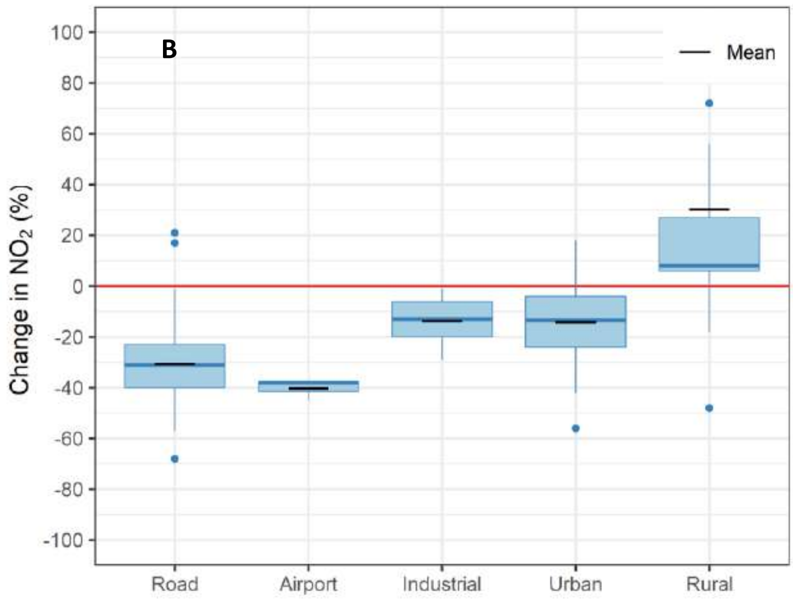
6 It should be noted that all three airport sites are associated with a single airport (Heathrow).

7 Building meteorology-based BRT models to describe concentrations at rural sites is considered less robust than at roadside sites, owing to the importance of regional patterns and the smaller concentration range (which is either rounded or truncated in the data archives) against which to train the models. This is borne out by tests of model fit, which are not presented here. The regional episode at rural sites nevertheless appears to be genuine.

terms). Most urban background sites experienced improvements; albeit smaller than those seen at roadside sites. Some urban background sites saw increases in both NOx and NO_2, which may reflect the influence of the rural episode.



No. Sites	122	3	9	57	14
Mean	-30%	-36%	-10%	-16%	23%



No. Sites	123	3	9	56	13
Mean	-31%	-40%	-14%	-14%	15%

Figure 2: Relative Change in BRT-adjusted [A] NOx and [B] NO_2 Concentrations at 205 UK Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020) (Boxes show Q1 and Q3 ranges, vertical lines extend to points no further than 1.5 times interquartile range from Q1 and Q3, dots show remaining points. Outliers at 195% (NOx) and 242% (NO_2) are not shown)[8].

[8] Mean NO_2 measured at Eskdalemuir prior to March is currently reported as negative. This site has been removed from Plot B.

2.4 While the period-average reductions of BRT-adjusted NOx and NO_2 at roadside sites are similar, the detailed patterns show increases in NO_2 as the lockdown period proceeds, while the NOx concentrations remained steady (Appendix 2). This is likely to be a response to the higher photochemical activity during this period affecting O_3 concentrations and to some extent the higher rural NO_2, which may be due to an influx of air from continental Europe (see Section 3). O_3 has not, at this time, been included as a BRT model parameter in the NO_2 analysis.

2.5 Figure 3 shows the relationship between the reduction in BRT-adjusted NOx and the total, pre-restrictions, BRT-adjusted concentration measured at each road site. There is general pattern of larger relative improvements at the more polluted sites, which is likely to reflect the greater relative importance of road traffic at these sites when compared with other sources. A similar relationship exists for NO_2 but is not shown here.

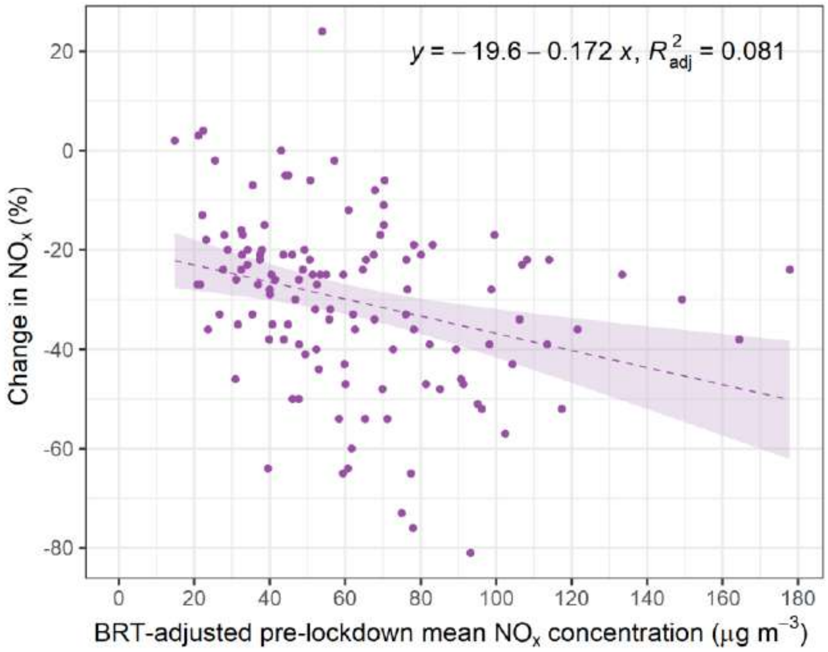
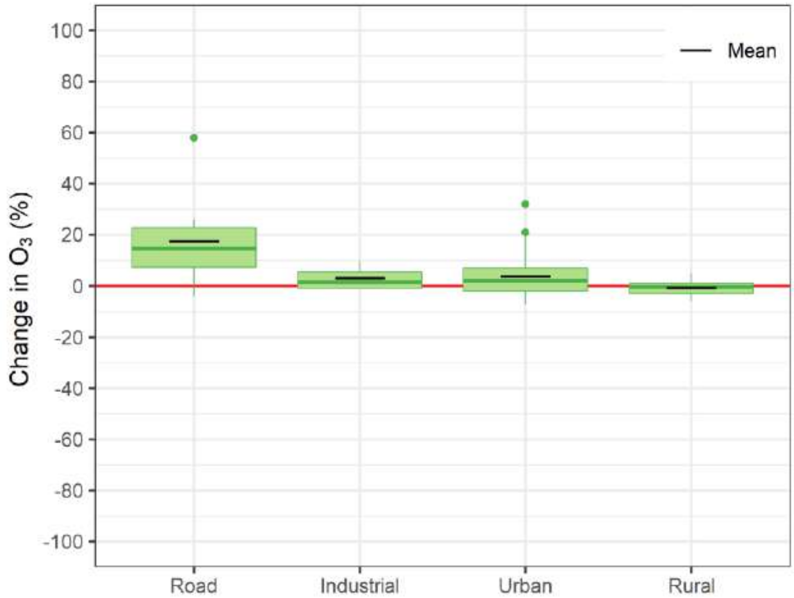


Figure 3: Relative Change in BRT-adjusted NOx at 122 Road sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020) vs BRT-adjusted Mean NOx for period 1st Jan to 14th Mar 2020. Shading shows 95% confidence interval.

3 Ozone

3.1 Appendix 1 summarises the measured and BRT-adjusted daily-mean O_3 concentrations at 74 UK sites. The relative changes, grouped by site type, are summarised in Figure 4 and shown in detail in Appendix 2. Higher daily-mean O_3 concentrations have been recorded at most rural sites since late March, but the BRT models appear able to accurately predict these events; meaning that BRT-adjusted O_3 concentrations during 2020 are relatively constant (Figure 5). At road sites, lower NOx emissions appear to have caused locally-elevated O_3 concentrations in late March, although there is some evidence that O_3 concentrations at the roadside have reverted toward their pre-COVID-19 mean levels (Figure 5) in early April. The current analysis ends on 9th April (immediately before Easter) and so the apparent downward trend for roadside O_3 in April (Figure 5) may be misleading.



No. Sites	10	4	40	20
Mean	17%	3%	4%	-1%

Figure 4: Relative Change in BRT-adjusted O_3 Concentrations at 74 UK Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020) (Boxes show Q1 and Q3 ranges, vertical lines extend to points no further than 1.5 times interquartile range from Q1 and Q3, dots show remaining points)

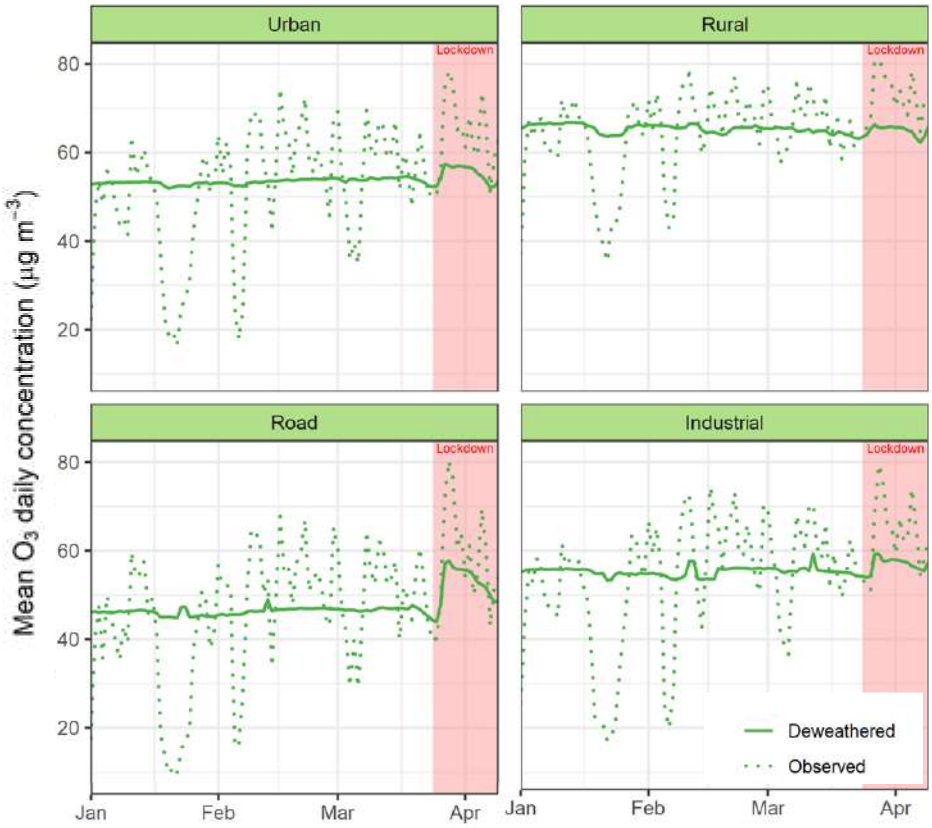


Figure 5: Average Daily-mean O_3 at Different Site Types (Average of site-specific daily-mean values)

3.2 Figure 6 shows the relationship between the reduction in BRT-adjusted O_3 and the total, pre-restriction BRT-adjusted concentration at each road site. It suggests that the largest increases were at the sites where O_3 is typically most effectively titrated.

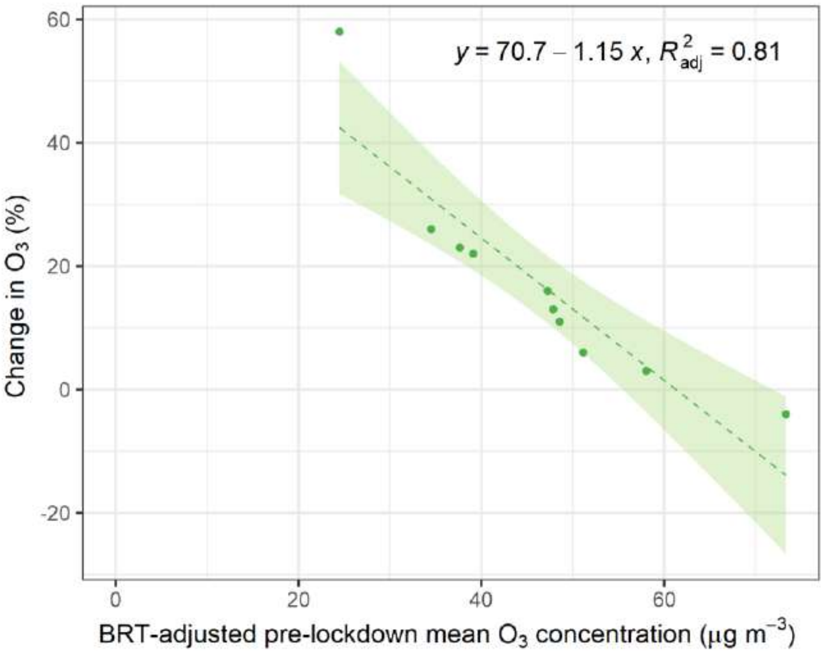


Figure 6: Relative Change in BRT-adjusted O_3 at 10 Road sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020) vs BRT-adjusted Mean O_3 for period 1st Jan to 14th Mar 2020. Shading shows 95% confidence interval.

Appendix 1 Relative Changes in NOx, NO_2, and O_3 at 205 Sites

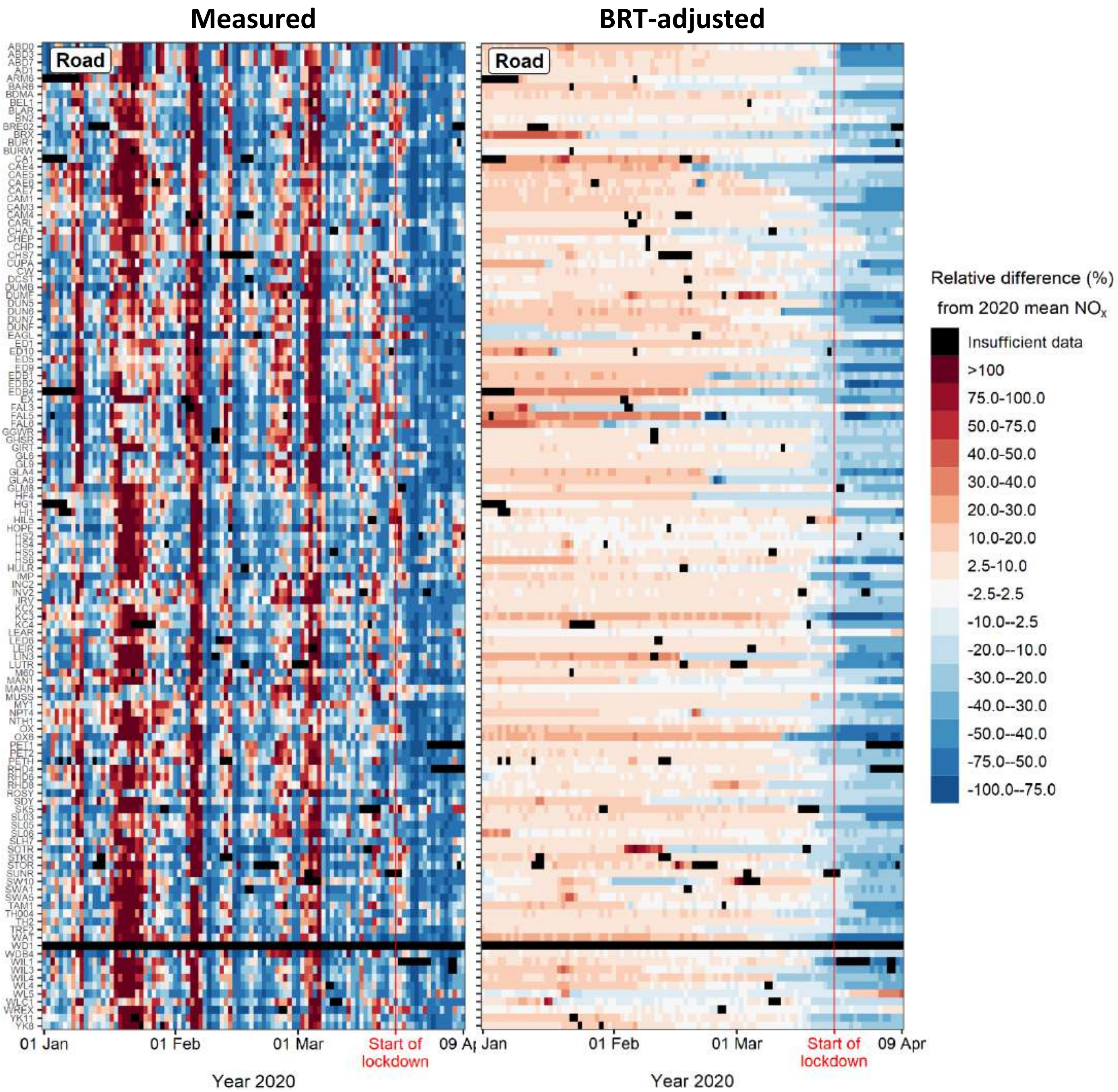


Figure A1.1: Relative Change in Raw Measured and BRT-adjusted Daily-mean NOx Concentrations at UK Roadside Sites – 1st Jan to 9th April 2020. Each row of pixels represents a single site, with the site code given on the y-axis.

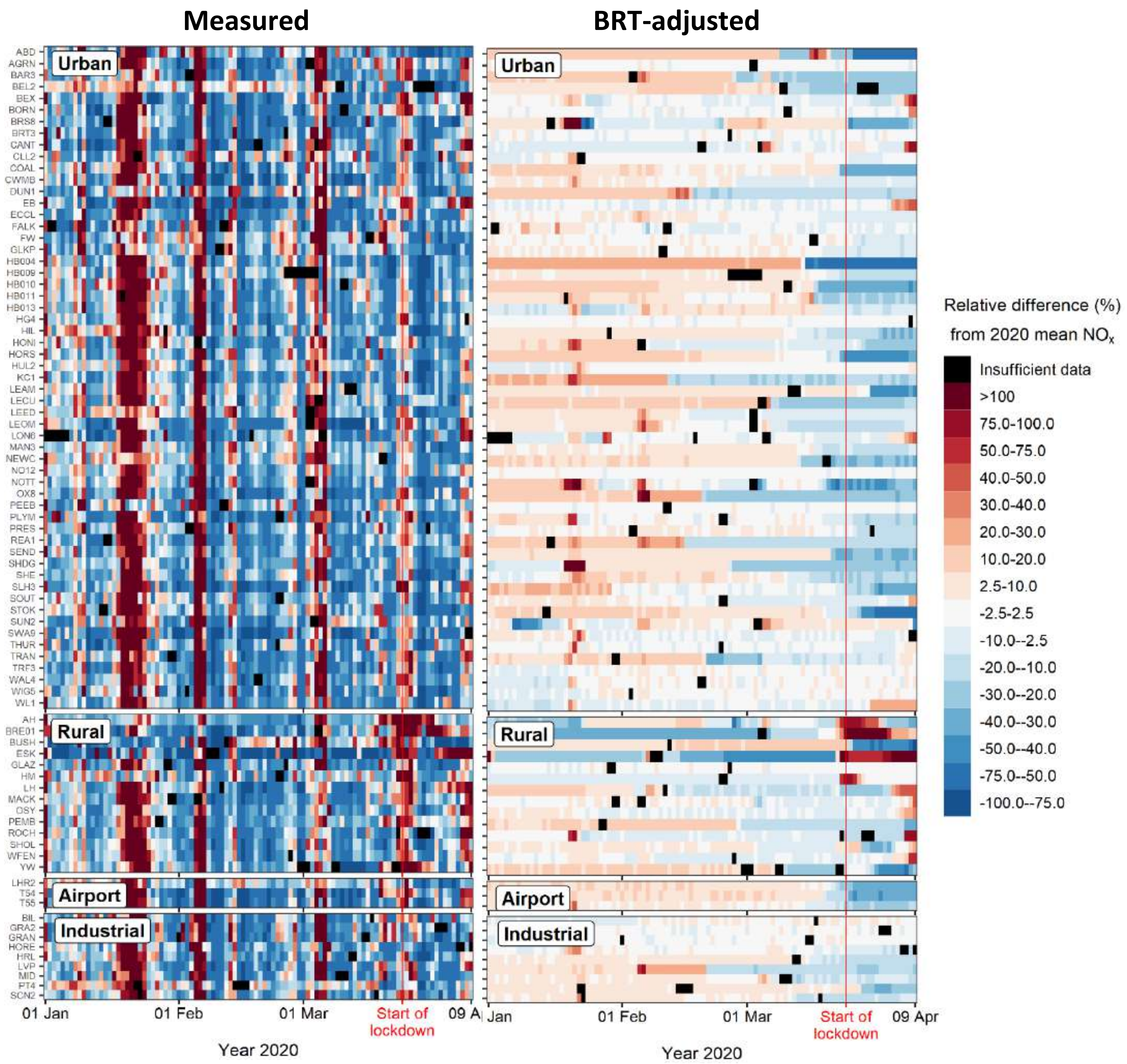


Figure A1.2: Relative Change in Raw Measured and BRT-adjusted Daily-mean NOx Concentrations at UK Non-roadside Sites – 1st Jan to 9th April 2020. Each row of pixels represents a single site, with the site code given on the y-axis.

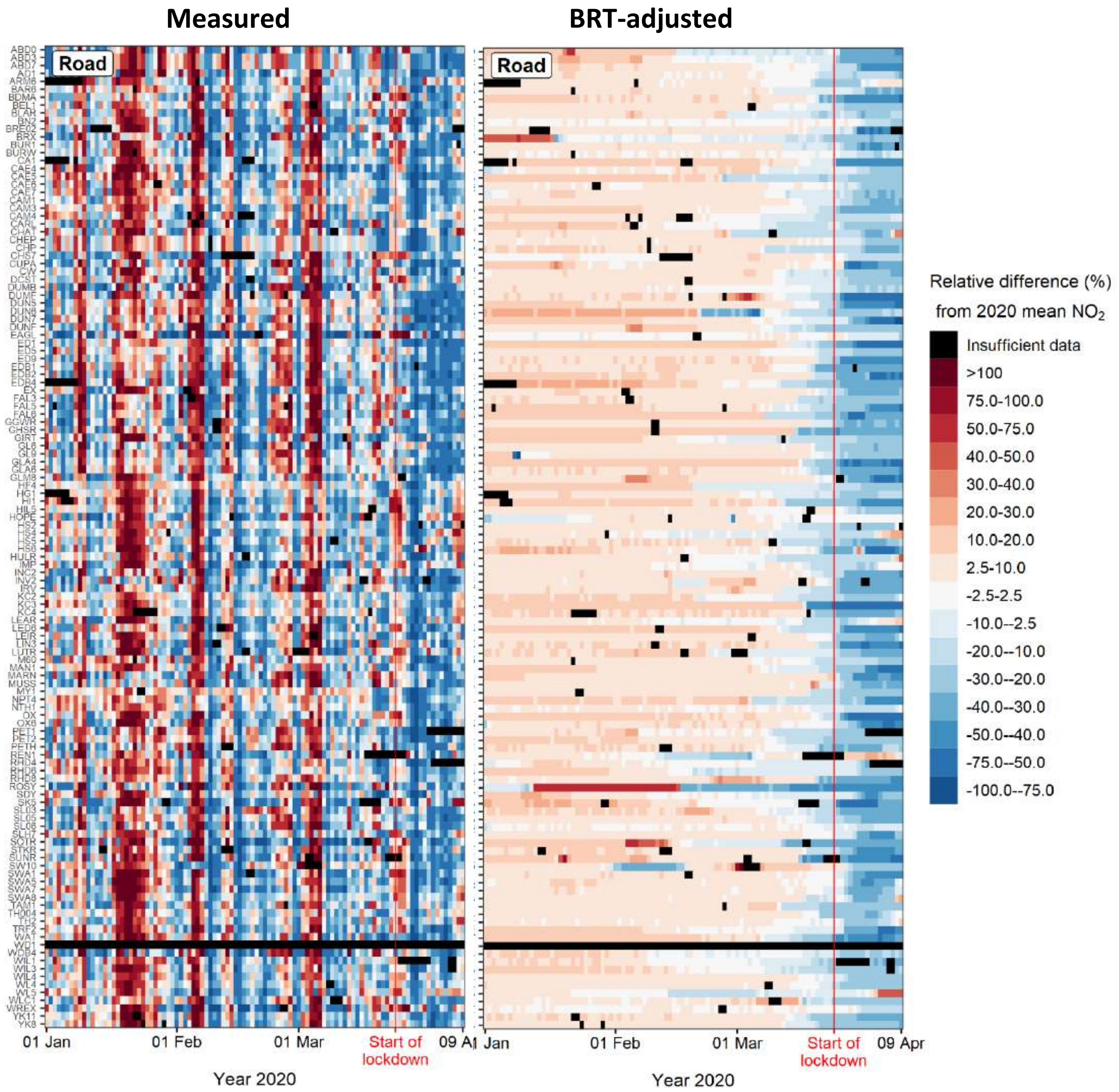


Figure A1.3: Relative Change in Raw Measured and BRT-adjusted Daily-mean NO_2 Concentrations at UK Roadside Sites – 1st Jan to 9th April 2020. Each row of pixels represents a single site, with the site code given on the y-axis.

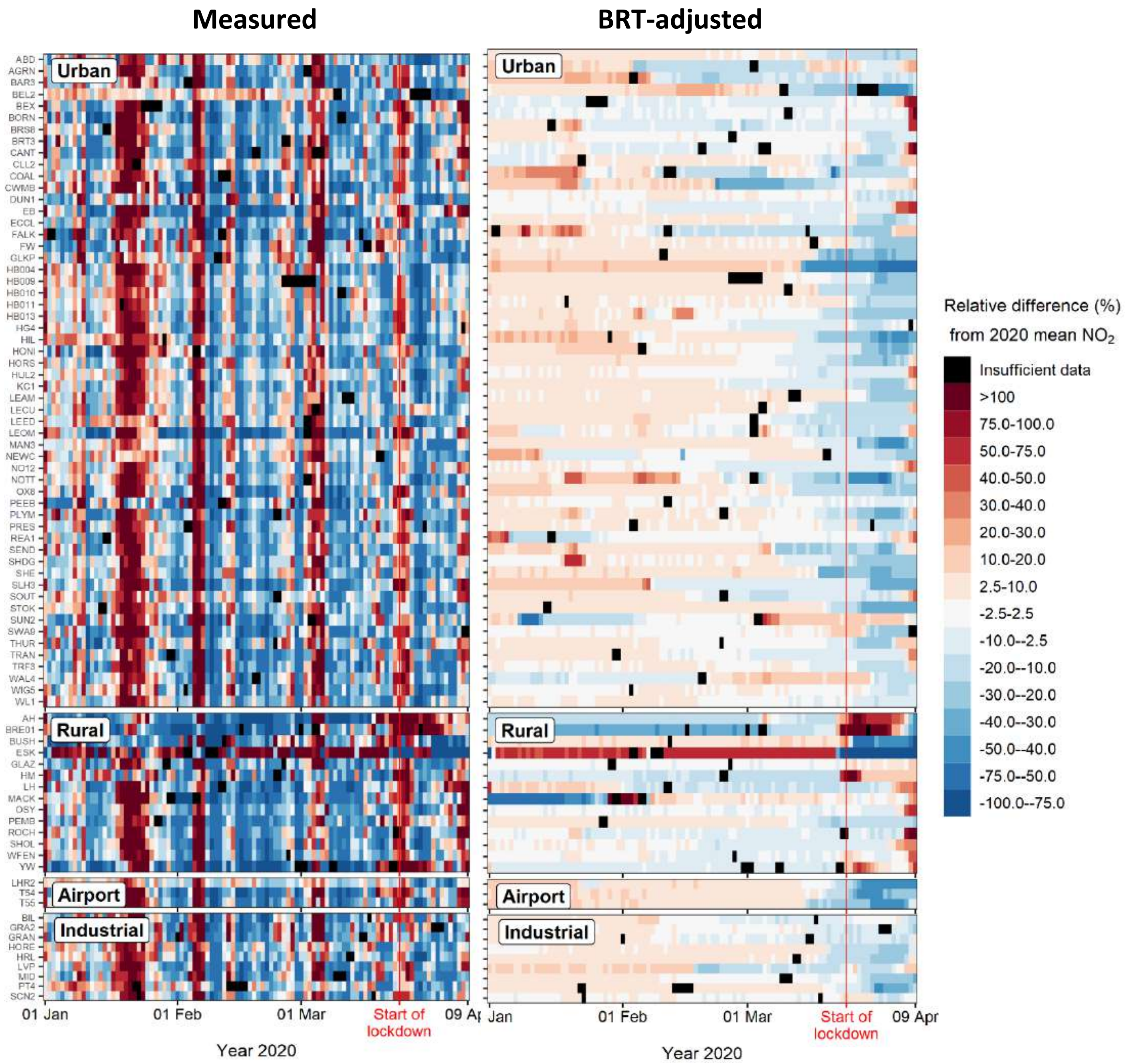


Figure A1.4: Relative Change in Raw Measured and BRT-adjusted Daily-mean NO_2 Concentrations at UK Non-roadside Sites – 1st Jan to 9th April 2020. Each row of pixels represents a single site, with the site code given on the y-axis.

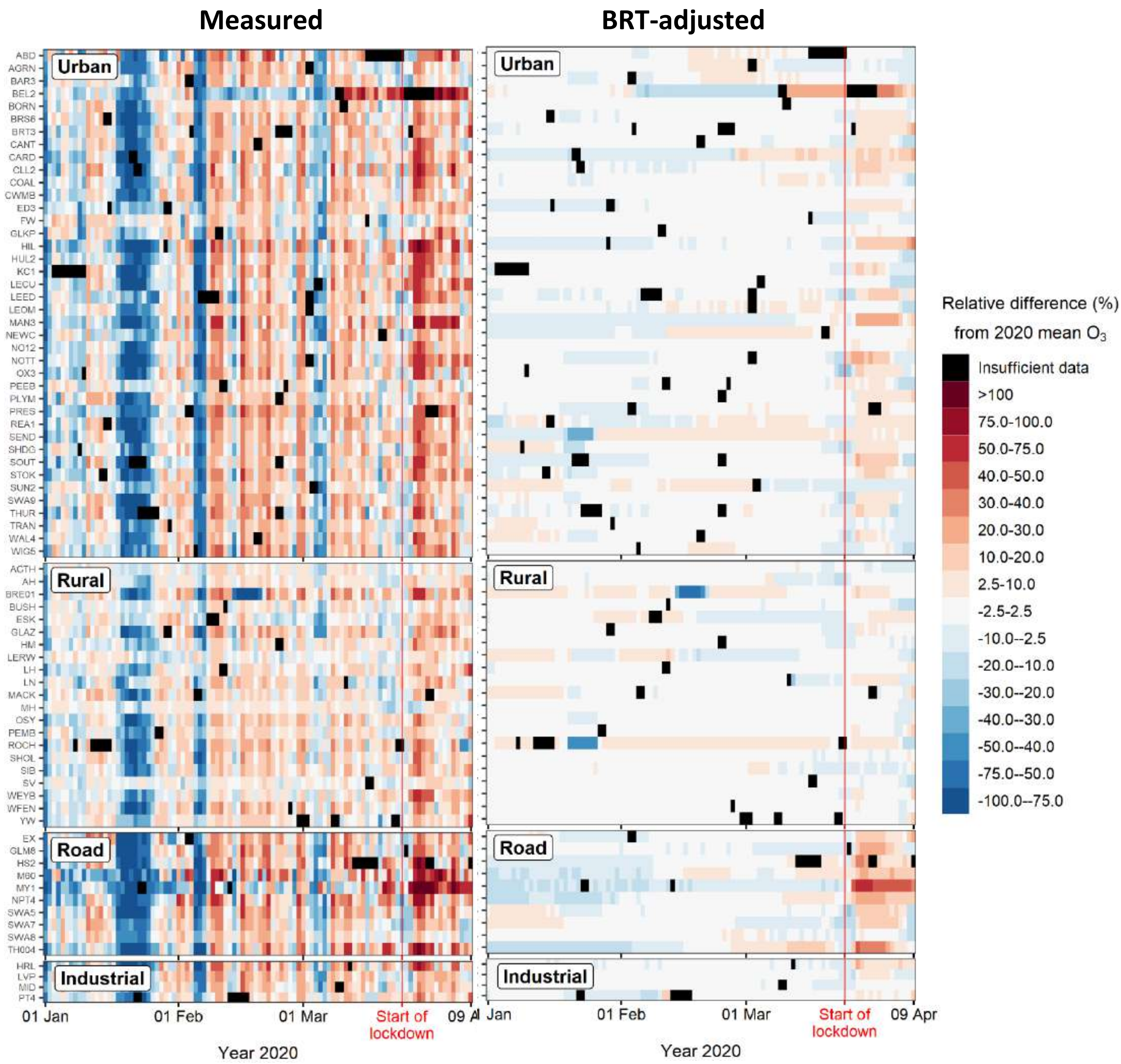


Figure A1.5: Relative Change in Raw Measured and BRT-adjusted Daily-mean O_3 All UK Sites – 1st Jan to 9th April 2020. Each row of pixels represents a single site, with the site code given on the y-axis.

Appendix 2 Aggregated Time-series for NOx and NO_2

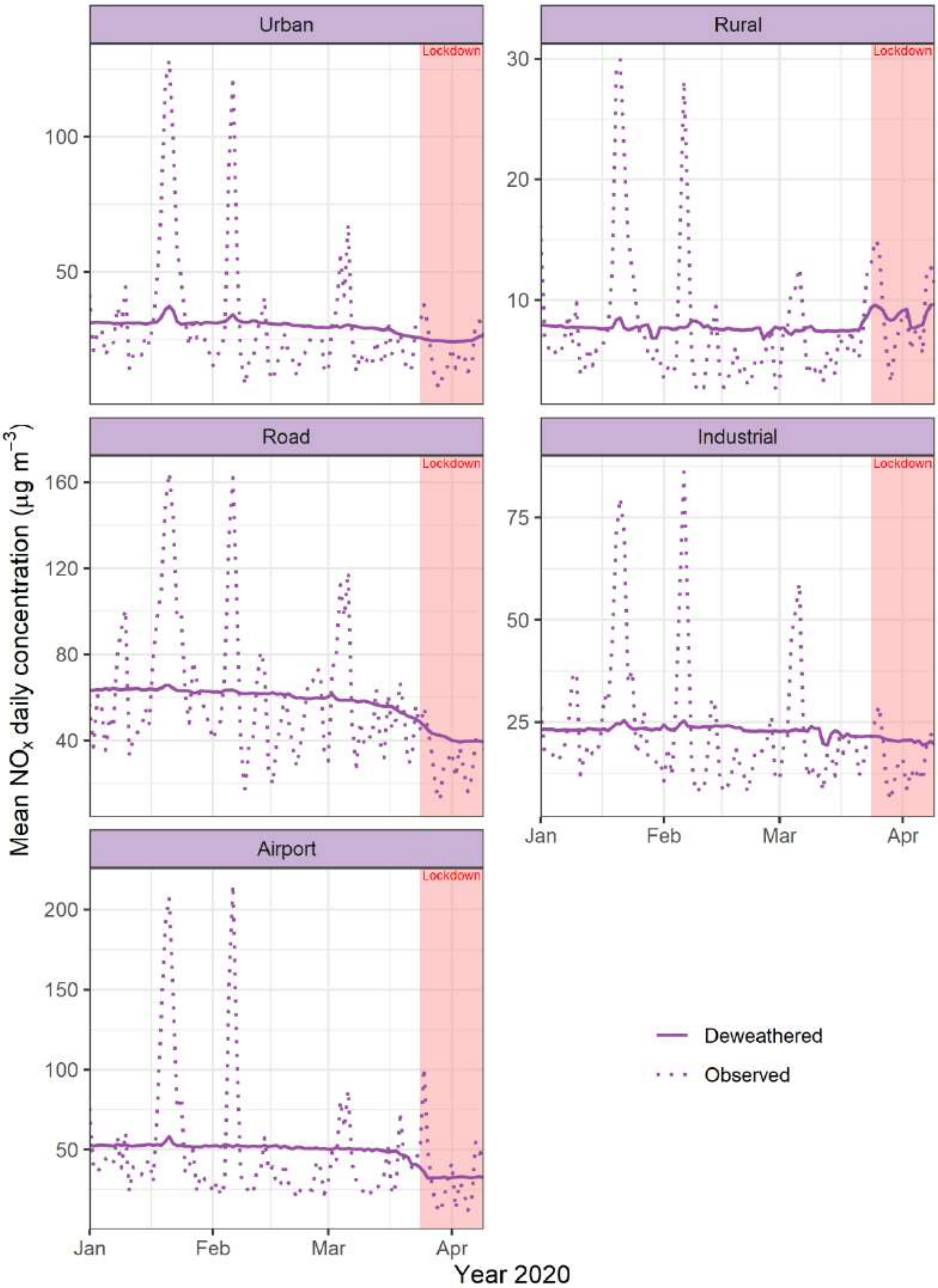


Figure A2.1: Average Daily-mean NOx at Different Site Types (Average of site-specific daily-mean values)

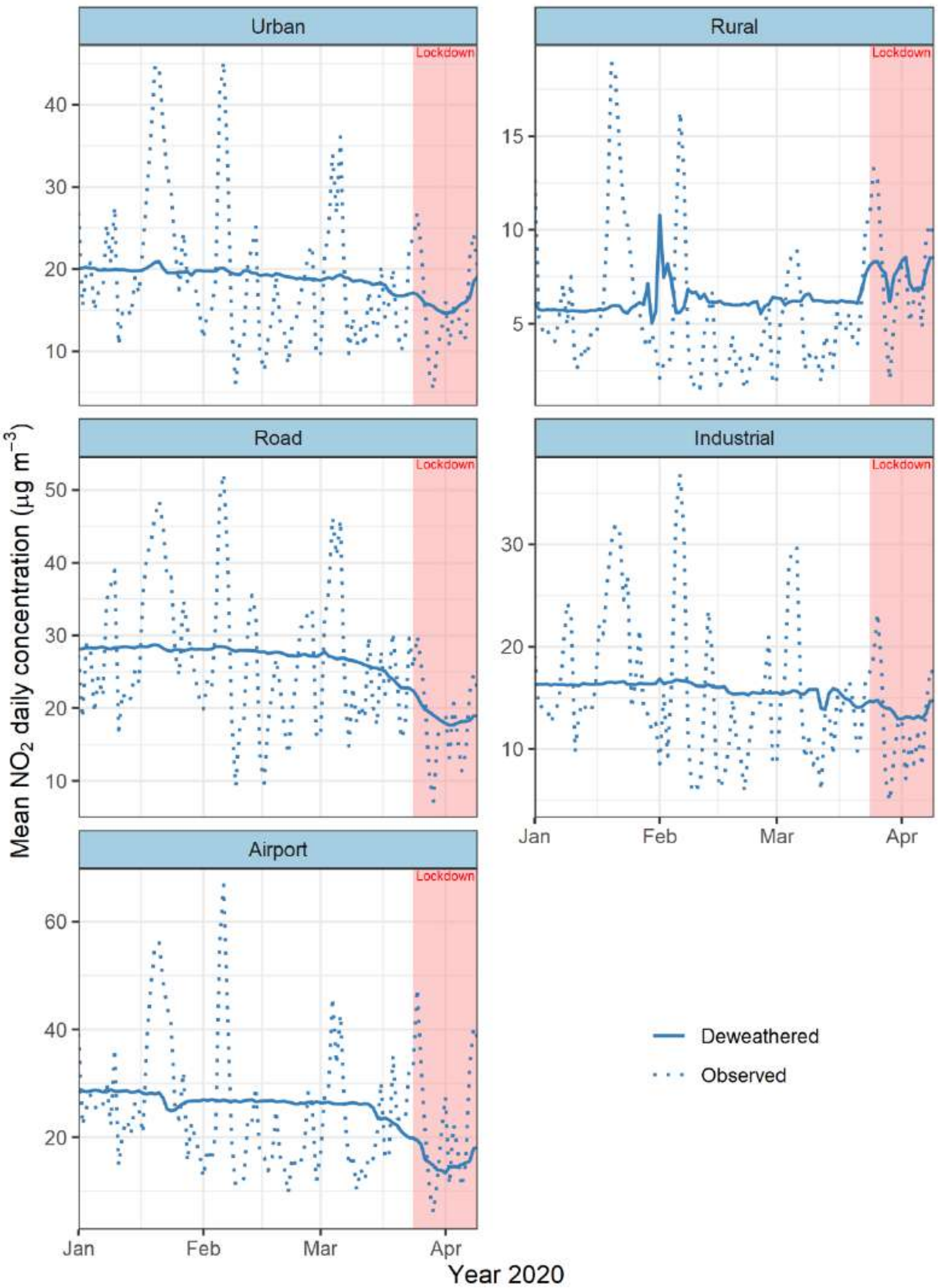


Figure A2.2: Average Daily-mean NO_2 at Different Site Types (Average of site-specific daily-mean values)

Appendix 3 Relative Changes at Each Site

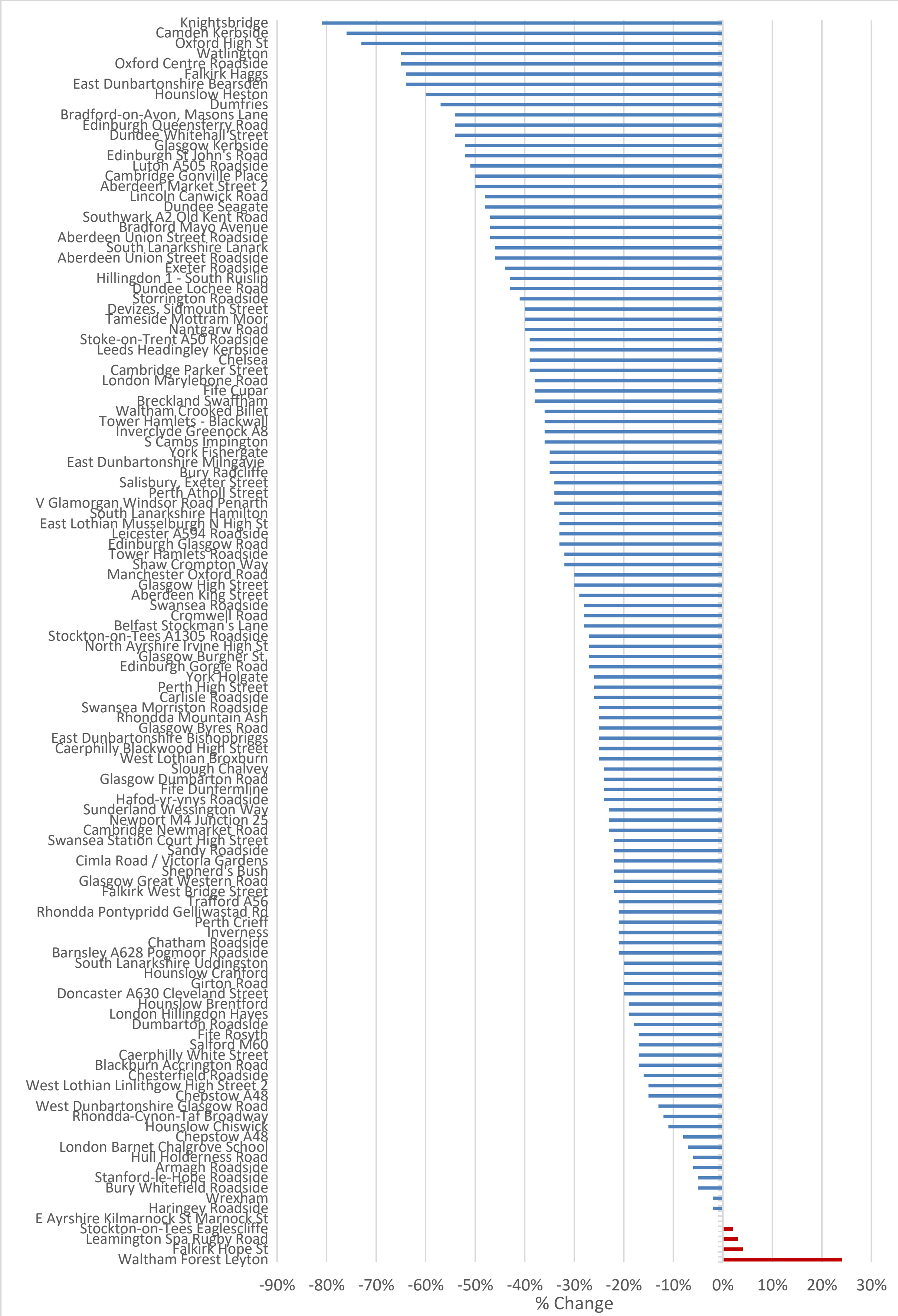


Figure A3.1: Relative Change in BRT-adjusted NOx at 122 Road Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020)

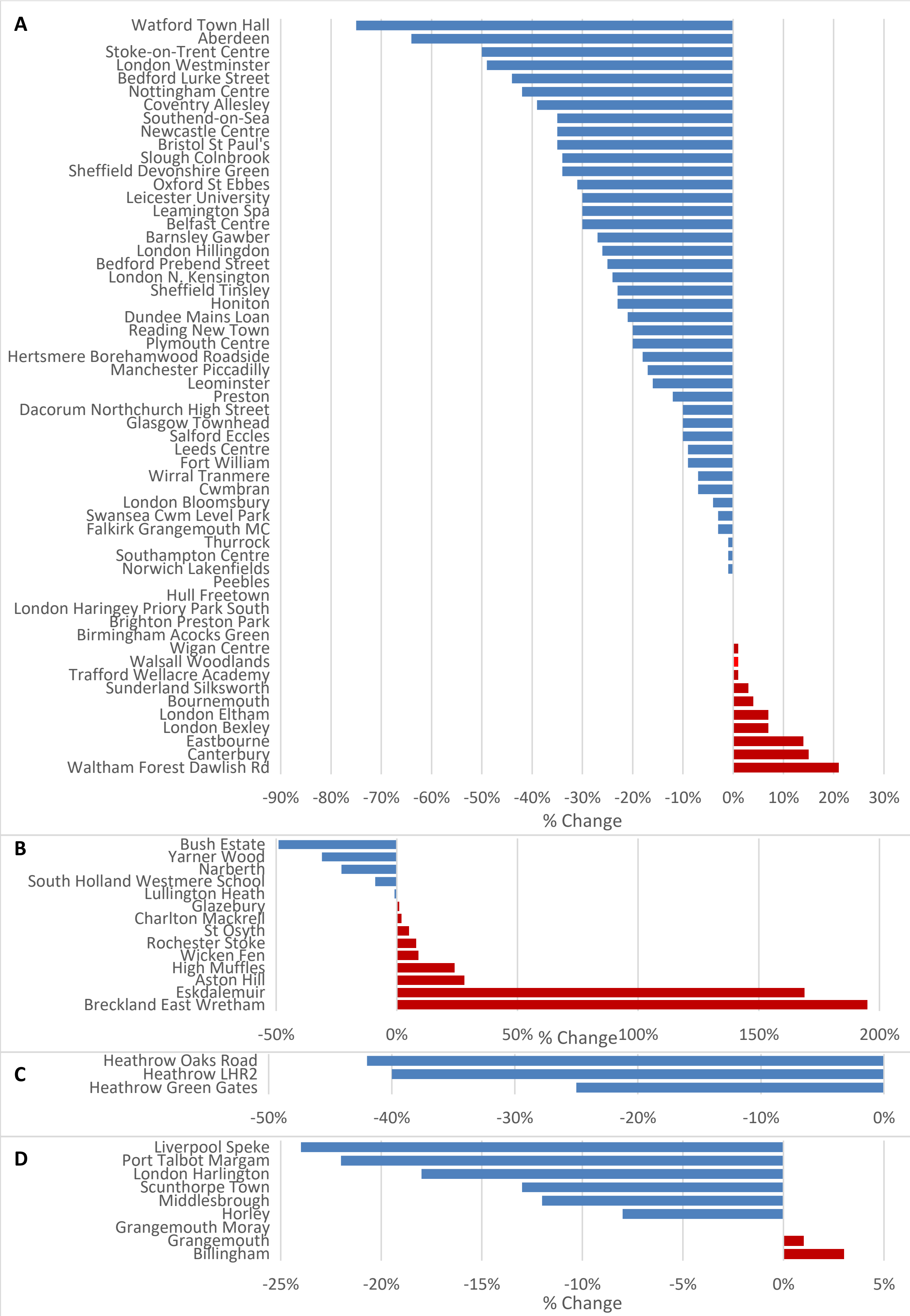


Figure A3.2: Relative Change in BRT-adjusted NOx at [A] 57 Urban Sites and [B] 14 Rural Sites, [C] 3 Airport Sites, and [D] 9 Industrial Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020). Shown using Different Horizontal Scales

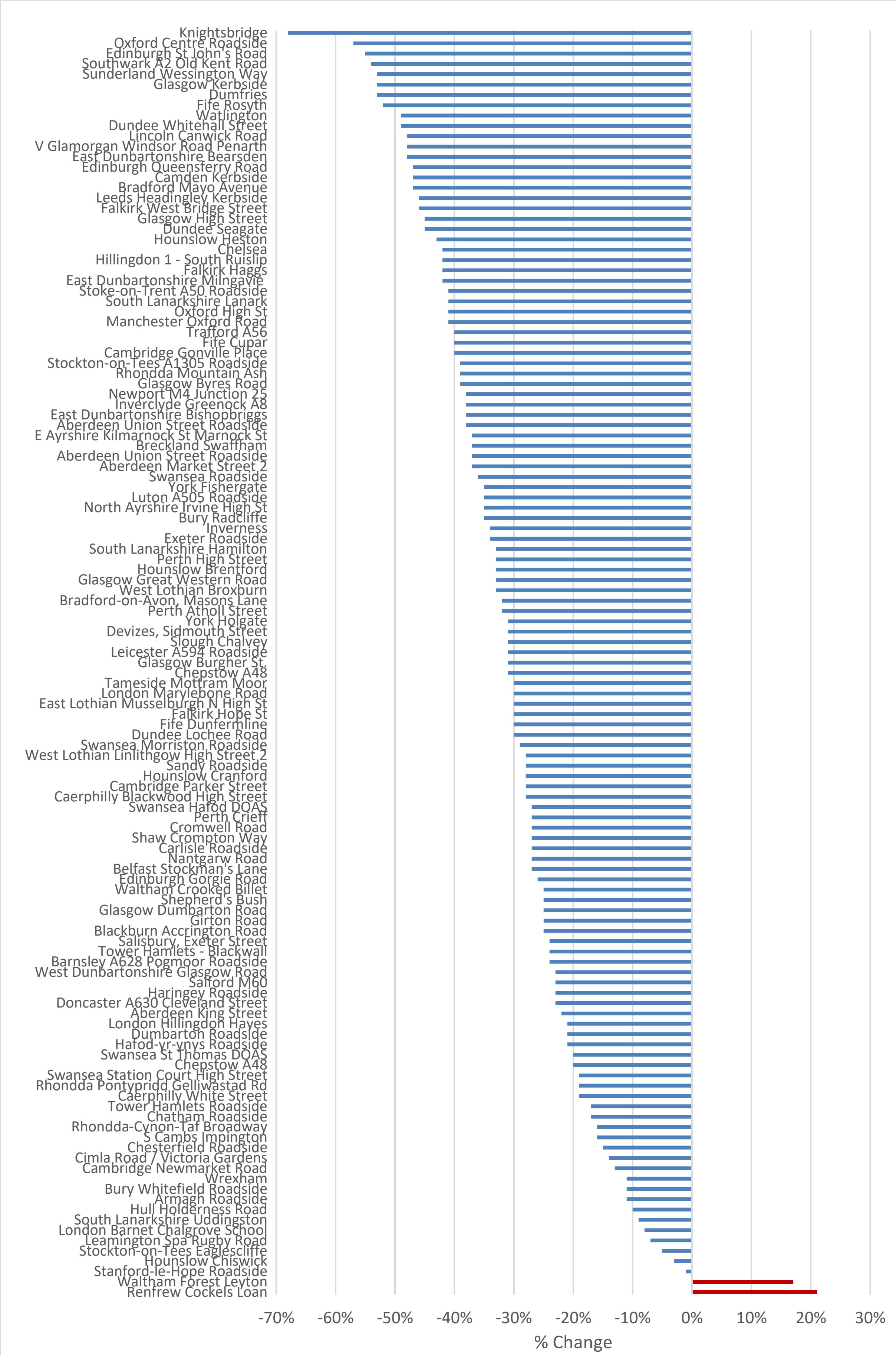


Figure A3.3: Relative Change in BRT-adjusted NO_2 at 123 Road Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020)

Figure A3.4: **Relative Change in BRT-adjusted NO_2 at [A] 56 Urban Sites and [B] 13 Rural Sites, [C] 3 Airport Sites, and [D] 9 Industrial Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020). Shown using Different Horizontal Scales**

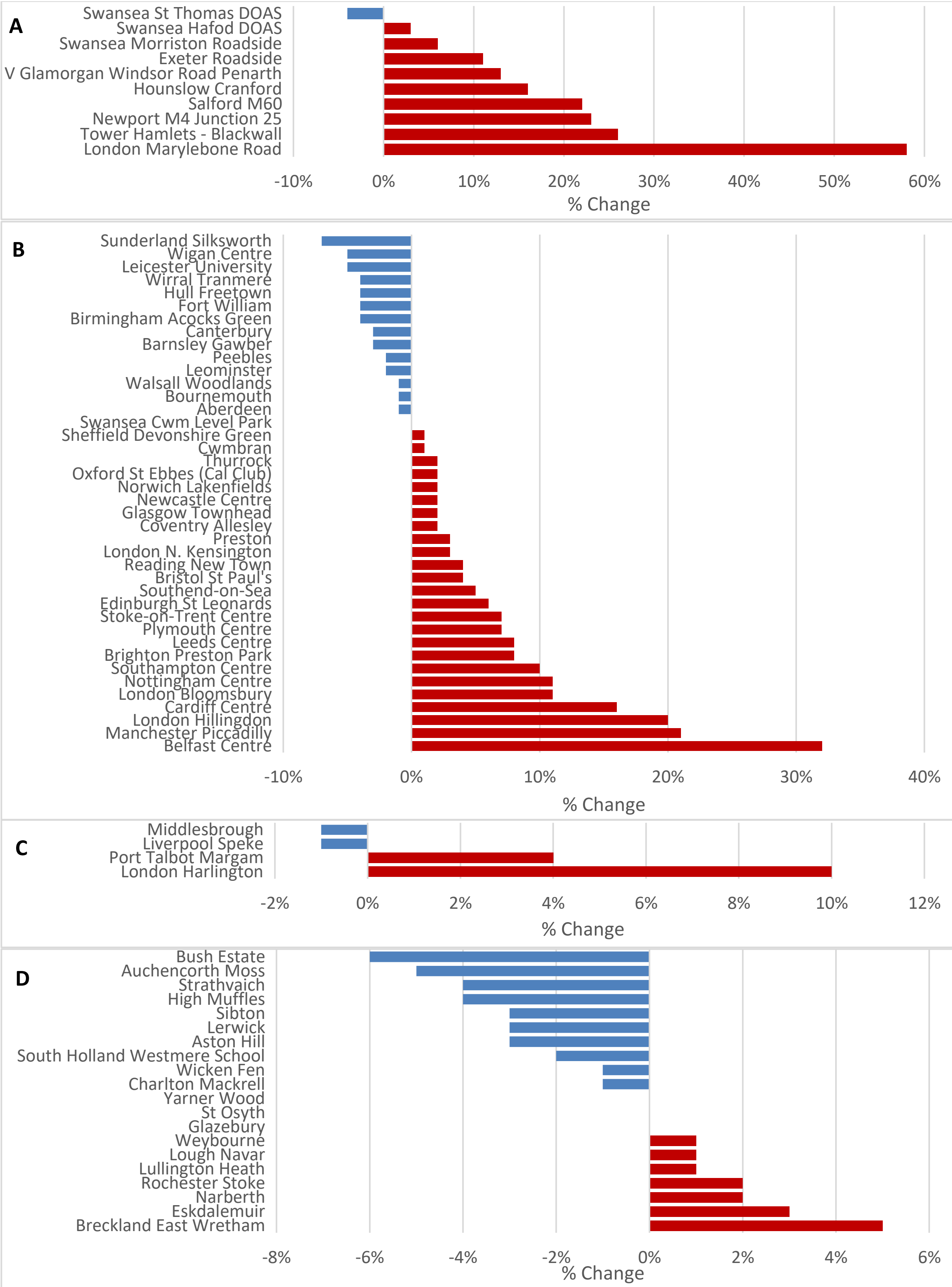


Figure A3.5: Relative Change in BRT-adjusted O_3 at [A] 10 Road Sites, [B] 40 Urban Sites, [C] 4 Industrial Sites, and [D] 20 Rural Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020). Shown using Different Horizontal Scales

Appendix 4 Geographical Distribution for NOx and NO_2

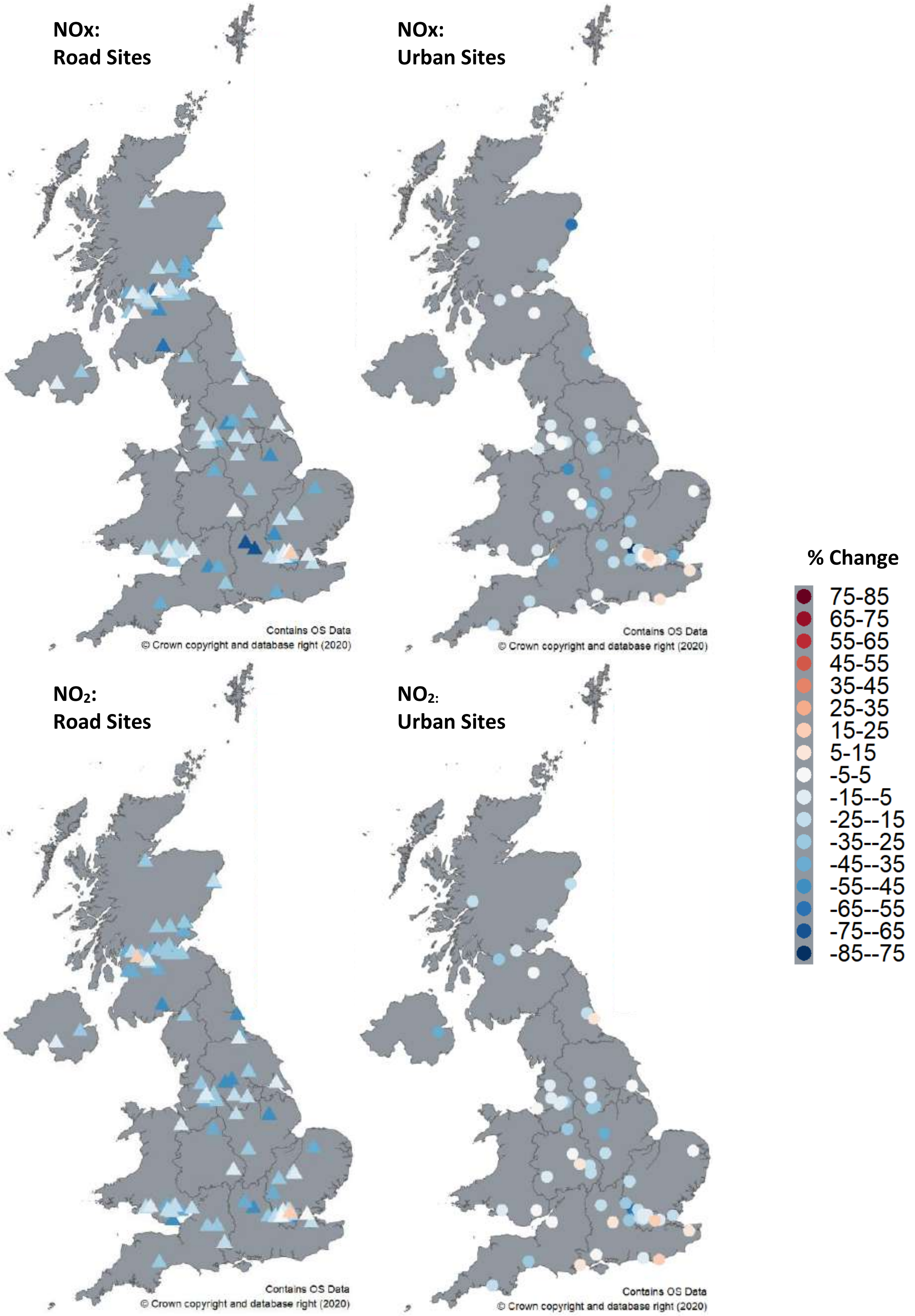


Figure A4.1: Relative Change in BRT-adjusted NOx and NO_2 at Road and Urban Sites (comparing mean of period 24th Mar to 9th Apr 2020 with period 1st Jan to 14th Mar 2020).

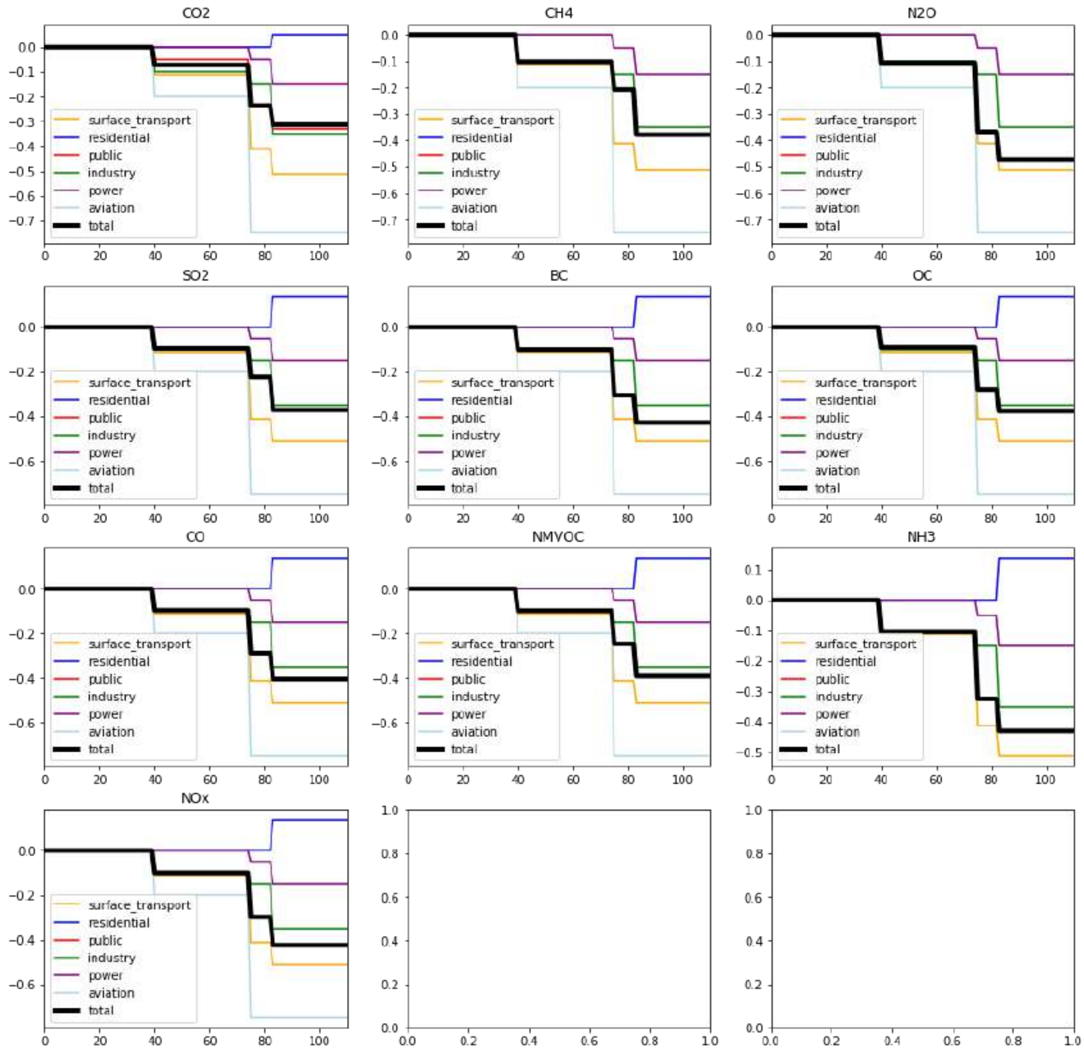
Submission to DEFRA on COVID-19 by the EU H2020 CONSTRAIN Project by P. Forster and D. Rosen.
Contact: d.z.rosen@leeds.ac.uk

This analysis estimates changes in daily UK emissions since the start of the COVID-19 pandemic for a range of atmospheric species (CO_2, CH_4, N_2O, SO_2, black carbon, organic carbon, CO, non-methane volatile organic compounds, NH_3, and NO_x). Given the lack of real-time emissions data, estimates are made using data representing changes in activity, such as electricity demand or road and air traffic, rather than direct changes in emissions themselves. For example, changes in the residential sector are inferred from smart meter data.

In the plot below, the estimated emissions during the pandemic are compared to mean daily emissions for 2019 (i.e. the latest available year) to provide a measure of relative change compared to pre-COVID conditions (i.e. all changes are relative to the typical activity level pre-pandemic).

Fractional changes in UK emissions (y-axis) are estimated for days since the start of the pandemic (x-axis) for the following sectors: surface transport, residential, public buildings and commerce, industry, power, aviation and industry; using power consumption, mobility (Google and Apple), and flight data. The fractional change by sector (coloured lines) is the same across all species; the total fractional emission change (black) is species dependent.

Next steps include developing emissions projections for the next 2-3 years, based on proposed Government action and economic incentives. Our intention is to use this to assess how the emissions changes impact on a range of climate parameters, including radiative forcing and temperature change, but the data may also find other useful applications.



Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK

Air Quality Expert Group response from Prof John Gulliver, Prof Anna Hansell, and Dr Calvin Jephcote, Centre for Environmental Health and Sustainability, University of Leicester

Background

The Centre for Environmental Health and Sustainability is a multidisciplinary research centre at the University of Leicester with particular interest in outdoor and indoor exposures to air pollution and their potential long-term health consequences. It also hosts a new NIHR Health Protection Research Unit (HPRU) in Environmental Exposures and Health, starting April 2020 with a focus on the built environment, including indoor exposures. We provide responses to four AQEG questions below.

Q. Can you provide estimates for how emissions and ambient concentrations of NOx, NO2, PM, O3, VOC, NH3 etc may have changed since the COVID outbreak? Where possible please provide data sets to support your response.

Introduction: Dramatic falls in air pollution related to lockdown were widely reported in the media in March and April. Reports appeared to be mainly related to NOx measurement data from roadside sites in the UK and from international satellite data on NO_2 relating to countries such as China and Italy with much stricter lockdowns and restrictions on industrial and farming activity than in the UK. However, at the same time as these media reports, Defra (https://uk-air.defra.gov.uk/forecasting/) were forecasting air pollution episodes (with AQI 10) in the southeast of England. We investigated these apparent contradictions using data from selected monitoring sites.

Methods: We assessed changes in air pollution concentrations as a result of "lockdown" during April 2020. We did not include the first week of lockdown (24^{th} – 31^{st} March 2020) as this was a period of transition in the reduction of passenger transport and adjusting to new patterns of behaviour. We analysed data on air pollution concentrations from routine, fixed-site measurements in the Defra air quality network(https://uk-air.defra.gov.uk).

We chose the seven sites contrasting in location and site type that had complete data for both NO_2 and $PM_{2.5}$: Birmingham Acocks Green (urban background), Leamington Spa Centre (urban background), Leamington Spa Rugby Road (urban traffic), London Bexley (suburban background), London Marylebone Road (urban traffic), London North Kensington (urban background), and Norwich Lakenfields (urban background).

Data were extracted from the Defra data archive using Openair (http://www.openair-project.org) software in R. We calculated daily concentrations of NO_2 and $PM_{2.5}$ for the period 1^{st} April to 28^{th} April 2020 and 4-weekly average concentrations for the same period in 2019. We counted the number of days in the period 1^{st} April to 28^{th} April 2020 where the daily average concentration was higher than the 4-week average concentration in 2019.

Results: A table of April 2020 averages compared with those for 2019 and of days in 2020 above the 2019 average are shown in Table 1. Time-series of daily average concentrations during April 2020 for London Marylebone (urban traffic site type) and London North Kensington (urban background site type) are shown in Figure 1, with similar display for the other 5 sites shown in Figure 2.

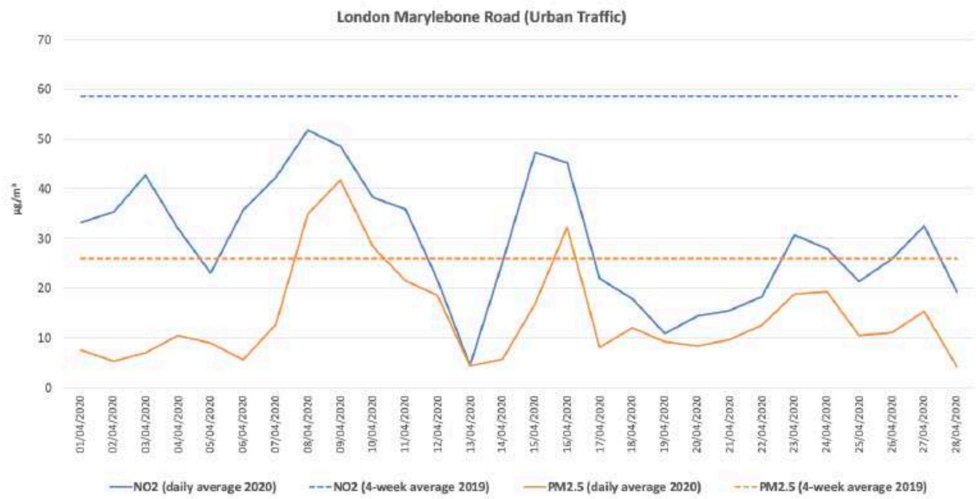
There were overall reductions in four week mean concentrations of NO_2 and $PM_{2.5}$ in April 2020 compared to the same period in 2019 with the exception of NO_2 at London Bexley (suburban site), which increased. Further, some sites experienced days exceeding the four-weekly average of the previous year.

Table 1: Comparison of 4-week (1st to 28th) average NO_2 and $PM_{2.5}$ concentrations (µg/m³) in April 2019 and 2020

		NO_2			$PM_{2.5}$		
Site name	Site type	4-week average 2019 (µg/m³)	4-week average 2020 (µg/m³)	Number of days in 2020 above the 2019 4-week average	4-week average 2019 (µg/m³)	4-week average 2020 (µg/m³)	Number of days in 2020 above the 2019 4-week average
Birmingham Acocks Green	UB	20.2	12.1	2	18.0	12.7	5
Leamington Spa Centre	UB	15.8	9.3	1	20.9	10.4	2
Leamington Spa Rugby Road	UT	16.7	9.4	3	21.0	11.3	3
London Bexley	S	28.2	29.2	11	23.6	17.3	6
London Marylebone Road	UT	58.6	29.3	0	26.0	14.3	4
London North Kensington	UB	30.3	19.0	5	21.0	13.7	4
Norwich Lakenfields	UB	13.5	10.0	7	20.3	11.5	2

UB – urban background; UT - urban traffic; S – Suburban

Figure 1: Daily concentrations of NO_2 and $PM_{2.5}$ from 1st - 28th April 2020 compared to 4-week average concentrations from 2019 at London Marylebone Road and London North Kensington



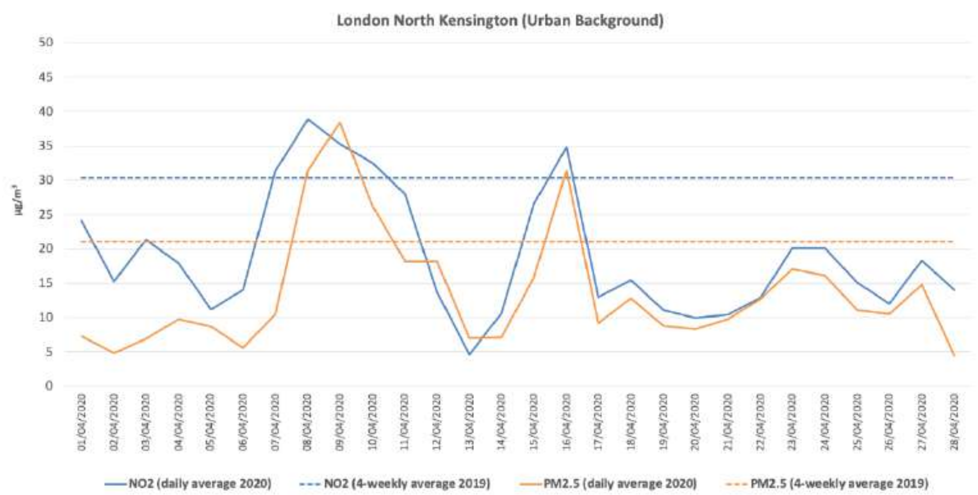
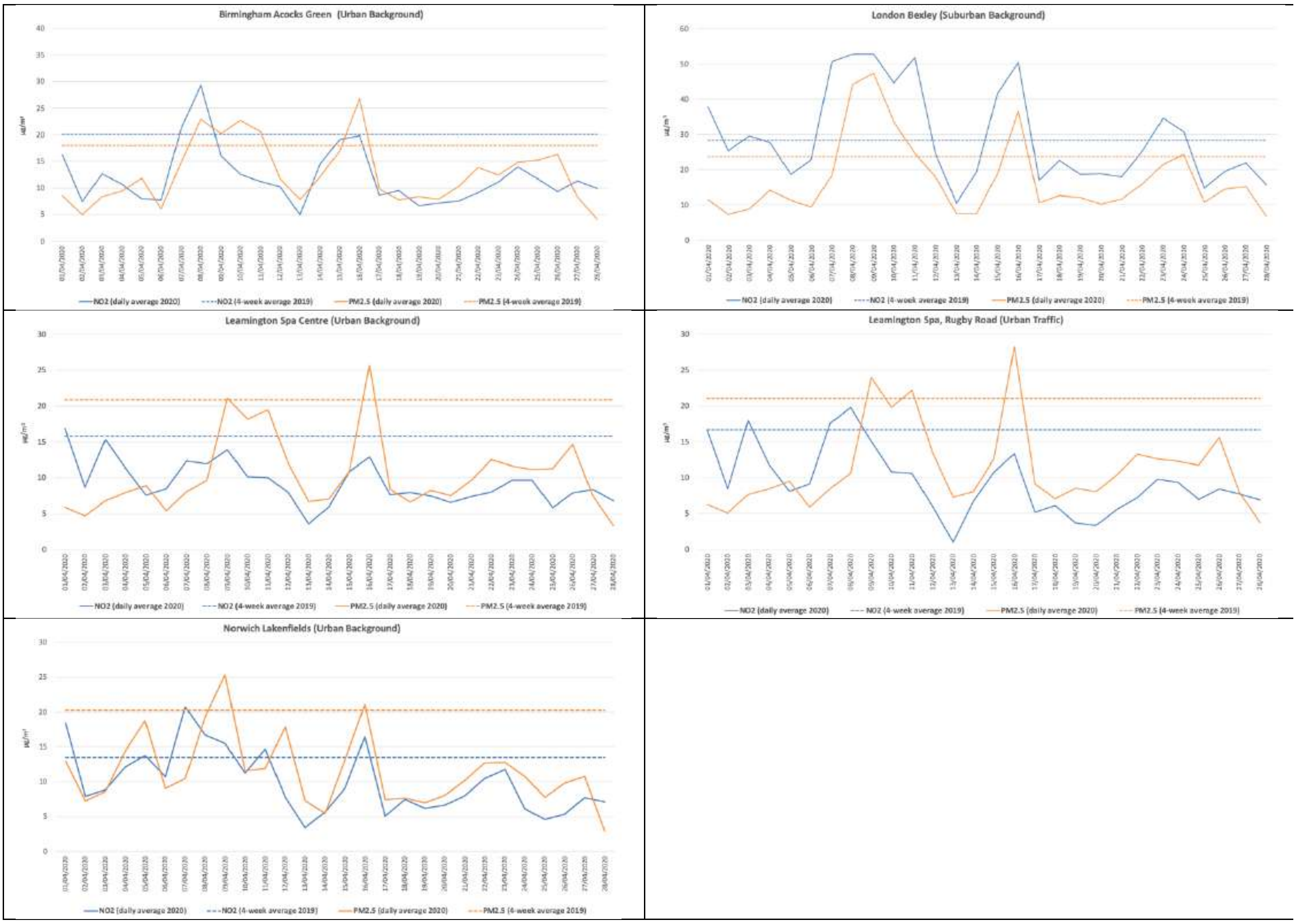


Figure 2: Daily concentrations of NO_2 and $PM_{2.5}$, 1st - 28th April 2020 compared to 4-week average concentrations from 2019 at Birmingham Acocks Green (UB), London Bexley (S), Leamington Spa Centre (UB), Leamington Spa Rugby Road (UT), Norwich Lakenfields (UB)



The relative extent of the drop in mean concentrations was approximately 25-50% depending by site and pollutant, with lower relative falls seen in non-roadside sites in metropolitan areas. For the Marylebone Road site (urban traffic) in London, concentrations fell substantially in April 2020 for NO_2 (on average by 50%) and $PM_{2.5}$ (on average by 45%) compared to April 2019, with no days exceeding the 4-week average NO_2 concentration for 2019. Similarly, for the urban traffic site in Leamington Spa NO_2 concentrations fell on average by 45% and $PM_{2.5}$ concentrations fell on average by 50%. There were also substantial reductions in NO_2 and $PM_{2.5}$ at other sites, including background sites such as Birmingham Acocks Green (40% for NO_2 and 29% for $PM_{2.5}$).

With the exception of at London Bexley for NO_2, the daily average concentration of NO_2 and $PM_{2.5}$ exceeded the 4-week concentration average for 2019 on <25% (7) of days. As Figure 1 and the Figure 2 shows, there were periods of elevated levels of NO_2 and $PM_{2.5}$ during April 2020, over several days, which likely relate to periods of warm and sunny weather, which are associated with secondary formation of air pollution, plus far-travelled pollution on easterly winds from neighbouring continental countries.

We have focussed on NO_2 and $PM_{2.5}$ as two of the main pollutants of concern for health. We did not look at NOx but we expect NOx concentrations to be substantially lower as they relate directly to expected reductions in emissions. We noted that due to seasonally good weather for much of April

2020, including warm days with unbroken sunshine (i.e. high levels of UV), ozone concentrations were generally higher in April 2020 than in April 2019 at some locations.

Discussion: There were overall reductions in four-week mean concentrations of NO_2 and $PM_{2.5}$ in April 2020 compared to the same period in 2019 with the exception of NO_2 at London Bexley (suburban site), which increased. The relative extent of this drop was not uniform across sites, with lower relative falls seen in non-roadside sites in urban areas. Despite the reductions on average, there were days with air pollution episodes, but further work is needed to establish if these were of shorter duration than expected from previous years. Spatial variability is important to consider in health studies where we are interested in changes in exposures at places where people live – for which urban background sites are likely a better indicator than roadside sites. We plan to follow up this initial investigation with a spatial analysis that also include ozone. Changes in air pollution during lockdown are likely to provide insights into changes in air pollution exposures with transport interventions such as a move to electric vehicles.

Q. How might public exposure to air pollution have changed as a consequence of recent restrictions on movement?

There are an estimated 10 million key workers in the UK, whose exposure probably has not changed greatly. In those working from home or furloughed or children home-schooling, we expect that time spent in indoor vs. outdoor settings for those of school and working age will not have changed greatly given the restrictions, but that indoor exposures are almost all from the home environment rather than home plus work/school/social settings i.e. the home environment has become a much larger determinant of indoor exposure than previously.

Further data on this may come from questions planned for UK cohorts, but time-activity data in lockdown setting would be valuable.

Q. What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?

We anticipate higher adult and childhood population exposures to NO_2 from gas appliances, from working from home/furlough and from cooking at home given closure of restaurants and many take-aways. This is likely to also result in more CO exposure with faulty or poorly maintained appliances. We also expect higher personal exposures to VOCs and other chemicals from increased use of cleaning products (e.g. bleach, scented products). This is of concern as cleaning products have been associated with higher rates of chronic respiratory disease. Further, VOCs are associated with not only irritation of eyes and respiratory tract, but also with allergies, asthma and cancer risks.

There is no regulatory monitoring of indoor exposures, but this topic is one of the work areas in the new HPRU on Environmental Exposures and Health at University of Leicester starting April 2020.

Q. Based on what is already known about air pollutants as respiratory irritants or inflammatory agents, can any insights be gained into the impact of air quality on viral infection?

There are a number of epidemiological studies finding associations between air pollution and exacerbations of chronic respiratory disease such as COPD and asthma that are often driven by viral infections. Most regulatory air pollutants are irritants and/or pro-inflammatory agents. There are a number of toxicological mechanisms that need further evaluation including non-specific impacts of inflammation on host immunity, specific impacts of pollutants on receptors such as ACE2 by which SARS-CoV-2 enters cells, and contribution of air pollution to cytokine production during infection thereby potential contribution to the cytokine storm that is a feature of Acute Respiratory Distress Syndrome ARDS seen in severe COVID-19 disease.

Breathe-London submission to the AQEG/Defra Covid-19 call.
This document was prepared by the BL partners (Air Monitors/ACOEM, Cambridge Environmental Research Consultants, Environmental Defense Fund, National Physical Laboratory and the University of Cambridge).
Queries to: David Carruthers (david.carruthers@cerc.co.uk) and Rod Jones (rlj1001@cam.ac.uk).

The Breathe London (BL) project is a Clean Air Fund (CAF) funded initiative to investigate air quality across London. The three measurement components of BL are a network of around 100 low cost AQMesh air quality sensors (pods) which have run from October 2018 to the present, two Google Street view cars fitted with reference standard air quality instrumentation which ran for 14 months from September 2018, and a study using wearable air quality sensors. Data and details of the partners involved with BL can be found at https://www.breathelondon.org. This document uses data from the AQMesh air quality network measurements and includes associated air quality modelling. High resolution versions of figures are available in the Appendix.

1) *Observed changes in air quality associated with the Covid-19 traffic reductions*

Nitrogen dioxide (NO_2) and NOx

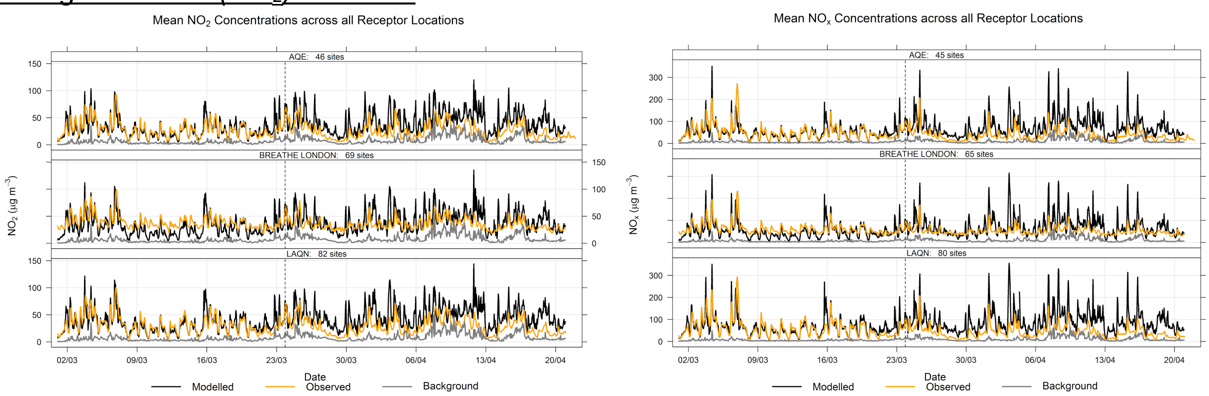


Figure 1.1 Modelled (black) and observed (yellow) AQE, LAQN and BL network hourly average time series for NO2 and NOx for March 2020 onwards. Modelled data were calculated using the ADMS-Urban dispersion model. The dotted line marks 8.30pm on 23 March, when the lockdown was announced.

Figure 1.1 shows hourly averaged time series for March 2020 onwards for NO_2 and NOx, both measured by the Air Quality England (AQE), BL and London Air Quality Network (LAQN) networks and modelled using the air quality model ADMS-Urban. Similar features are observed in all three networks, although as network sites differ, some differences are to be expected.

The BL network shows the same patterns as seen in the reference networks, though a small positive bias is evident. Diagnostic work has shown that this appears to be largely due to an increasing O_3 cross interference as the NO_2 sensors age. A correction algorithm to account for this is in the final stages of development, but has not been applied to the data shown in this document.

The ADMS model calculations, which in these figures assume no change in traffic characteristics following lockdown, show generally good consistency prior to the 23rd March, but a significant overestimation following, consistent with a reduction in emissions from traffic in the observations. This is discussed further in section 2. Observations from all three networks (and model results) also show significant variability, including periods of elevated pollution levels associated with more stagnant meteorological conditions.

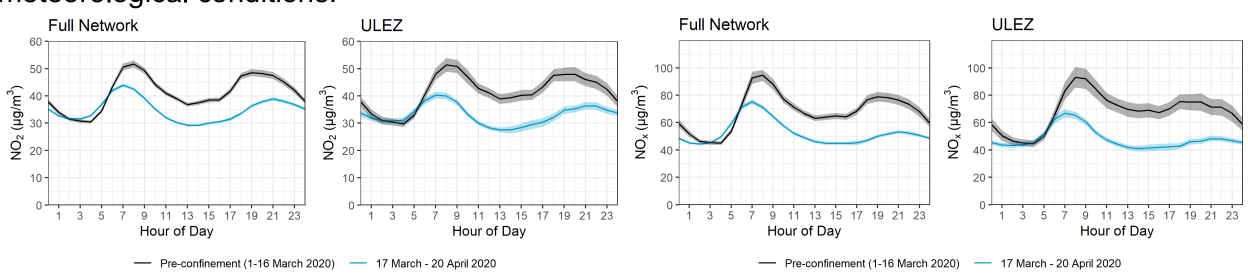


Figure 1.2 Differences in the diurnal patterns of NO_2 and NOx for the pre- and post-lockdown periods for the full BL network and for the BL sites inside the ULEZ. The shading shows the 95% confidence limits of the network averages.

Figure 1.2 shows diurnal patterns of NO_2 and NOx pre- and post-lockdown (1st - 16th March, 17th March - 20th April respectively). Key conclusions from figure 1.2 are that the BL network shows that there have been statistically significant reductions in both NO_2 and NOx during the covid-19 lockdown period across the BL network (15% and 23% respectively). They also show that the effects are greater within the ULEZ (20% and 29% respectively), but that the reductions are much reduced and in many cases are statistically insignificant during the night-time where traffic contributions to NO_2 and NOx are expected to be much lower.

Particulate matter ($PM_{2.5}$)

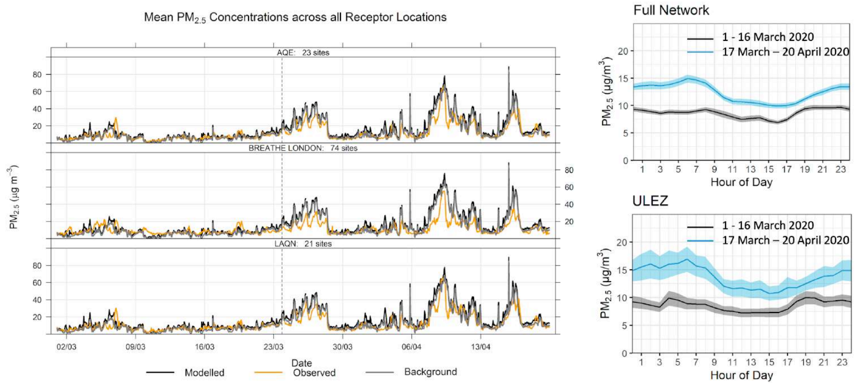


Figure 1.3 Left: Modelled (black) and observed (yellow) AQE, LAQN and BL network hourly average time series for $PM_{2.5}$ for March 2020 onwards. Right: Differences in the diurnal patterns of $PM_{2.5}$ for the pre- and post-lockdown periods for the full BL network (upper) and for the BL sites inside the ULEZ (lower). The shading shows 95% confidence limits of the network averages.

Modelled and observed hourly averaged time series for March 2020 onwards are shown for $PM_{2.5}$ in figure 1.3 for the AQE, BL and LAQN networks. Similar changes are again observed in all three networks. Unlike for NO_2 and NOx, however, in this case there is no obvious suggestion of model overestimation during the lockdown period, and observations and model show a series of elevated PM episodes post lockdown likely associated with long range transport of secondary PM. Reflecting this, BL PM measurements show increases across the network and in the ULEZ (43% and 59% respectively).

Covid-19 conclusions:

- The BL network suggests that NO_2 and NOx levels in London have reduced post lockdown, with the reductions greater within the ULEZ (15 - 23% *vs* 20 - 29%).
- Implementation of the O_3 cross interference correction algorithm and other ratifications will lead to a refinement of these numbers.
- $PM_{2.5}$ concentrations from the BL network have shown increases directly post the covid-19 lockdown. However, this appears to be associated in significant part with meteorological effects and is not necessarily a reflection of changes in local emissions. This requires further analysis.

2) Assimilation of air quality measurements into models to improve emission inventories

CERC have developed a data assimilation scheme that applies a Bayesian inversion technique to a high resolution (street-level) atmospheric dispersion model to modify pollution emission rates based on local measurements (Carruthers *et al.*, 2020). Results for NOx for the Covid-19 period using ADMS-Urban are presented below in figures 2.1 to 2.3. In this experiment, assumed *a priori* uncertainties were 100%, 20%, 10% and 30% for road traffic emissions, other emission types, LAQN and AQE measurements and AQMesh measurements respectively. While agreement between modelled and measured NOx concentrations in London was generally good during the early part of March, most sites in both the BL and reference networks show marked reductions in measured and *modified* model NOx (i.e. results from model integrations into which observations had been assimilated) from mid-March onwards. This general behaviour is reflected in the increased visibility of the red lines (un-modified model) in the individual panels post lockdown in figures 2.1 and 2.2.
Figure 2.3 shows how measured concentrations and road traffic emissions derived by data assimilation have changed since the Government first issued social distancing advice. The derived emissions first show a marked drop over the weekend of 21st/22nd March, which coincides with the announcement on the evening of Friday 20th March that bars and restaurants should close.

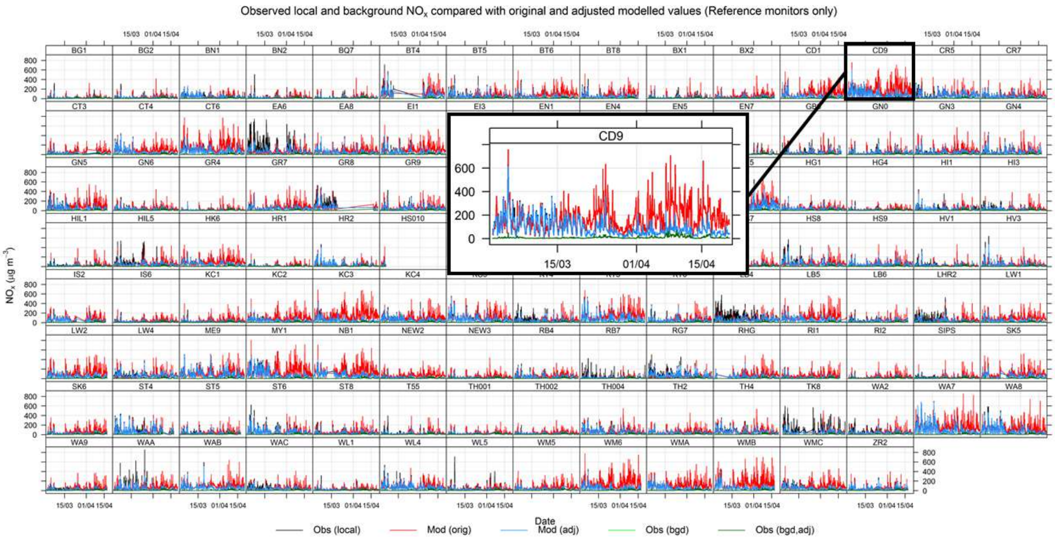


Figure 2.1: Time series of hourly ADMS-Urban modelled (red) and observed (black) NOx (ug/m3) at the LAQN and AQE sites in London from 1 March to 20 April 2020. The 'adjusted' modelled concentration (blue) is calculated using road traffic emissions derived by assimilating local measured data with modelled concentrations. One site (CD9) has been highlighted for clarity (see text for further discussion).

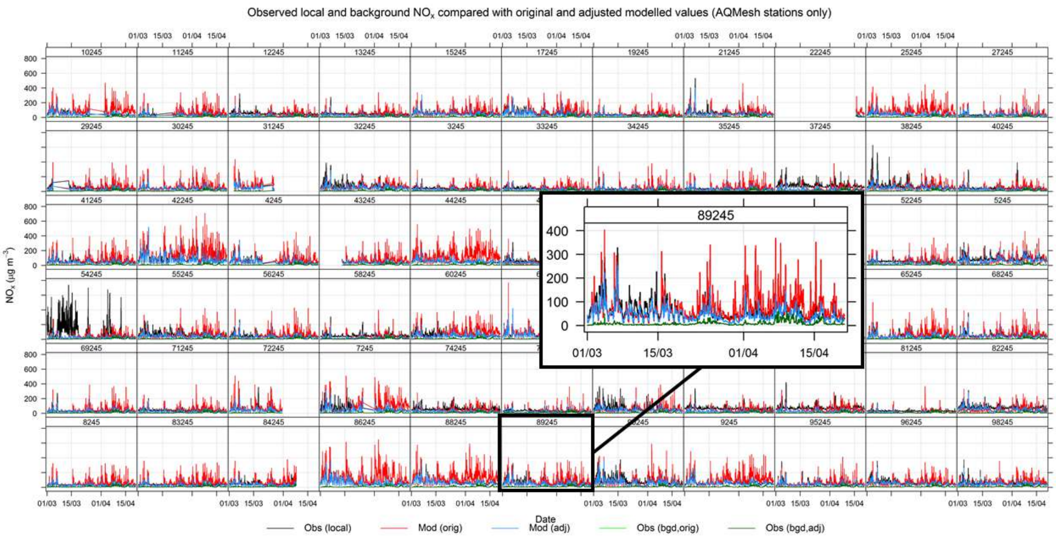


Figure 2.2: As figure 2.1 except for the BL network. In this case pod 89245 has been highlighted (see text).

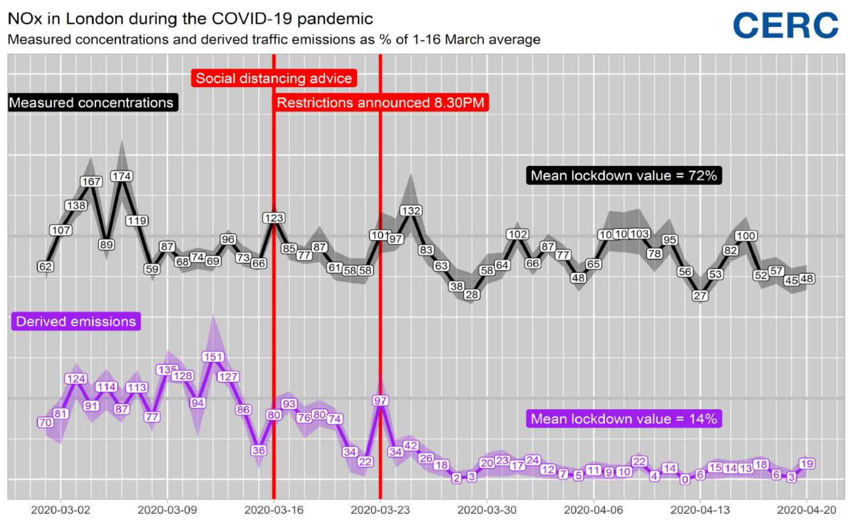


Figure 2.3: London measured concentrations (black) and derived road traffic emissions (purple), both as a percentage of the average over the 1-16 March pre-lockdown period. Measured concentrations are from all available sites in the LAQN, AQE and BL networks (184 sites in total); traffic emissions have been calculated by assimilating measurements using the ADMS-Urban model. The numbers indicate the median value; the shaded areas give the inter-quartile range. Key dates are shown by the red lines, as are the mean lockdown values (24th March onwards).

Monday 23rd March saw a widely-reported rush hour in London; this can be seen in both the measured concentrations and the derived emissions. The impact of the restrictions announced in the evening of 23rd March can be seen in the derived emissions in the following days, even though measured concentrations continued to rise through Wednesday 25th March, due to the sunny and still weather conditions. The lowest derived emissions are seen on the weekend of 28th/29th March, averaging 2-3% of pre-restriction levels. Weekday derived road traffic emissions of consistently around 14% of pre-restriction levels contrast with a second peak in measured concentrations on 1st April, again caused by still weather conditions. During April, road traffic emissions derived by data assimilation have remained very low, with the lowest levels at weekends, even when measured concentrations suggest little overall change if compared with only the first two weeks of March (e.g. 7-9th April).
The derived road traffic emissions depend on the assumed uncertainties input to the data assimilation scheme, but by directly linking measurements, including those from BL, with modelling these results provide quantifiable evidence that NOx emissions from road traffic in London have reduced dramatically during the lockdown.

Covid-19 conclusions:

- Assimilation of air quality observations into the ADMS model allows direct quantification of changes in NOx emissions associated with the Covid-19 lockdown; initial results suggest a reduction in road traffic NOx emissions to around 10-20% of pre-lockdown levels.
- The next step is to apply the data assimilation scheme to $PM_{2.5}$ over the same period.
- While the impact of the assimilation methodology is clear, further work is required to assess more fully the implications of the *a priori* assumptions made.

3) Future directions

Direct determination of emission indices (EIs)

The inclusion of measurements of CO_2 in all BL pods allows emissions indices (EIs - pollutant to CO_2 ratios) to be derived directly from the BL measurements (see e.g. Popoola et al., 2018).

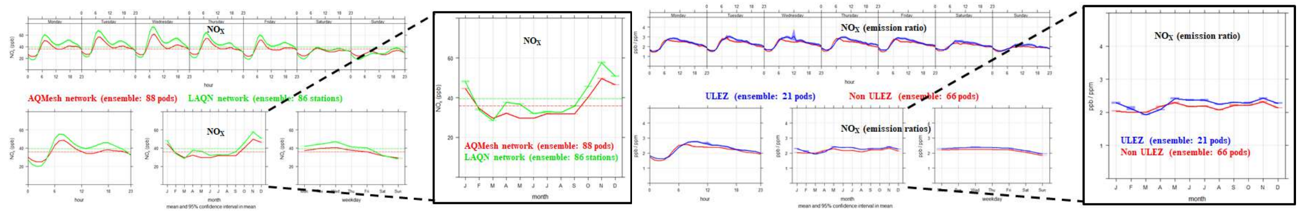


Figure 3.1. Diurnal statistics derived from the London networks. Left: comparison of the LAQN and BL networks. Right: emission indices from the BL network split by ULEZ and non-ULEZ.

This provides an important additional diagnostic of traffic (emission source) mix, but also removes the confounding effects of meteorology in assessing the effectiveness of interventions (in this case the seasonal increase in NOx apparent in the BL and LAQN networks, but not in the derived EIs - see figure 3.1).

Covid-19 conclusions:

- Future work including analysis of EIs for the Covid-19 lockdown period and development of the assimilation methodology to incorporate these observationally determined EIs will provide additional insights into the impacts of the Covid-19 lockdown.

Relevant references:

Carruthers D, Stidworthy A, Clarke D, Dicks J, Jones R, Leslie I, Popoola OAM and Seaton M, 2020: *Urban emission inventory optimisation using sensor data, an urban air quality model and inversion techniques.* International Journal of Environment and Pollution, vol. 66, issue 4, pp. 252-266 Available online

Popoola OAM , Carruthers D, Lad C, Bright VB, Mead MI, Stettler MEJ, Saffell JR and Jones RL, 2018: *Use of networks of low cost air quality sensors to quantify air quality in urban settings.* Atmospheric Environment 194 (2018) 58–70, doi.org/10.1016/j.atmosenv.2018.09.03

Appendix

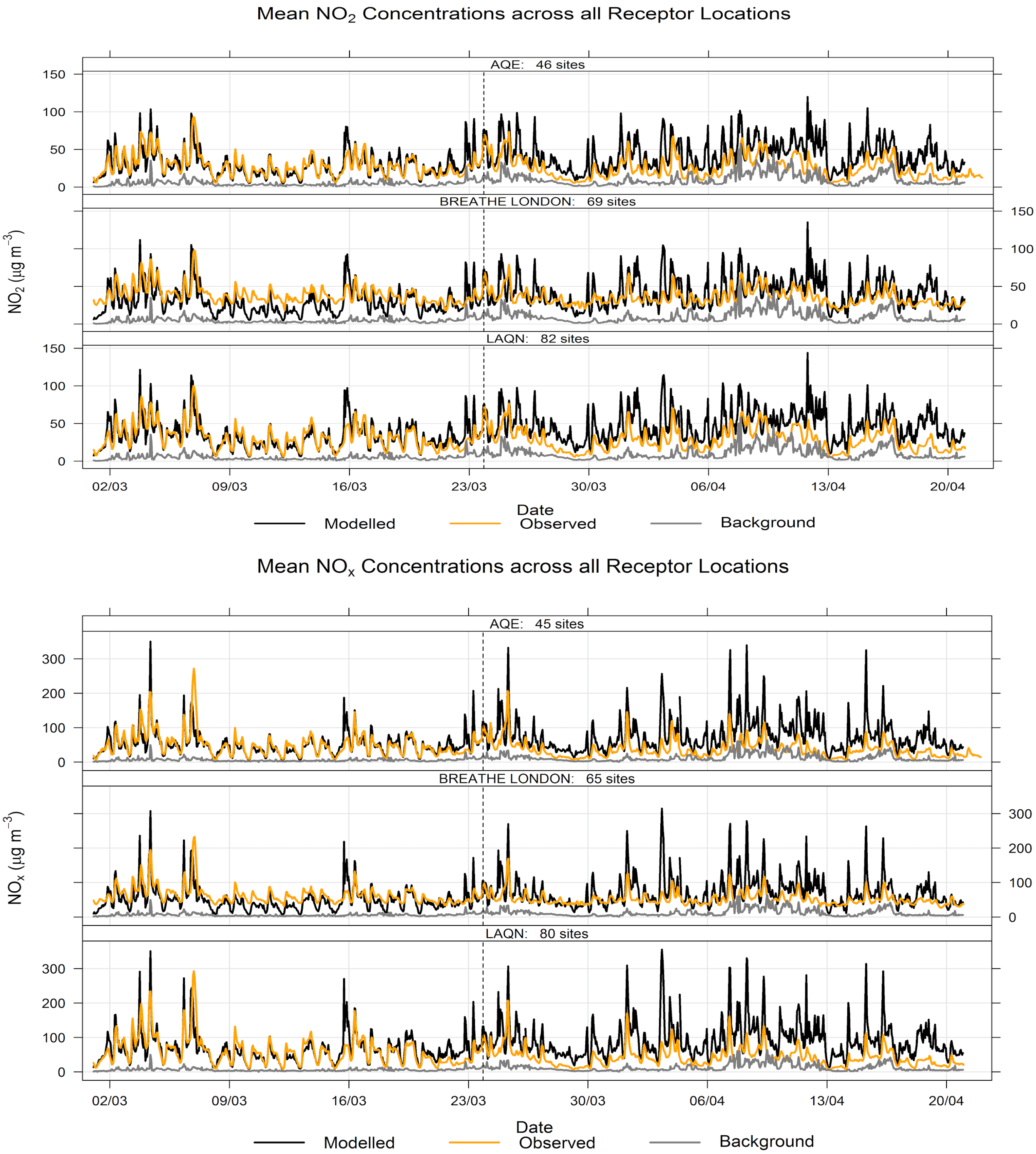


Figure 1.1 Modelled (black) and observed (yellow) AQE, LAQN and BL network hourly average time series for NO2 and NOx for March 2020 onwards. Modelled data were calculated using the ADMS-Urban dispersion model. The dotted line marks 8.30pm on 23 March, when the lockdown was announced.

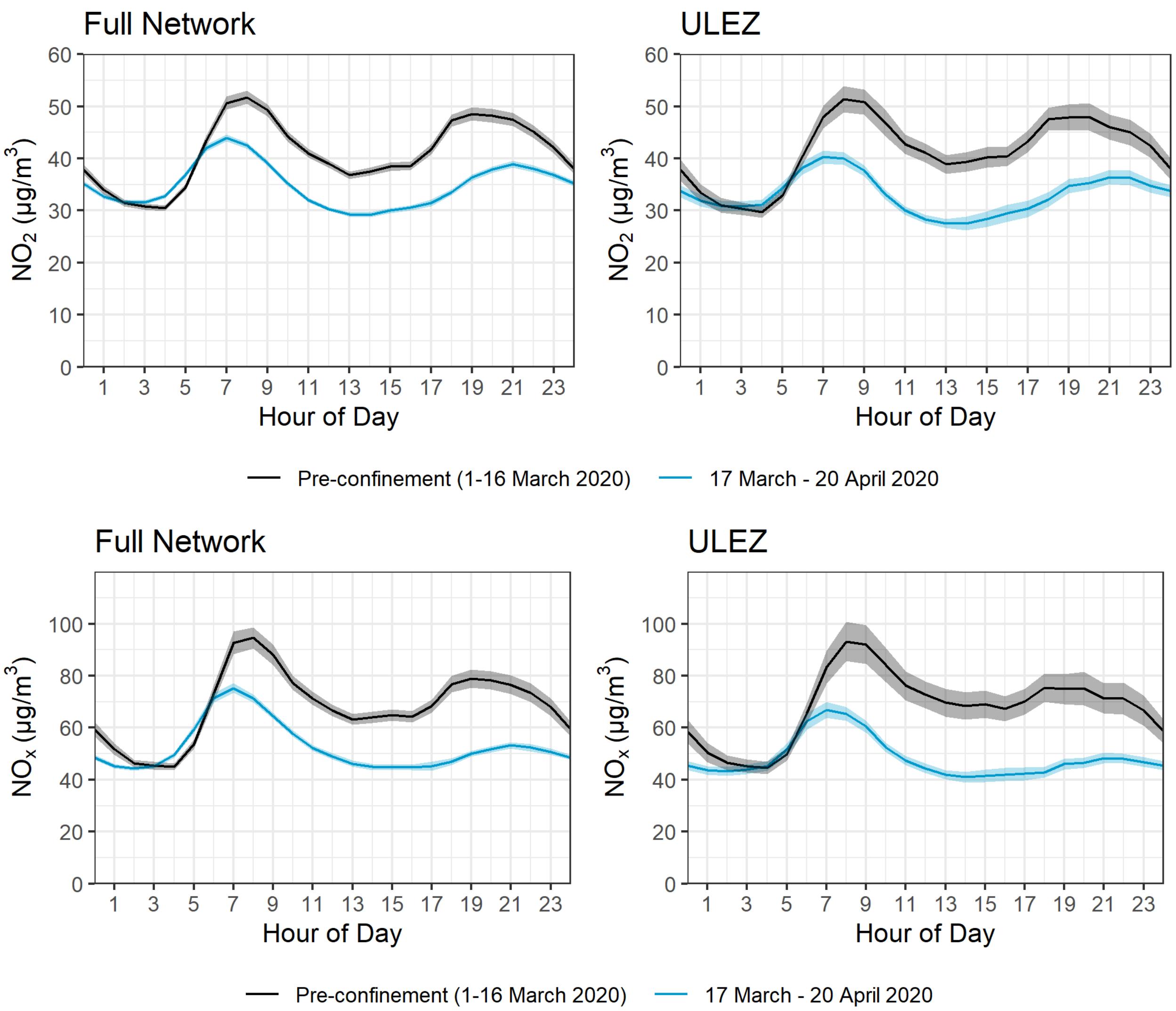


Figure 1.2 Differences in the diurnal patterns of NO_2 and NOx for the pre- and post-lockdown periods for the full BL network and for the BL sites inside the ULEZ. The shading shows the 95% confidence limits of the network averages.

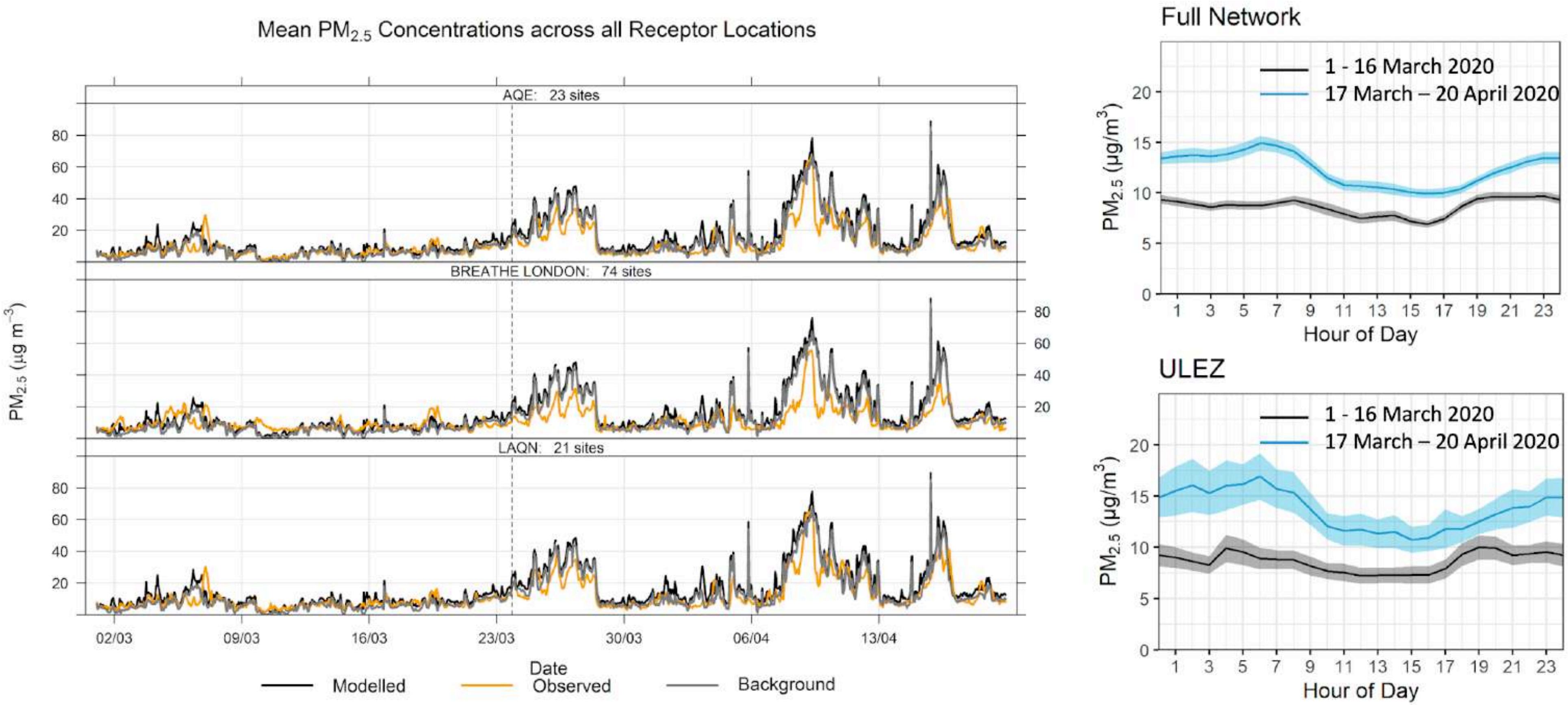


Figure 1.3 Left: Modelled (black) and observed (yellow) AQE, LAQN and BL network hourly average time series for $PM_{2.5}$ for March 2020 onwards. Right: Differences in the diurnal patterns of $PM_{2.5}$ for the pre- and post-lockdown periods for the full BL network (upper) and for the BL sites inside the ULEZ (lower). The shading shows 95% confidence limits of the network averages.

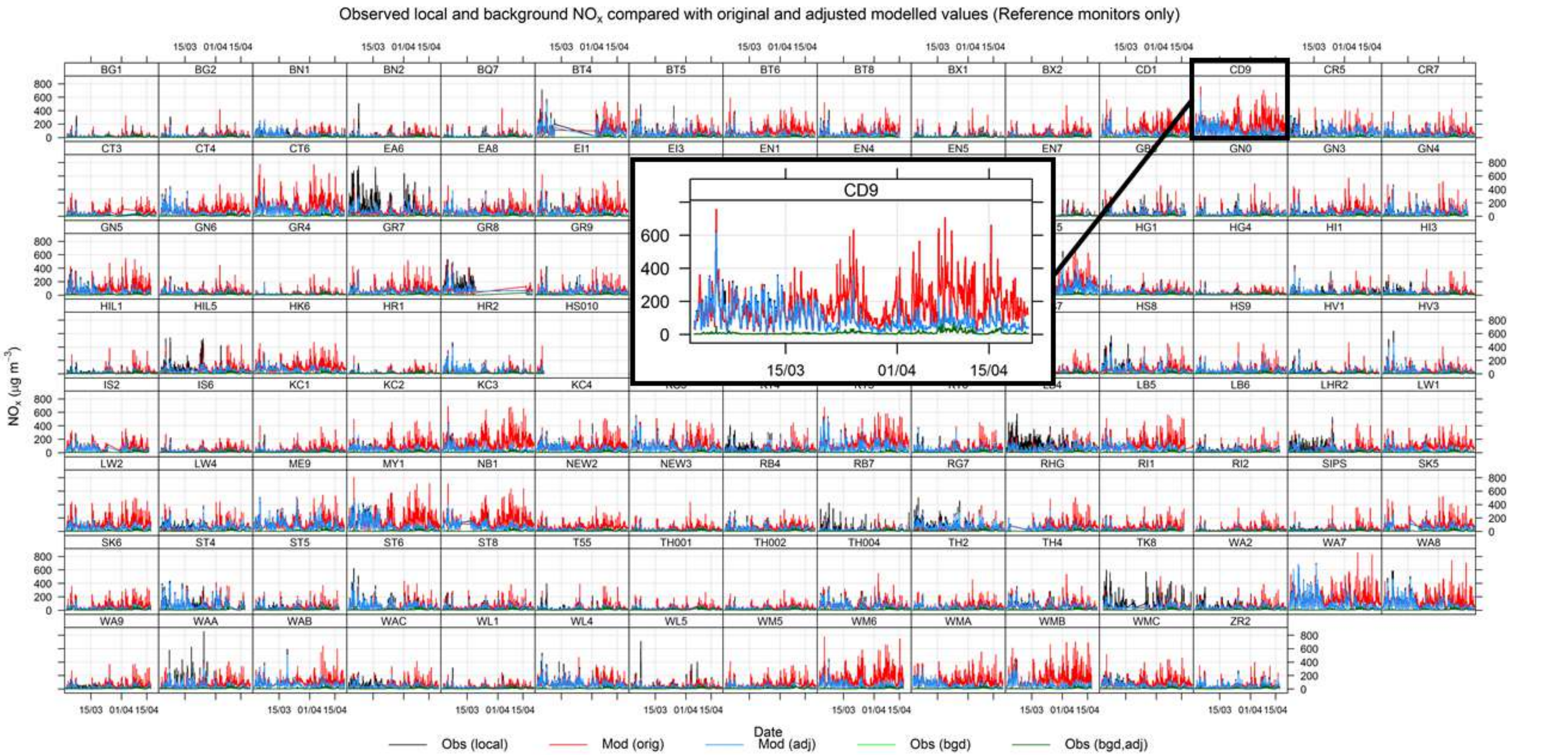


Figure 2.1: Time series of hourly ADMS-Urban modelled (red) and observed (black) NOx (ug/m3) at the LAQN and AQE sites in London from 1 March to 20 April 2020. The 'adjusted' modelled concentration (blue) is calculated using road traffic emissions derived by assimilating local measured data with modelled concentrations. One site (CD9) has been highlighted for clarity (see text for further discussion).

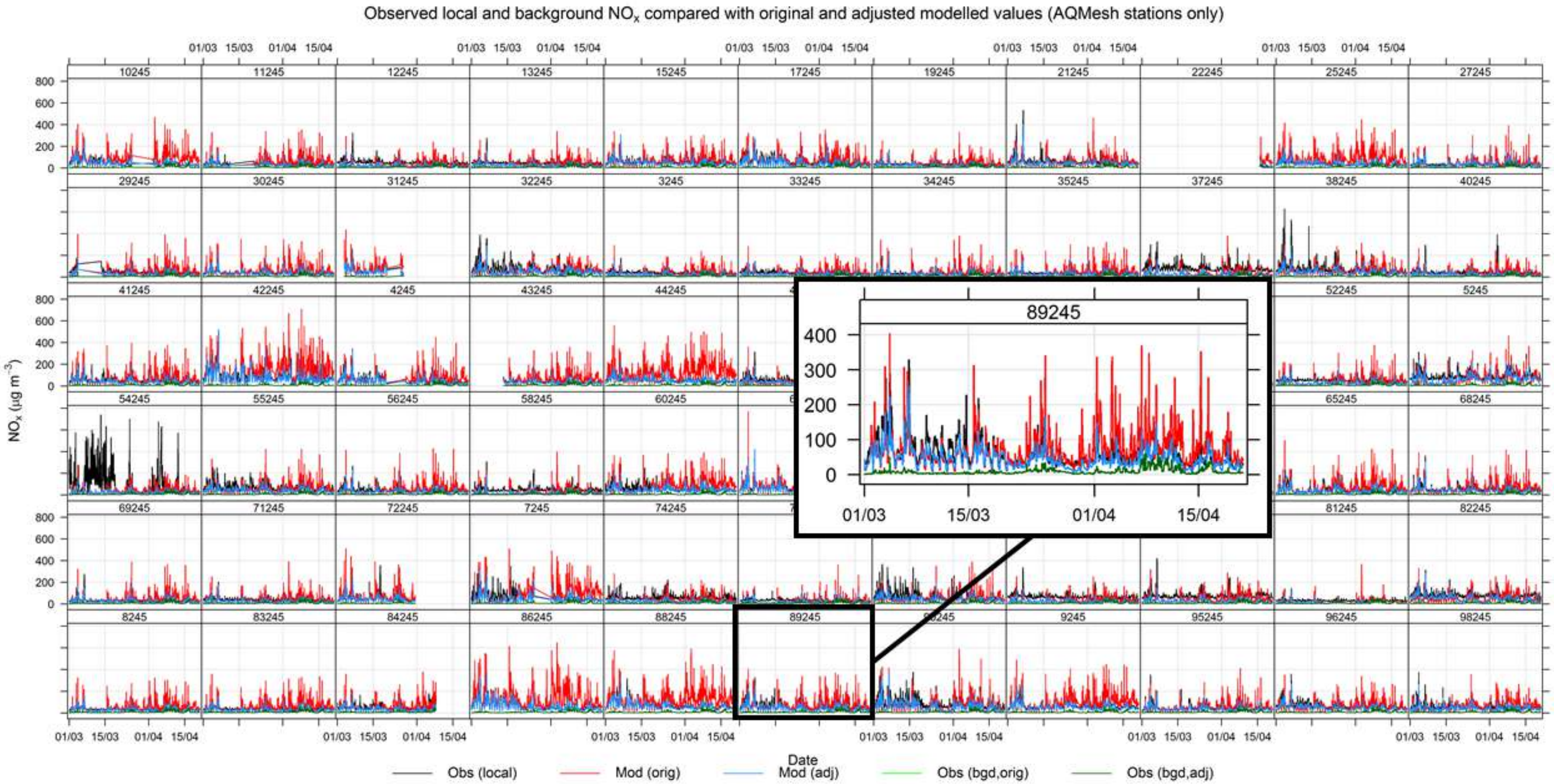


Figure 2.2: As figure 2.1 except for the BL network. In this case pod 89245 has been highlighted (see text).

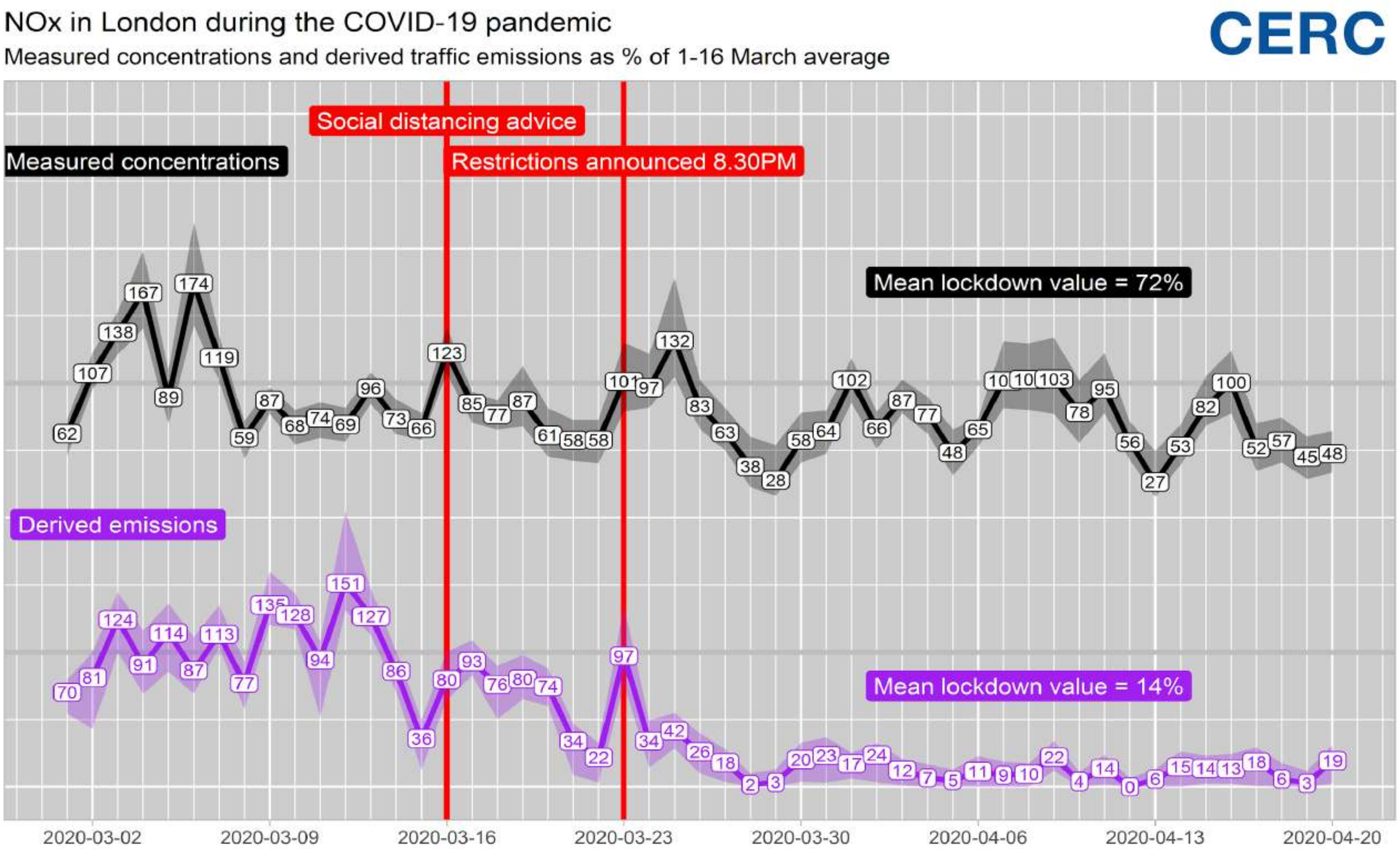


Figure 2.3: London measured concentrations (black) and derived road traffic emissions (purple), both as a percentage of the average over the 1-16 March pre-lockdown period. Measured concentrations are from all available sites in the LAQN, AQE and BL networks (184 sites in total); traffic emissions have been calculated by assimilating measurements using the ADMS-Urban model. The numbers indicate the median value; the shaded areas give the inter-quartile range. Key dates are shown by the red lines, as are the mean lockdown values (24th March onwards).

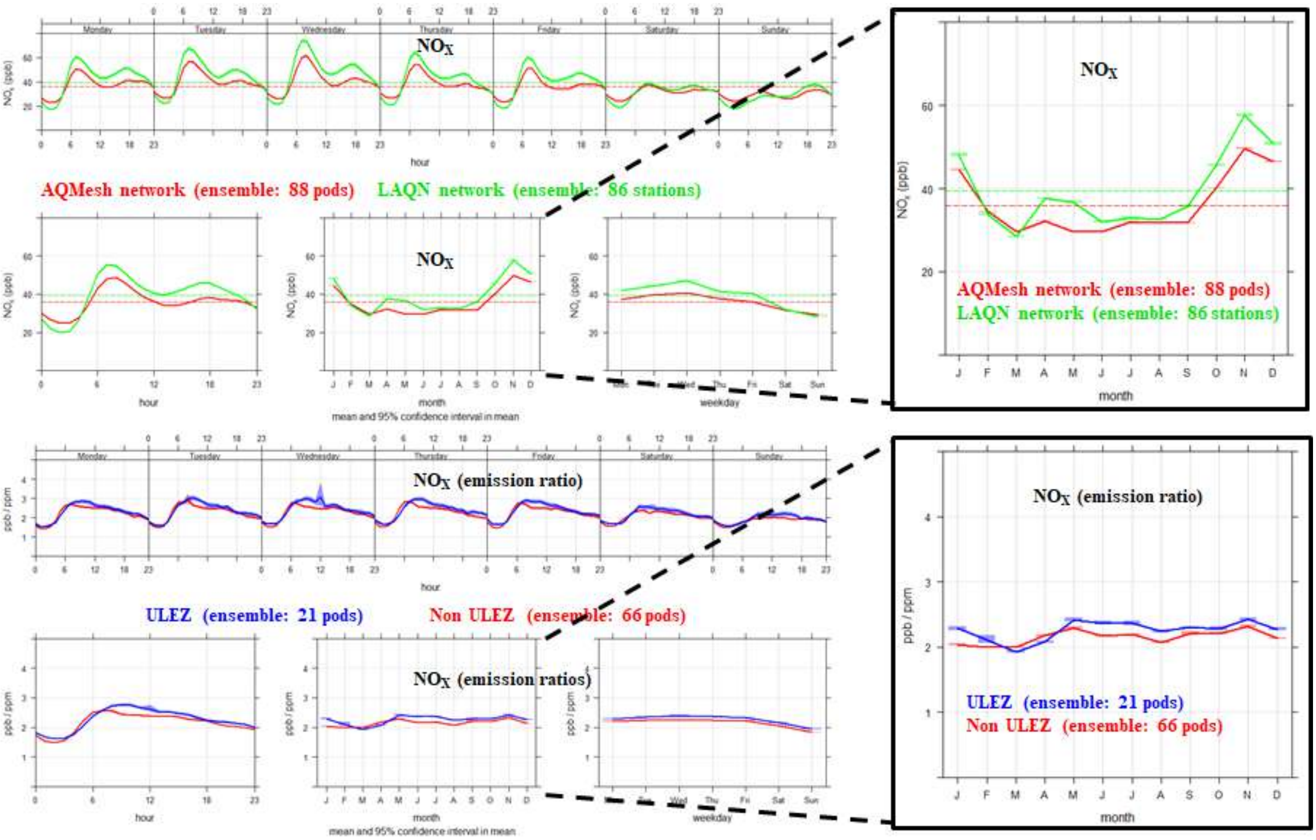


Figure 3.1. Diurnal statistics derived from the London networks. Left: comparison of the LAQN and BL networks. Right: emission indices from the BL network split by ULEZ and non-ULEZ.

Response to "Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK"

Jonathan Reid, School of Chemistry, University of Bristol

Are there any insights that can be gained from aerosol science on possible viral transmission mechanisms?

Much remains unknown surrounding the airborne transmission of SARS-CoV-2, however aerosol science should (and is, in some cases) tackling the following challenges:

- *Is the virus airborne?* There are a handful of studies which have now reported RNA signatures, identified by PCR, of the airborne virus, using air samplers/filters, collecting samples from room ventilation etc [1–5]. These studies do not mean the virus remains viable when in aerosol when generated by coughs, sneezes, breathing, aerosol generating procedures by clinicians, resuspension of material etc.
- *How long does the virus remain viable and infectious when airborne?* There have only been two studies in Goldberg drums so far, each at only 1 relative humidity (RH) [6,7]. A full RH and temperature dependence is required to understand the likelihood of airborne transmission. This will be important to understand seasonal variations in transmission. The impact of engineering controls in buildings (air conditioning, filters, UV light etc.) can be guided by measurements of these dependencies helping to reduce airborne transmission.
- *What dose is required by airborne transmission?* Even though the first studies investigating viability have suggested the virus remains infectious for between 2 and >16 hours, determining the inhaled dose required for infection is a significant unknown [8,9].
- *Is the distinction between droplets and aerosols relevant?* The conventional view is that large droplets (~100 μm) sediment rapidly within 1-2 m from source [10,11]. This provides the rationale for guidance on physical distancing, a crucial non-pharmaceutical intervention. However, recent work has shown that large droplets can remain suspended in the turbulent air cloud from sneezes and coughs for much longer than previously thought travelling over larger distances (at least 7-8 m) [12,13]. Small respirable particles (<5-10 μm) are known/expected to transport over longer distances. Coughs and sneezes generate far more particles of respirable size than the large droplets that sediment out [14]. So, the arbitrary definition of airborne and droplet (contaminating surfaces as fomites) spread provides a poor representation of the problem. Some have also suggested that this should lead to revisions of guidelines on social distancing and the use of personal protective equipment (e.g. the importance of using respirator masks vs surgical masks).
- *How do interactions with background and urban PM impact on the airborne transmission of the virus?* One study has identified RNA from SARS-CoV2 internally mixed within urban PM [15]. However, the impacts of this on viability/infectivity and the mechanism and range of transport are unclear.

Using a novel instrument developed at the University of Bristol, we are now addressing questions surrounding airborne transport (viability, infectivity, RH dependence, temperature dependence, dependence on light/UV, droplet transmission range of droplets and aerosols in exhaled jets). The instrument is now housed in a containment level 3 laboratory and we are working with virologists at the University of Bristol with expertise in coronaviruses.

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Title:

Measuring the Impact of Covid-19 on Traffic flow and Air Quality in Leeds

Authors:

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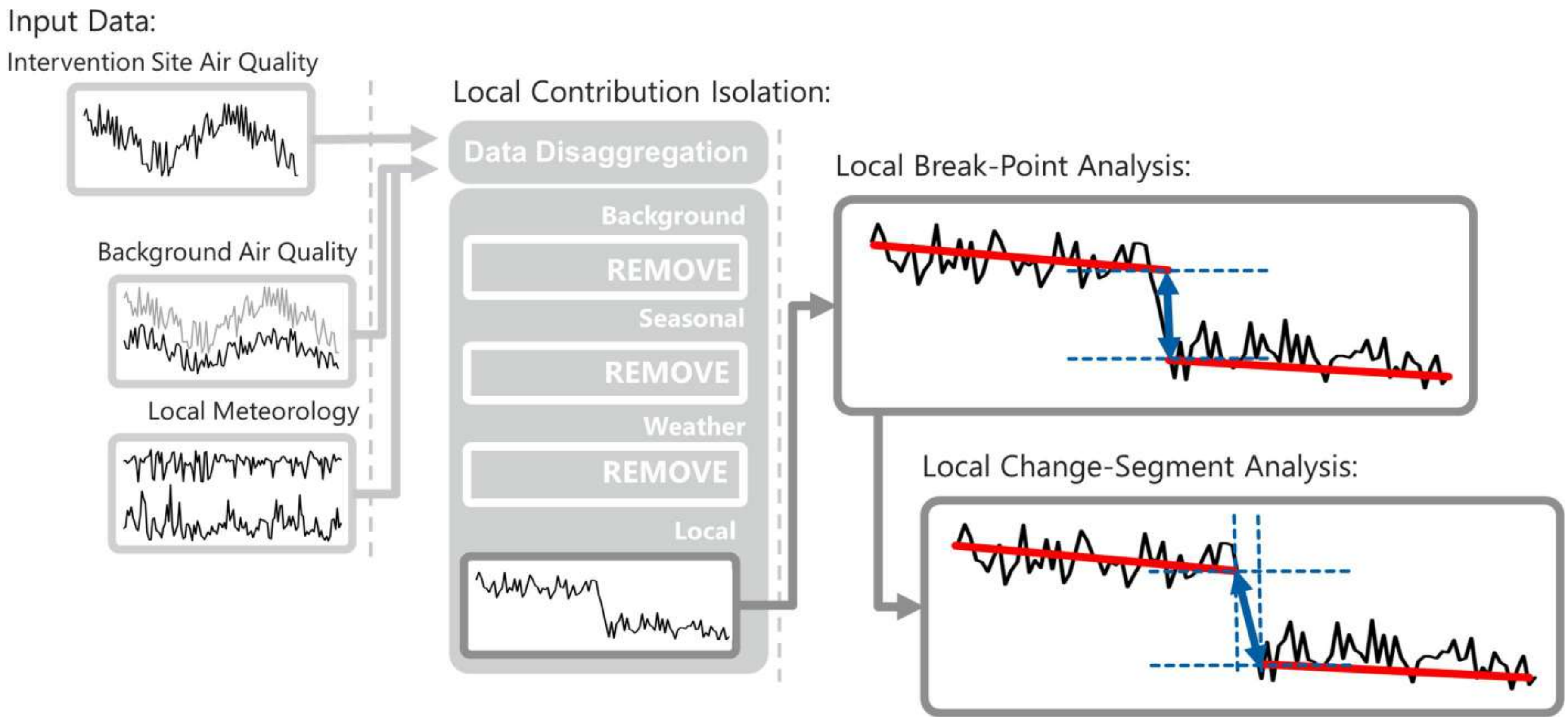
Highlights:

- The impact of traffic and air quality management plans post implementation are rarely evaluated
- The impact of the Covid-19 lock-down on both traffic flow and air quality are assessed
- Weather normalisation, break-point and change-segment detection methods are packaged and used in combination to detect impact of the Covid-19 restrictions on air quality concentrations
- Subtraction of a time varying (hour-by-hour) ambient background concentrations estimate increases the detection sensitivity
- The analysis is being routinely up-dated each week as part of a project supporting DEFRA assessing the impact of Local Air Quality Plans. The analysis will be replicated for other Local Authorities implementing Local Plans that have submitted NO_2 data from continuous analysers
- The analysis methods are coded into an 'AQEval' www.r-project.org package that is under development, which is expected to be released for 'user' testing in May.

Graphical Abstract:



1. Background:

Break-point detection methods test for points in a series of observations that are better explained, with greater statistical significance, by an abrupt change within the monitored system rather than chance, noise or underlying trends (Bai, 1997; Zeileis et al., 2003; Lee, 2010; Amiri & Allahyari, 2012). They have been widely used in many commercial and research areas, including several air quality applications (e.g. Carslaw et al., 2006; Carslaw & Carslaw, 2007; Barnett, 2012; Grange & Carslaw, 2019). Various signal isolation methods have also been used as a data 'clean-up' step prior to air quality data analyses. Background subtraction or correction methods have perhaps been most widely used, and provide a measure of local contributions or 'increments' (see e.g. Stedman et al., 2006; Visser et al., 2015; Sayegh et al., 2016; Basagaña et al., 2018.). Classical trend deconvolution methods such as 'deseasonalisation' assume there are regular frequency cycles in time-series, e.g. hour-of-day, day-of-week, and week-of-year cycles, and that modelling and subtracting these frequency patterns from time-series provides a clearer measure of underlying trends (Kendall & Stuart, 1983). Weather normalisation, often termed 'deweathering', extends this approach to the removal of more direct measures of the by modelling variance as a function of meteorological measures, such as wind speed and direction, air temperature and humidity (Grange & Carslaw, 2019). Conditional extraction (Malby et al., 2013), molecular tracers (Cass, 1998) and diagnostic ratios Watson et al, 2008; Tobiszewski & Namieśnik, 2012), amongst other methods, have also all been used to isolate source-specific contributions. In the few cases where such signal isolation methods have been applied in combination with break-point methods, improved sensitivity (Carslaw & Carslaw, 2007) and/or easier trend visualization (Grange & Carslaw, 2019) have been reported.

Here, a novel combination of local contribution isolation, break-point and change-segment analysis methods are applied to nitrogen dioxide (NO_2) air quality data from two Leeds real-time stations as part of an investigation of the potential environmental impact of a the Covid-19 "lock-down" restrictions. The break-point and change-segment analysis methods are also applied to continuous automatic traffic count data.

Traditionally, change detection methods have been applied to air quality applications in relative isolation: methods applied, results reported and interpreted on the basis of what was expected (e.g. seen elsewhere or predicted using modelling). However, there is a need to extend research efforts and investigate the likely performance of break-point methods, if we are to ask the authorities tasked with the delivery of air quality improvements to use methods to benchmark their efforts and inform future activities. With this in mind, data sources, data handling, method refinements and simulation testing strategies have been explored as part of this work to provide measures of both intervention impact and method performance, but are not included in this note for brevity.

2. Materials and Methods:

2.1. Data Sources:

Automatically collected traffic flow data is available from the Sheepscar (A58 – J0302) up until the 14th of April 2020 (download 20th). More recent data was available for a site on Headingley Lane (A660) until the 22nd of April. The near continuous 15-minute averaged data from Automatic Traffic Count (ATC) sites (two inductive loop per lane configuration) was accessed via the Drakewell C2-Cloud interface (https://www.drakewell.com/c2-web).

'Provisional' hourly NO_2 data is available from the two AURN sites in Leeds: Headingley roadside (A660 - UKA00527) and Leeds Centre (UKA00222).

The NO_2 data analysis in this note is up until the end of 19th of April 2020, therefore including the Covid-19 'lock-down' and Easter holiday period. Corresponding 'provisional' hourly NO_2 data from nearby rural AURN background sites around Leeds is used to estimate a time varying (hour-by-hour) background level (Glazebury, High Muffles, Ladybower and Market Harborough). These were selected as the nearest sites (all within 200 km of Headingley) that were classified as 'Rural Backgrounds' and had NO_2 data for the study time period, and accessed using R package 'openair' (R Core Team, 2019; Carslaw & Ropkins, 2012). The local background levels were estimated for Headingley roadside and Leeds Centre using 1-hour resolution maps extrapolated from available AURN background site data. Figure 1 illustrates the locations of the surrounding background stations, Leeds traffic and AURN sites.

Modelled meteorological data generated by the Ricardo WRF model (https://ee.ricardo.com/air-quality) and supplied with AURN data when downloaded using 'openair' function importAURN was also used in this analysis.

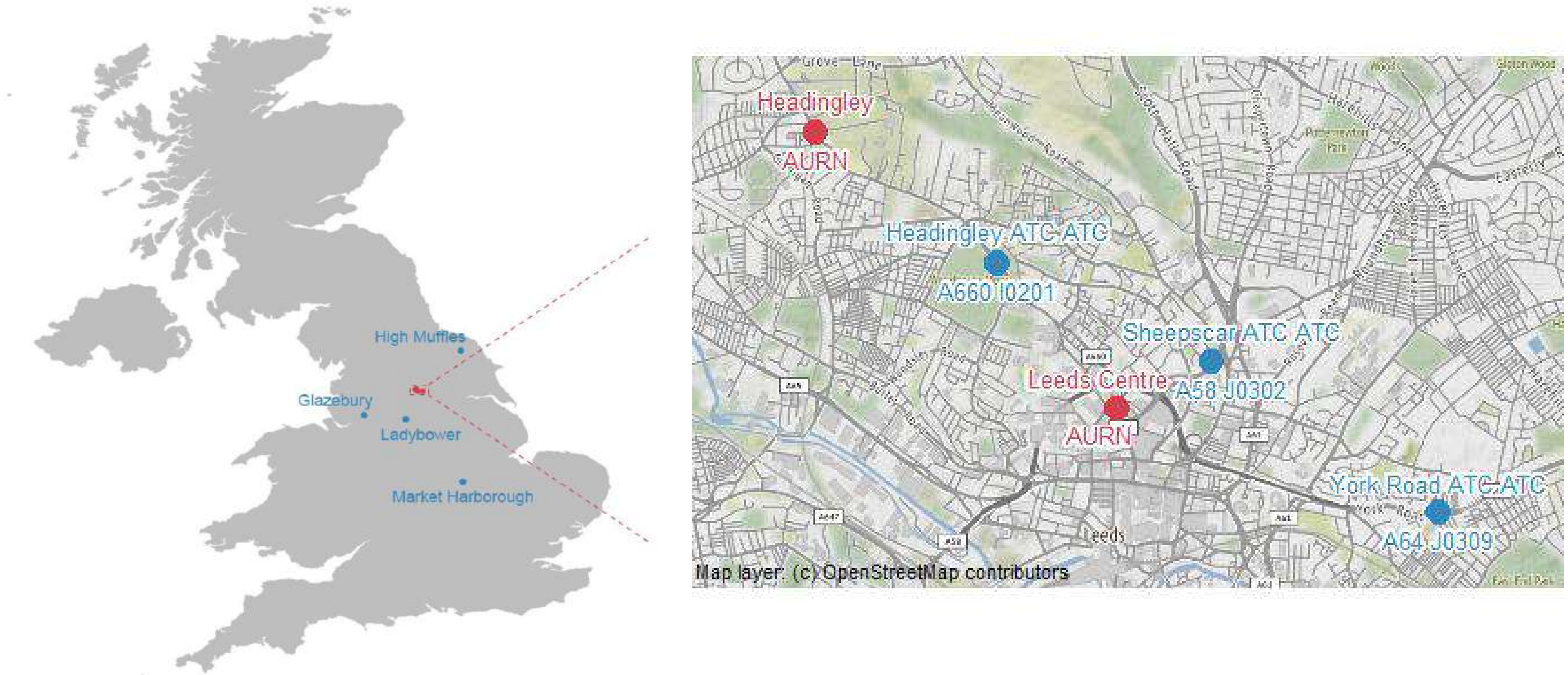


Figure 1: Locations of monitoring stations: UK map (left) with Automatic Urban and Rural Network (AURN) Rural Background sites used (in blue), on Leeds local map (right) with the Leeds AURN monitoring stations (in red). Leeds automatic traffic count (ATC) sites (in blue). (Maps tiles produced by Stamen Design, under CC BY 3.0. Data under ODbL using R package OpenStreetMap; Fellows & Stotz, 2019.)

2.2. Break-point Detection and Change-segment Estimation:

Change detection methods based on those of Zeileis et al. (2003) for use in non-static systems, and Bai and Perron (2003) for detecting multiple changes, as implemented in R package 'strucchange' (Zeileis et al., 2002), were used in this study. Here, a rolling window strategy is applied to test for changes in the linear regression properties of the investigated data-series. The associated hypothesis is that a change exists wherever the surrounding data is better explained by two discrete models rather than one general model, and significant changes, called break-points in the work of Zeileis and Colleagues, were assigned on the basis of statistical significance.

3. Results and Discussion:

3.1. Traffic Flow:

The Sheepscar ATC site is a core arterial connecting the North of Leeds with the City centre but also the Leeds urban motorway (A58M) and routes to the south including the M1 and M621/M62. It is a 2-3 lane bi-directional link. The Headingley Lane (A660) is a major arterial heading North-West from the City centre, the ATC site located on a isolated free-flowing stretch between the two AURN sites.

The simple time series in Figures 2 and 3 (a) for Q1 2020 illustrate flows at all three sites started to drop from the middle of March, then dropped further in the 3rd and 4th weeks of March. The remaining traffic flow is supporting essential services and key workers, including public transport operations (Bus services).

The change-segment detection methods identified the rise in traffic demand after the Christmas holiday period at the Headingley (A660) site, then substantial reductions in the third and fourth weeks of March when the UK was in Covid-19 "lock-down" (in Figures 2 and 3 (b)). The impact of the "lock-down" looks to be sustained through to the 22nd of April at the Headingley site (Figure 3).

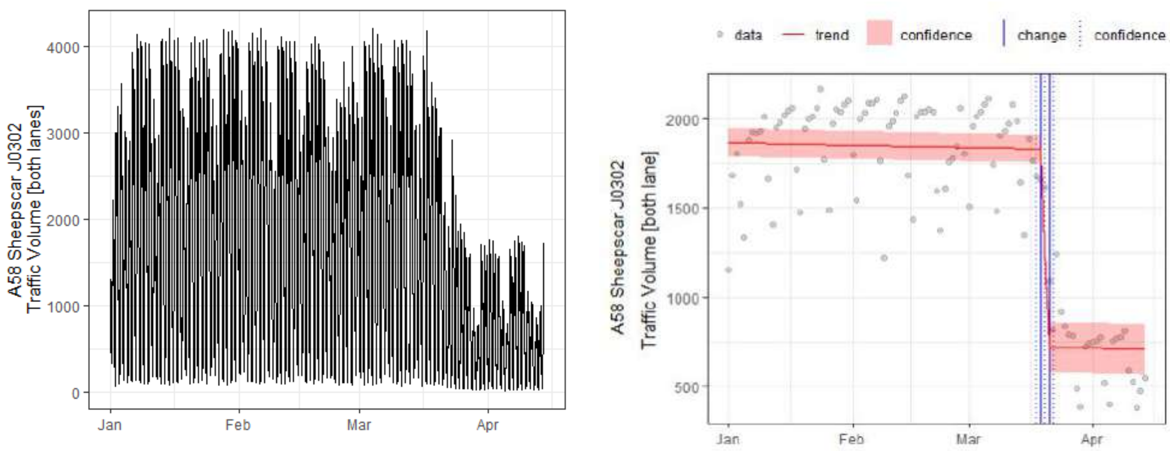


Figure 2. Sheepscar (A58) ATC Traffic Flow (a) Time Series [left-panel] (b) Change-segment detection

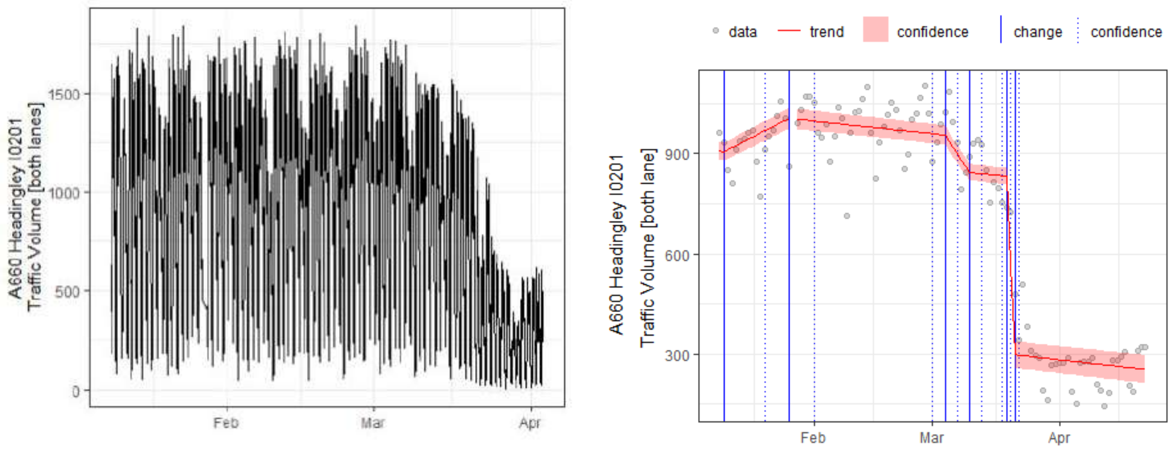


Figure 3. Headingley Lane (A660) ATC Traffic Flow (a) Time Series [left-panel] (b) Change-segment detection

3.2. Ambient Air Quality concentrations:

The local background NO_2 prediction using the rural background AURN sites surrounding Leeds (see Figure 1) is illustrated in Figure 4 (a) time-series and (b) with change-segment techniques. No discernible trend or change-points or –segments were identified. Change-segment techniques have been run on the resulting 'local NO_2 increment' we are terming 'provisional' due to the nature of background estimate and 'un-ratified' hourly data. The background estimate for NO_2 concentrations is stable for the 2020 data to date.

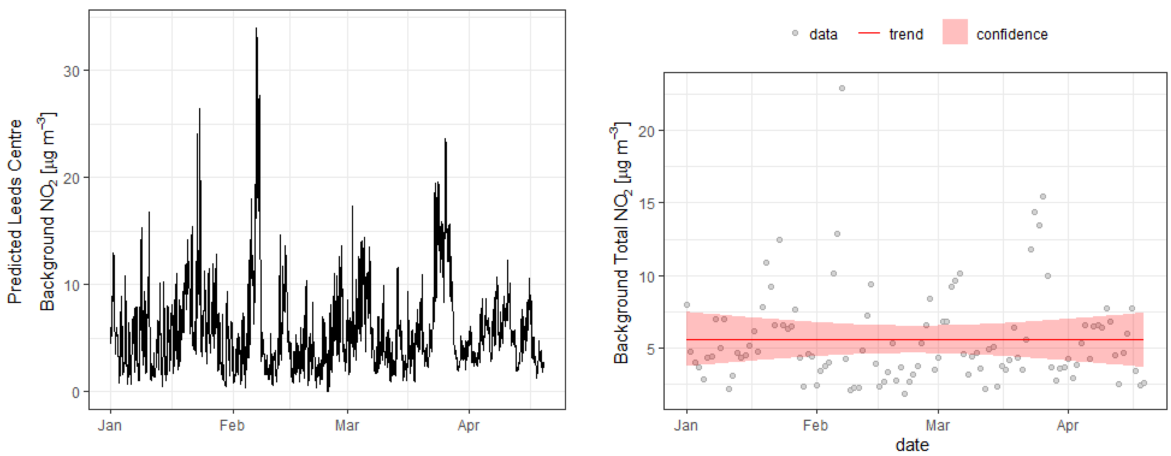


Figure 4. Leeds 'provisional' background estimate (a) Time Series [left-panel] (b) Change-segment detection (non-detected)

The Leeds Headingley roadside (A660) and Leeds Centre AURN provisional NO_2 data has been ‘de-seasonalised’ and ‘de-weathered’ using a long time series history. The change-segment analysis clearly illustrates the reduction in the local increment (traffic) levels from the start of March (Figures 5 and 6), falling to a nominal local contribution during the current Covid-19 ‘lock-down’ period at both sites. This is sustained through the Easter holiday period and the continued ‘lock-down’. The local NO_2 contribution at the Leeds Centre site is deemed higher, as being in closer proximity to the centre of Leeds, with a greater contribution of emissions from other sectors including construction and residential/commercial heating. These will raise the urban background concentrations above the rural estimate. Traffic activity is also suggested to be greater in the vicinity of the Leeds Centre AURN site as it is closer the Leeds General Infirmary teaching hospital (separation distance entrance 250m).

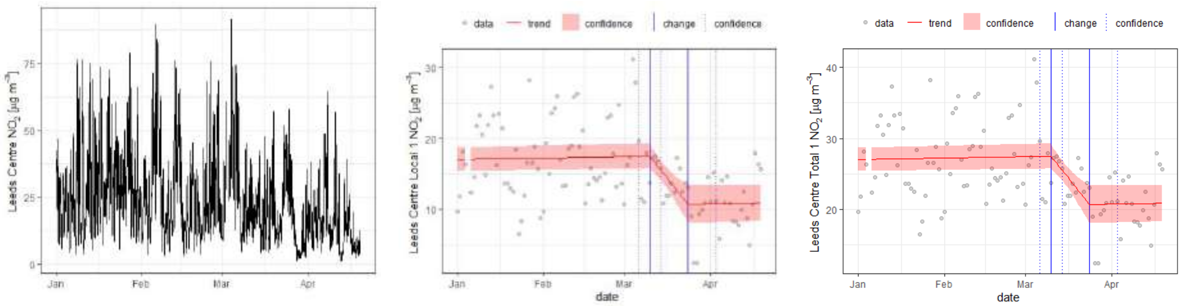


Figure 5. Leeds Centre AURN ‘provisional’ Local NO_2 (a) Time Series [left-panel] (b) Change-segment detection on the local increment (traffic) (c) Change-segment detection on the ambient concentrations.

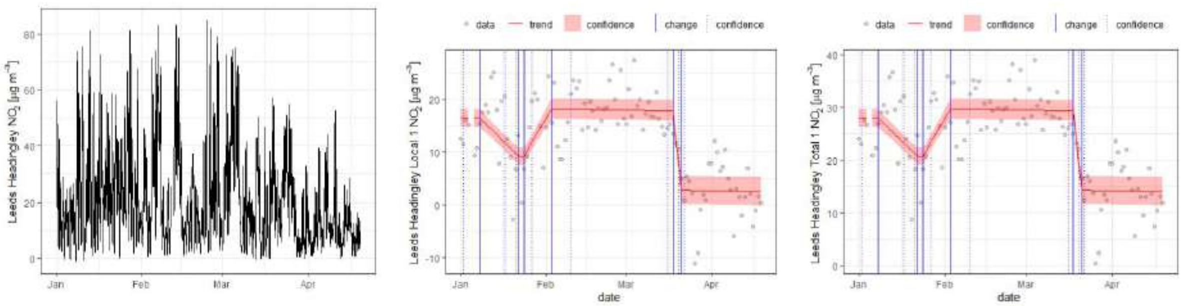


Figure 6. Leeds Headingley Roadside AURN ‘provisional’ Local NO_2 (a) Time Series [left-panel] (b) Change-segment detection on the local increment (traffic) (c) Change-segment detection on the ambient concentrations.

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Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK

CIEH submission to the Air Quality Expert Group at DEFRA

April 2020

About the Chartered Institute of Environmental Health (CIEH)

CIEH is the professional voice for environmental health representing over 7,000 members working in the public, private and third sectors, in 52 countries around the world. It ensures the highest standards of professional competence in its members, in the belief that through environmental health action people's health can be improved.

Environmental health has an important and unique contribution to make to improving public health and reducing health inequalities. CIEH campaigns to ensure that government policy addresses the needs of communities and business in achieving and maintaining improvements to health and health protection.

For more information visit www.cieh.org and follow CIEH on Twitter @The_CIEH.

Any enquiries about this response should be directed to:
Tamara Sandoul
Policy and Campaigns Manager
Chartered Institute of Environmental Health
Email: t.sandoul@cieh.org

Key points

We would expect to see a reduction in air pollution, mostly due to the reduced road traffic associated with people working from home and not doing school runs. However, concentrations later in the year will depend on how the restrictions are eased by the Government.

The Summer holiday season will be a time when emissions could rise again as people take holidays in the UK rather than going abroad. Use of barbeques and fire pits in the warmer months may also contribute to localised pollution and exposure.

All areas we received information from reported an increase in complaints about bonfires. This will have an impact on the local exposure of potentially vulnerable population during lockdown.

We have also been told that areas are seeing increases in ozone levels in the air due to reductions in NOx concentrations in the air, which usually 'mops up' ozone particles.

About this submission

We have received contributions from environmental health teams at the following local authorities. We have used these contributions as the basis for our response and can provide more detailed information if required.

Adur & Worthing Councils
BCP council
City of London
Dartford & Sevenoaks Councils
East Cambridgeshire District Council
East Northamptonshire Council
Exeter City Council
Harborough District Council
Harrogate Borough Council
Herefordshire Council
Milton Keynes Council
North Devon Council
Sedgemoor District Council
Shropshire Council
Southend on Sea Borough Council
Stafford Borough Council
Wakefield Council

1. What sectors or areas of socioeconomic activity do you anticipate will show a decrease in air pollution emissions, and by how much?

We expect the dramatic reductions in road transport and traffic to translate into reductions in air pollution, particularly NO_x. The absence of school runs and the drops in the numbers of people commuting in relation to the lockdown are key drivers. Emissions from commercial buildings and industrial premises that are temporarily closed are also expected to contribute to reductions to air pollution concentrations but it is difficult to quantify these accurately. However, teams suspect that one sector that has been relatively unaffected, in terms of air pollution generation, has been agriculture and power generation. The NHS is also not expected to have reduced pollution for obvious reasons.

Some teams have told us that large drops in pollution levels measured by diffusion tubes locally. However, others cited problems with continuing measurements locally, owing to the closure of laboratories processing the tubes.

The data submitted below is meant to give an indication of the direction that air pollution is moving in across different sectors, however it should be used with some caution as these are estimates only.

Specific changes observed from local areas:

- In East Cambridgeshire DC, diffusion tube monitoring showed NO_2 concentrations in March were up to 30% lower than in February in some cases and substantially lower than in March 2019. Overall decrease of around 20% is expected.
- In Southend Council, data from passive diffusion tubes for March 2020 has shown a decrease in NO_2 of between 4-50% compared to March 2019. Commercial data from the council also confirms drops in activity across most sectors, which are expected to contribute to drops in emission of pollutants.
- In Stafford BC, emissions of solvents from industrial production are declining locally by around 30%. Emissions from commercial and public buildings, due to reduced heating and power use, are likely to represent a reduction of around 25% compared to equivalent times last year because of the lockdown. Transport emissions appear to be declining by around 25% estimated from road traffic reduction, mineral production (sand and gravel) appears to have reduced as demand has fallen by about 50%.
- In Herefordshire Council, recent monitoring indicated that NO_2 levels were reduced by 20% during the initial lockdown period.
- In Wakefield Council, data from a continuous monitoring station in Wakefield City centre shows that the diurnal peaks are still present but are reduced and fall away to significantly lower levels outside times of peak traffic flow when compared to a week prior to lockdown in the UK.

2. Are there any emissions sources or sectors which might be anticipated to lead to an increase in emissions in the next three months?

Whilst most respondents have told us that the overall trend will be a reduction in air pollution, there will be industries and sources of pollution that are likely to be increasing their emissions during the lockdown. In particular, some areas told us that *"Ozone and PM episodes as witnessed during the recent Easter weekend are locally significant from a public health viewpoint."* Ozone increases are associated with the drops in NO_x pollutant concentrations in the air due to reduced road traffic.

Activities contributing to increases:

- Medical waste incineration
- HGVs and home deliveries
- Domestic bonfires (garden and other), bonfires on building sites and burning of fly-tipped waste
- Solid fuel burning, depending on the weather
- Use of crematoriums in Boroughs
- Increase in traffic to hospitals, including newly built Nightingale hospitals

Bonfires

When gathering information for this submission, we specifically asked about increases in complaints about bonfires, as our previous research on noise and nuisance revealed an increase across almost every area. It should be noted that number of complaints does not necessarily correlate with number of fires. There may be the same number of fires this year but more people exposed to the effects due to lockdown so more complaints are received. Equally there could be ten times as many fires this year but people are tolerating it and not complaining as they appreciate people need to dispose of waste in some other way than visiting recycling centres.

Specifically, many respondents called for national press releases by DEFRA calling on the public not to have bonfires and to save their waste until recycling centres reopen and to compost garden waste instead. Local areas are already trying to raise awareness locally but feel that an amplified message would be a huge help. Promoting the message around being considerate to neighbours would be helpful as smoke from bonfires could have an impact on neighbours suffering from COVID19 or other respiratory conditions.

As well as garden bonfires, the mild/warm weather in the weeks to come may prompt people to use fire pits and barbeques in their gardens, contributing to particulate pollution locally.

Below is a summary of responses we received in relation to bonfires:

- Sedgemoor DC - A massive increase in bonfires complaints. 5 reports in April 2019, compared to 28 reports in April 2020.

- East Cambridgeshire – An increase, especially for April. For 2019, we recorded 5 bonfires in Feb, 4 in March and 5 in April. For 2020, we recorded 4 bonfires in Feb, 3 in March and 9 in April (up to 21/4/2020).
- East Northamptonshire Council have had an increase in bonfire complaints:

Month	2020	2019
March	4	3
April (to 21 April)	11	3

- Exeter - We have seen a large increase in complaints about bonfires and smoke. Since March 26th we have received 23 complaints, compared to just 3 in the same period last year.
- In Southend, complaints about bonfires have doubled in the first weeks of April. During 1-21 April 2019, number of complaints about bonfires was 8, whilst for the same dates this year it was 20.
- Complaints to Wakefield Council of smoke nuisance have more than doubled when compared to the same period in 2019.
- Stafford reported an increase particularly as garden waste collections have halted.

	23-29 March	30 March – 5 April	6-12 April	13-19 April	TOTALS
2019	0	0	0	2	2
2020	1	1	0	4	6

- Harborough DC also saw an increase during March and April:

Year	**February**	**March**	**April**
Y2017	1	4	1
Y2018	2	1	0
Y2019	2	3	1
Y2020	2	7	9

- Milton Keynes reported that from 16 Mar - 21 Apr 2019 they received 24 bonfire complaints and for the same period this year the number was 60.
- Adur & Worthing reported a marked increase. The number of complaints has reduced slightly since last week, since they put out messages to residents not to burn waste.

	January	February	March	April
2020	8	2	10	39
2019	3	6	4	7

- Harrogate has reported a tripling of complaints received on bonfires. Between 1 March – 20 April 2019 there were only 5 complaints about bonfires, the same period this year, we have received 14 complaints.
- North Devon reported a 30% increase in bonfire complaints compared to same period in recent years although they acknowledged that there are other factors which could influence this, such as when Easter falls and the weather.

- Bournemouth, Christchurch and Poole - With regards to bonfires, they have experienced a significant rise in complaints with more people stuck at home and the green waste service being stopped, although this is starting up again very shortly. "Across Bournemouth, Christchurch and Poole between 15/3/20 and 15/4/20, we've had 114 domestic smoke complaints compared to 33 during the same period last year, so a significant increase, but we are hopeful the reopening of the green waste service will help reduce this."
- Shropshire also reported more bonfire complaints - both domestic and industrial. "We have had a slight increase in complaints of fires in lock down compared to the same three weeks of dates last year (24 compared to 16). However, it should be noted that last year the Easter break was late and therefore some fires after bank holiday clearances and gardening may have come later last year… We are still receiving complaints."

3. What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?

All the practitioners we heard from anticipated reductions in air quality indoors. This is due to more people spending more time indoors, people doing more cooking, cleaning, home decorating and undertaking DIY work. This is likely to vary between individual households and will likely be dependent on socio-economic factors. For example, small dwellings with larger numbers of residents may see an increase in indoor pollution as a result of human activities such as cleaning. There may be more exposure to particulates and volatile organic compounds from more home cooking. Household fuel use will have a significant impact, with solid fuels producing worse indoor air quality than gas or electric forms of heating. Households with smokers are likely to be the most impacted. If restrictions continue or are repeated in winter months, indoor air quality will be more of a problem.

However, better and warmer weather means windows tend to be open, which mitigate against poor air quality. Furthermore, it is possible that as outdoor pollution levels fall in urban areas, indoor pollution levels may also improve.

4. How might public exposure to air pollution have changed as a consequence of recent restrictions on movement?

Overall, most practitioners responding to our request for information expected the overall exposure to air pollution to be lower than a similar time last year, due to the reasons cited in response to questions 1 and 2. Many people are not commuting to work. As a result, the exposure in vehicles and on buses would be reduced. Even for those who are still travelling, the exposure should be slightly lower, given that traffic levels are lower on the roads. Taking exercise outdoors may also be associated with lower exposure due to reduced traffic levels. Exposure to air pollution in the home is likely to be higher but as long as the weather stays warm and windows are opened, the net effect should be less exposure overall.

However, the above summary is a very general trend. This will vary between individuals, locations and households. Some who would ordinarily commute and work in more polluted environments will see a benefit. Others will have less access to 'fresh air' and may be impacted by increases in indoor air pollution. Children living with smokers are likely to be the most impacted. Furthermore, as a result of increased bonfires, on a micro scale, some individuals will have increased exposure.

5. How might altered emissions of air pollutants over the next three months affect UK summertime air quality?

If the lockdown continues in its current form, this should lead to an overall improvement in UK summertime air quality. The emissions of pollutants will depend on how long lockdown lasts and how quickly it is eased and for which sectors. If restrictions are relaxed but many people may continue to work from home, the improvement in air quality would last longer. However, if and when we get back "to normal" we could see a re-surge in emissions.

It is important to look forward to summertime and to ask what will happen to people's holiday plans. At the moment, it seems likely that foreign travel will remain restricted for some time meaning continuing reductions in emissions from air traffic. However, if restrictions to movement within the UK are eased, it is likely that day trips, short breaks and holidays within the UK would increase, possibly leading to an increase in emissions from road vehicles. As a result, it is possible that air quality in rural areas may be adversely affected with people heading for the coast and to national parks.

Most of our respondents would expect to see a reduction of particles and combustion gasses from traffic, but an increase in ozone in urban areas. Increased summertime ozone episodes are expected due to decreased traffic emissions, which would normally locally scavenge (reduce) exhaust emissions. Ground level ozone related to crop production is also likely to remain the same as for other years, along with all agricultural emissions.

Ozone and PM episodes originating from the continent will continue to be a public health issue whatever happens in the UK, so it will be important to consider what neighbouring countries will be doing and how quickly they ease restrictions. The weather will also play a part.

Modelling dilution of exhaled breath using ADMS

David Carruthers, James O'Neil and Martin Seaton, CERC, April 28 2020

This short note is in response to the AQEG call 'Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK' and in particular the following item:

Are there any insights that can be gained from aerosol science on possible viral transmission mechanisms?

Introduction

Whilst 2m has become the separation distance beyond which it is implied that there is little danger of a person being infected by a sufficient number of virus particles to contract COVID-19, there has been little if any discussion of the appropriateness or otherwise of this distance. This is examined in an idealised way using the ADMS5 dispersion model.

Whilst transmission in air can be by coughing or sneezing the focus here is on regular breathing for which the virus particles are small enough that settling can be ignored. Simple assumptions are made that 'breathing out' is at a height of 1.5m through a mouth diameter of 1.5cm [1] at a rate of 0.5 litre over a 2 second period [2]. Since the magnitude of the source of virus particles is unknown, we examine the fractional reduction in the dose relative to the dose at the source at 2m downwind of (and at the same height as) the source, for a range of air flow velocities from the source to the receptor for different turbulence levels. The model is able to calculate both the ensemble mean dose and fluctuations about this mean over a large number of breaths; for the example calculations below we show the mean and the 95th percentile of the dose.

Results

Table 1 shows the ratio of the dose at source to the dose at 2m from the source for a range of mean flow and turbulence levels relevant to both outdoor and indoor conditions. Ratios for both the mean dose and the 95th percentile are shown for each combination of mean flow and turbulence. Noting that the lower values represent lower dilutions and therefore higher doses we see that the highest doses occur for the highest mean airflow speed and lowest turbulence levels – initial dilution at source due to higher airflow having less impact than the low levels of subsequent mixing due to short transport time and low turbulence. Conversely low airflow speed and high turbulence levels result in much higher rates of dilution and lower doses (over 200 times lower than the worst case shown). The 95th percentile dilution ranges from about 20% to 50% of the mean dilution.

Conclusion

These idealised model runs do not describe either the breathing process or the complex flows that occur close to the mouth or nose, however the very large sensitivity of the dilution at 2m distance to ambient airflow and turbulence is a strong indication that the dose at 2m is highly dependent on ambient conditions both indoor and outdoor.

Table 1. Ratios of dose at source to dose 2m ‘downwind’ from source at the height of the source (1.5m) for a range of ambient mean airflow (U) and turbulence levels (σ). For each U, σ combination both the mean (left) and 95th percentile (right) dilutions are shown.

		U (m/s)					
		0.3		**0.9**		**3**	
$\sigma_{u,v,w}$ (m/s)	**0.1**	1888	*383*	821	*211*	388	*120*
	0.3	11780	*2949*	4970	*1529*	1737	*776*
	1	83250	*21440*	40760	*13100*	15680	*7557*

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Modelled impact of COVID-19 restrictions on pollutant emissions and summertime air quality in the UK
Submission to AQEG call for evidence

Atmospheric Dispersion and Air Quality team, Met Office
Contributing authors: P. Agnew, V.B. Bright, K. Coward, B. Drummond, M. C. Hort, F. Malavelle, P. Molina-Jimenez, N. Nelson, B. Sherratt, E. Smith

1. Introduction

The Air Quality Expert Group (AQEG) has issued a call for evidence on a variety of topics and questions, including:

- Estimates for how emissions and ambient concentrations of NOx, NO_2, PM, O_3, VOC, NH_3, etc may have changed since the COVID-19 outbreak.
- How might altered emissions of air pollutants over the next three months affect UK summertime air quality?

Since the end of March, the Met Office have undertaken studies directed at these two questions in order to inform the impacts of COVID-19 restrictions on pollution prediction for the national air quality forecast. This short report summarises our initial findings.

2. NOx Emissions

The problem of estimating changes to pollutant emissions is a challenging one as there are no direct measurements available: inferences can only be made from measured air pollution concentrations. A major complication involves separating the effects of meteorological variations from emissions changes. Disentangling the effects of weather variations from changes in emissions can be addressed via approaches ranging in sophistication from full inversion modelling to simple approaches based on analytical Gaussian plume considerations. Our initial work in this area has used the latter approach to analyse changes in the quantity *cU* as a proxy for emissions - where *c* denotes the measured pollutant air concentration and *U* the mean wind speed (derived from the air quality model AQUM). We have estimated the fractional change in NOx emissions via:

$$\frac{\Delta NO_{x\ emissions}}{NO_{x\ emissions}} \approx \frac{(\overline{cU}) - (\overline{cU})_{ref}}{(\overline{cU})_{ref}}$$

where the average (denoted by overbars) is taken over all times in a given period (see Table 1) and '*ref*' denotes the pre-lockdown reference period, indicated below:

Table 1. Time averaging periods used in the estimation of emissions changes. Partial lockdown refers to the week before the official restrictions were implemented, but when responses including working from home were encouraged.

Pre Lockdown	Monday 9th March 2020 00:00 to Monday 16th March 2020 00:00
Partial Lockdown	Monday 16th March 2020 00:00 to Monday 23rd March 2020 00:00
Week 1 Lockdown	Monday 23rd March 2020 00:00 to Monday 30th March 2020 00:00
Week 2 Lockdown	Monday 30th March 2020 00:00 to Monday 6th April 2020 00:00
Week 3 Lockdown	Monday 6th April 2020 00:00 to Monday 13th April 2020 00:00
Week 4 Lockdown	Monday 13th April 2020 00:00 to Monday 20th April 2020 00:00
Week 5 Lockdown	Monday 20th April 2020 00:00 to Monday 27th April 2020 00:00

The results are shown below in Figure 1.

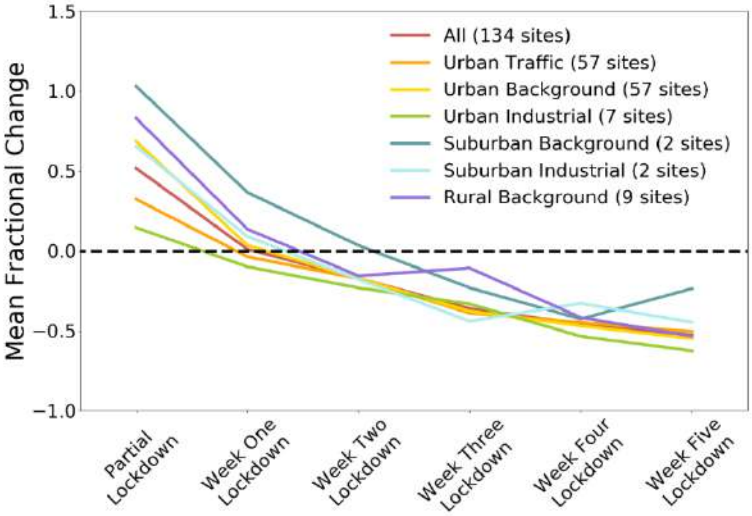

Figure 1. Fractional change in emissions of NOx estimated from AURN observations of NO_2 air concentrations and model wind speeds. The number of individual sites contributing to each mean dataset are indicated in the legend.

The simple approach we have used is most reliable for urban traffic sites, where the close proximity of source and receptor ensures a weak dependence on wind direction and meteorology in general. However the general pattern found in our results is consistent for all site types. A similar analysis based only on sites having both NO_2 and NO measurements, to allow the use of NOx rather than NO_2, gives comparable results. The apparent increase during the partial lockdown period (i.e. positive mean fractional change) is not yet fully understood. The results for Suburban Background, Suburban Industrial, Urban Industrial and Rural Background are only based on a small number of sites and are not robust; but for Urban Traffic and Urban Background sites it may reflect a limitation of this simplified approach. The reference (pre-lockdown) week was generally quite windy over the UK compared to the following period of generally light winds and settled weather which continued throughout April, and this may accentuate the shortcomings of this analysis method.

3. Impacts on spring and summertime air quality

Two impacts of the restrictions on spring and summertime air quality have been identified:

1. The possibility of altered ozone levels due to reductions in NOx and NMVOC emissions.
2. Changes to PM due to both reduced primary emissions and formation of secondary inorganic aerosol due to reduced emissions of NOx (and to a lesser extent SO_2).

To investigate these risks, we have performed several simulations using modified emission scenarios. The initial simulations involved running the AQUM model using adjusted emissions for 1st March - 30th April 2019. A limitation of the simulations is that pollutant lateral boundary conditions were left unchanged from those of the historical period, using values generated by the C-IFS global model operated by ECMWF using unmodified emissions. The main objective therefore was to investigate the reductions in local pollutants due to in-domain primary emissions and the modified production tendencies for secondary pollutants.

The emissions were informed by the following plots (Fig. 2) showing the distributions of NOx and NMVOC emissions across the Selected Nomenclature for source of Air Pollution (SNAP) sectors:

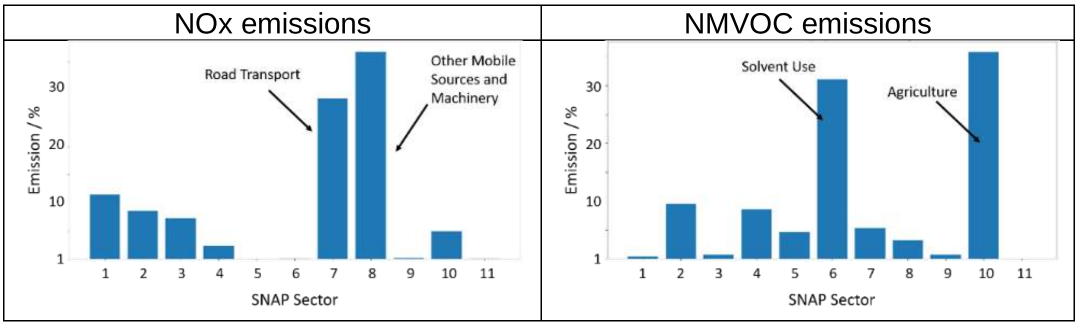

Figure 2. Percentage of NOx (left) and NMVOC (right) emissions in each SNAP sector 1-11 for April.

In the absence of detailed information about the reduction factors in each sector we opted to apply a fixed factor across all sectors as follows:

Table 2. Emissions scenarios

Scenario ID	Species of interest	NOx	SO_2	Primary PM*	NMVOC
1A (Control)	O_3	1	1	1	1
1B	O_3	0.7	1	1	1
1C	O_3	0.5	1	1	1
1D	O_3	0.3	1	1	1
2B	O_3	0.7	1	1	0.7
2C	O_3	0.5	1	1	0.7
2D	O_3	0.3	1	1	0.7
3B	PM	0.5	0.7	0.7	1
3C	PM	0.3	0.5	0.5	1

*The 'Primary PM' is actually a subset of the total primary component, excluding organic matter, but accounts for the majority of the total primary emissions.

3.1 Ozone Impacts

3.1.1 Impact on springtime ozone

A comparison of scenarios 1A-D over the period March-April 2019 gives the following changes in model ozone mean and maximum.

Table 3. Impact on model ozone of NOx reductions during springtime conditions.

NOx reduction factor (Scenario)	Model Mean O_3 ($\mu g/m^3$)	Model Max O_3 ($\mu g/m^3$)
1 (1A control)	69	128
0.7 (1B)	74	133
0.5 (1C)	76	151
0.3 (1D)	78	151

This case shows a significant rise ~ 23 μgm^{-3} in the maximum ozone during the period. Closer examination reveals this occurs over London, as shown in the following comparison of the ozone Daily Air Quality Index (DAQI) on 17th April 2019. This main impact here appears to be suppression of the 'urban decrement' to ozone due to reduced NOx emissions, allowing increases in peak ozone levels. If NMVOC reductions are also considered, then peak ozone values are not increased as much as when only NOx reductions are considered.

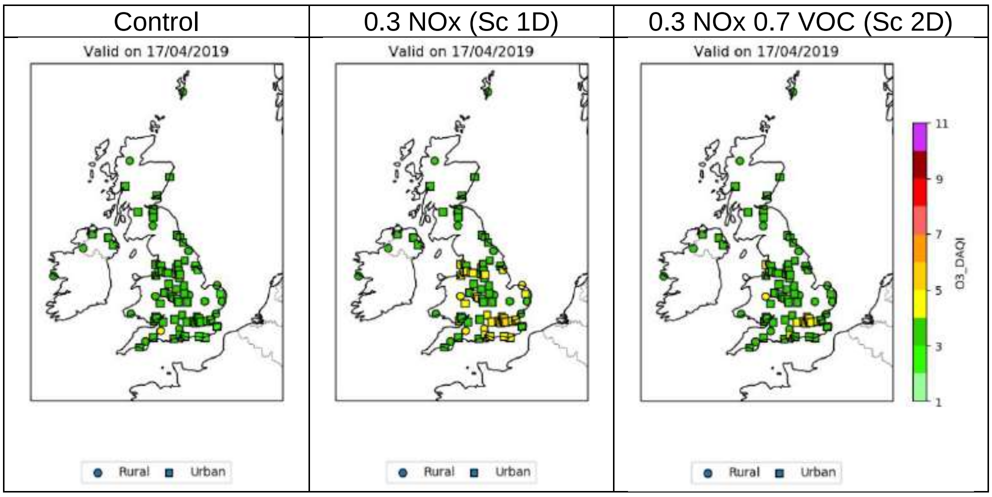

Figure 3. Comparison of ozone DAQI: control, 0.3NOx (1D) and 0.3NOx + 0.7NMVOC (2D).

3.1.2 Impact on summertime Ozone episode: 22 – 30th August 2019

Table 4. Impact on model ozone of NOx reductions for summer episode conditions.

NOx reduction factor (Scenario)	**Model Mean O_3 ($\mu g/m^3$)**	**Model Max O_3 ($\mu g/m^3$)**
1 (1A control)	62	135
0.7 (1B)	66	148
0.5 (1C)	69	171
0.3 (1D)	70	198

In these simulations the potential for a large increase in peak ozone values under reduced NOx emissions is again apparent. Figure 4 show the simulated ozone fields for the control and 1D scenario:

Control 0.3 NOx (1D)

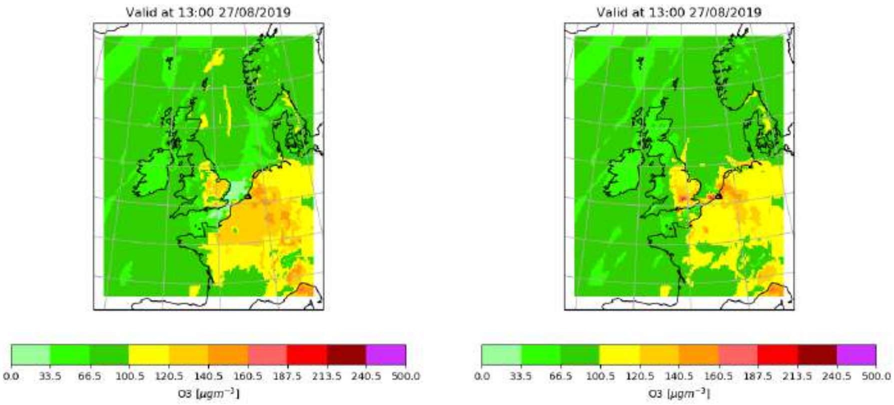

Figure 4. Control and 0.3NOx (1D) ozone fields at the peak of August 2019 summertime ozone episode.

This case again demonstrates the higher peak ozone values which can occur when the supressing effect of urban NO emissions are reduced. However it should be noted that this elevation of ozone over London is not a consistent feature of every episode. Figure 5 shows peak ozone during an episode in June 2017. In this episode no increase in ozone is apparent over London in the low NOx scenario.

0.3 NOx (1D)

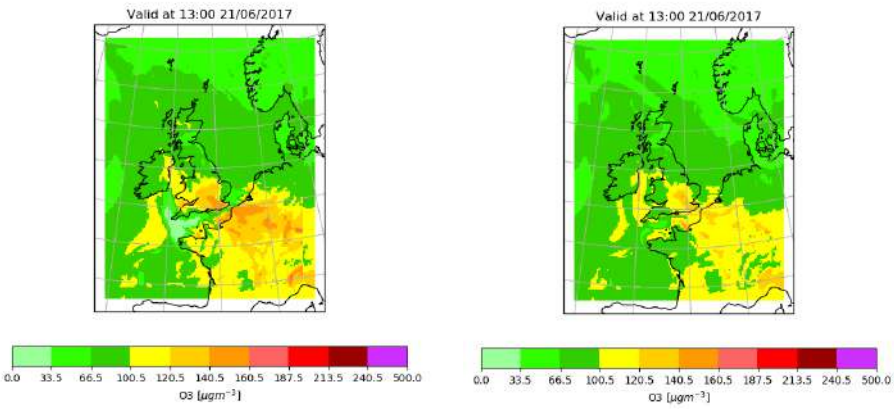

Figure 5. Control and 0.3NOx (1D) ozone fields at the peak of June 2017 summertime ozone episode.

3.2 PM Impacts

The second area of interest identified is the impact of COVID-19 restrictions on PM levels. Any reduction in primary emissions causes a direct reduction in measured PM of a magnitude less than or equal to the reduction in emissions. The impact on secondary aerosol is more complex. Under episode conditions, secondary nitrate (and to a lesser extent sulphate) aerosol forms a major component of the total $PM_{2.5}$ burden and reduced emissions of the precursor gases (NOx and SO_2) should lead to much lower aerosol formation. We sought to quantify the impact of reduced NOx emissions on the formation of secondary ammonium nitrate aerosol. We define a parameter α according to:

$$\frac{\Delta PM_{2.5\ nitrate}}{PM_{2.5\ nitrate}} = \alpha \frac{\Delta NO_{x\ emissions}}{NO_{x\ emissions}}$$

This parameter is a complex function of atmospheric parameters & processes, as such it can be expected to have a range of values. The main objective of this study was to determine the distribution of α over the range of conditions. Analysis of results over the period March-April 2019 gives the following distribution (Fig 6) for α

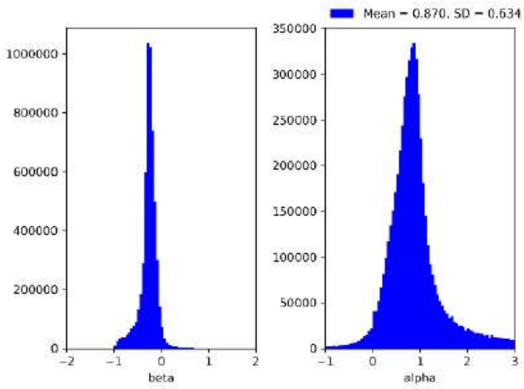

Figure 6. Left-hand plot: Distribution of the fractional change in nitrate aerosol concentrations (beta). Right-hand plot: Distribution of the parameter α, relating fractional change in the nitrate aerosol component of $PM_{2.5}$ to the fractional reduction in NOx emissions. The sample is made up of individual calculations of α for each grid box and each timepoint in the air quality model UK domain

The mean of the distribution of α is ~0.87 meaning that, on average, reductions in NOx emissions lead to a similar relative decrease in nitrate aerosol concentrations. Further analysis shows that α:

- exhibits little diurnal variation (somewhat unexpected based on the differences in day/night NOx chemistry)
- has a mean value which remains fairly constant in episode conditions, in the range ~0.85 - 0.88
- generally takes smaller values where nitrate aerosol air concentrations are higher.

The impact of reducing NOx emissions on total $PM_{2.5}$ levels is therefore significant under all conditions, with the greatest magnitude in reductions occurring under episode conditions.

The following tables show the impacts on total $PM_{2.5}$ levels comparing scenarios 3B and 3C with the control run 1A over two periods.

Table 5. Impact on model total $PM_{2.5}$ during March-April 2019 simulation period.

Scenario	**Model Mean PM2.5 ($\mu g/m^3$)**	**Model Max PM2.5 ($\mu g/m^3$)**
1A (control)	16	131
3B	13	121
3C	10	92

Table 6. Impact on model total $PM_{2.5}$ during 4th – 13th April 2020. This is a period of strong episode conditions, where nitrate aerosol was the dominant component of total $PM_{2.5}$.

Scenario	**Model Mean $PM_{2.5}$ ($\mu g/m^3$)**	**Model Max $PM_{2.5}$ ($\mu g/m^3$)**
1A (control)	42	163
3B	31	127
3C	24	85

Based on these results Scenario 3B was selected for implementation in the operational Met Office air quality forecast on 7th April. Subsequent verification confirmed the importance of making these emissions modifications in the forecast model.

4. Summary

1. A simple model using the product of mean wind speed and observed NO_2 air concentrations as a proxy for NOx emissions provides further evidence of the reductions in the latter as a result of COVID-19 restrictions.
2. The main impacts on the DAQI arise from changes to ozone and particulate matter due to reduced NOx emissions.
3. A key impact on ozone is the potential for significantly increased peak values, especially over Greater London, during *some* summertime episodes, as a result of the reduced supressing effects of emitted NO.
4. Reductions in primary particulate emissions and NOx lead to a general reduction in $PM_{2.5}$ levels.
5. The impact of reduced NOx emissions, leading to reduced production of secondary nitrate aerosol, gives a major reduction in $PM_{2.5}$ levels under episode conditions.

Short summary for AQEG: What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?

Nicola Carslaw, University of York

Introduction

Based on reports (from David!!) that ambient NO_X has decreased by ~40% and O_3 has increased by 30% in urban areas, I am interested in the effects this could have on indoor air chemistry. This is because indoor air chemistry is driven by ozone (see Weschler and Carslaw, 2019). Therefore, if ozone increases outdoors, it will also increase indoors, particularly given that the recent weather will have encouraged people to open windows more frequently.

The increased time spent indoors as a result of lockdown and the reasons why, mean that there is likely to be more use of cleaning and related hygiene products in the home. We are also likely to be cooking in our homes more than usual and cooking and cleaning both produce pollutants indoors that are well known to impact health (e.g. formaldehyde (HCHO), particulate matter (PM)). So potentially, there could be increased concentrations of these secondary pollutants indoors and longer exposures to them.

Methodology

The INDCM (Indoor Detailed Chemical Model) has been developed over the last 15 years or so. It combines a detailed chemical mechanism with terms that consider indoor/outdoor exchange of air, surface deposition and emissions, and photolysis reactions (both indoor lighting and attenuated sunlight through windows) and has been described in detail in the literature (e.g. Carslaw, 2007; Carslaw et al., 2012). The model can be set up to represent an indoor space of interest (e.g. office, home, classroom) and for a range of outdoor pollutant concentrations.

I have carried out a run with the model for a typical UK residence and assuming that outdoor O_3 has increased by 30% and outdoor NO and NO_2 reduced by 35% compared to pre-lockdown conditions. I have then simulated cleaning in the afternoon for lockdown conditions and compared it to the pre-lockdown conditions. I have assumed 10-minute use of a limonene based cleaner at 4 PM, such that the limonene concentration reaches 300 ppb. I have then looked at the difference in O_3, PM and HCHO (figures 1-3) concentrations pre- and during lockdown.

Results and Discussion

Figure 1 shows that despite outdoor ozone concentrations increasing by only 30%, indoor concentrations increase by closer to 50%. This is because there is an added impact of reduced outdoor (and hence indoor) NO_X concentrations. Figure 1 shows the removal of O_3 at 4 PM by the limonene in the cleaning product, followed by recovery afterwards.

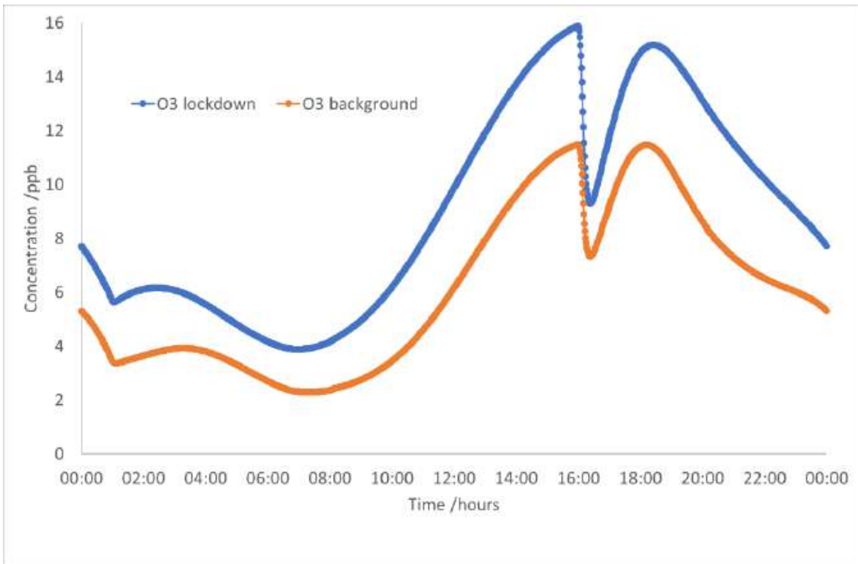

Figure 1: Ozone concentrations for a typical UK residence pre- and during lockdown.

The chemistry that ensues between ozone and limonene then leads to the production of PM and HCHO shown in figures 2 and 3 respectively.

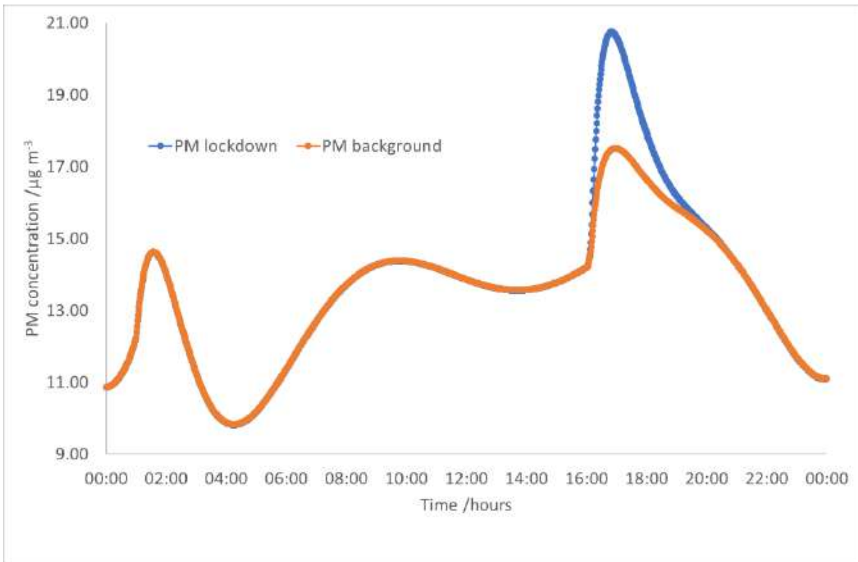

Figure 2: PM concentrations for a typical UK residence pre- and during lockdown.

Figure 2 shows that PM concentrations only really change during cleaning. Outdoor PM concentrations have been assumed to stay the same, as there does not appear to be evidence they have changed significantly. The cleaning activity at 4 PM forms particles (and these will be ultra-fine particles). It can be seen that particle formation is enhanced throughout the cleaning and for several hours afterwards under lockdown conditions owing to the higher ozone concentrations. In fact, the PM concentration is enhanced by 46% during cleaning under lockdown conditions, but only by 25% for the pre-lockdown conditions.

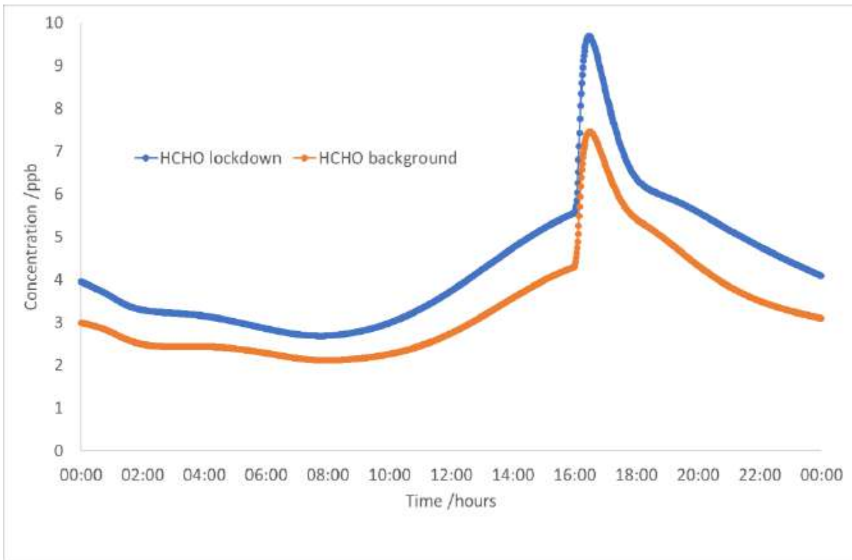

Figure 3: Formaldehyde concentrations for a typical residence pre- and during lockdown.

Figure 3 shows that formaldehyde concentrations are enhanced over the whole time period by about 30%. This is because of the enhanced ozone concentrations initiating more chemistry indoors, which then forms formaldehyde.

Conclusions

This short paper demonstrates that indoor air chemistry is likely to be enhanced significantly under lockdown conditions. This is because outdoor ozone concentrations have increased, which increases indoor ozone concentrations and a whole range of subsequent chemical processing. These reactions will lead to higher concentrations of PM and HCHO indoors, which could be enhanced further when activities such as cleaning occur. The extent of such increases will depend on the conditions in a particular building – ventilation, internal emissions etc. - but it likely that there will be wide ranging impacts on indoor air quality across the UK.

References

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Carslaw, N., Mota, T., Jenkin, M.E., Barley, M.H., McFiggans G., 2012. A significant role for nitrate and peroxide groups on indoor secondary organic aerosol. Environ. Sci. Technol. 46(17), 9290-9298.

Weschler, C.J. and Carslaw, N., Indoor Chemistry, Invited feature article for *Environmental Science and Technology*, 52, 2018, 2419-2428.

New Breathe London data: Covid-19 confinement measures reduce London air pollution

The Breathe London team hopes everyone is staying safe and healthy.

Covid-19 is a respiratory illness. Due to the potential impact of air pollution on respiratory and other illnesses, measurement data from the UK's air quality networks is important to the ongoing evaluation of risk to citizens during the pandemic.

The Breathe London consortium is committed to providing transparent air quality information for Londoners, and has evaluated data from the Breathe London monitoring network in the days before and after the government implemented restrictions* to reduce the spread of Covid-19. The government has stated it will keep these measures under constant review.

Preliminary results include:

Our preliminary results reveal substantial NO_2 pollution reductions after the measures went into place, particularly after social distancing was strongly encouraged on 16 March. Although further work is needed to determine the precise magnitude of reductions, we are sharing these results in provisional form and may periodically update them with additional data and analysis.

- The Breathe London network exhibits lower NO_2 pollution levels across Greater London starting around 18 March (see Figure 1, Figure 3a and 3b).
- From 17 March to 13 April, the following was measured in comparison to pre-confinement levels**:
    - Across the network, monitors register a 9-17% drop in NO_2 pollution.
    - In Central London, monitors show an average 20-24% reduction.
    - The greatest reductions occur during waking hours*** — between 6:00 and 22:00, average NO_2 reductions are 17-24% for the entire network and 28-30% in Central London (see Figure 2a).
- There is an apparent association between the reduced pollution levels and lower traffic congestion on London roads based on anonymous incident and slow-down information data from the Waze For Cities Program (See Figure 2 and 3).
    - Specifically, we assessed that traffic congestion reduced to such an extent that traffic was approaching free-flow in the vast majority of Greater London roads after the stay-at-home order (from 24 March to 13 April). We are further evaluating this and other methods to track changes in road transport emissions and the relationship to measured pollution.
    - Examining the daily pattern of traffic congestion also suggests a tie between Greater London's biggest drops in pollution and the biggest drops in congestion – which both occur in the late afternoon from around 3 to 7 pm (Figure 2a and 2b).

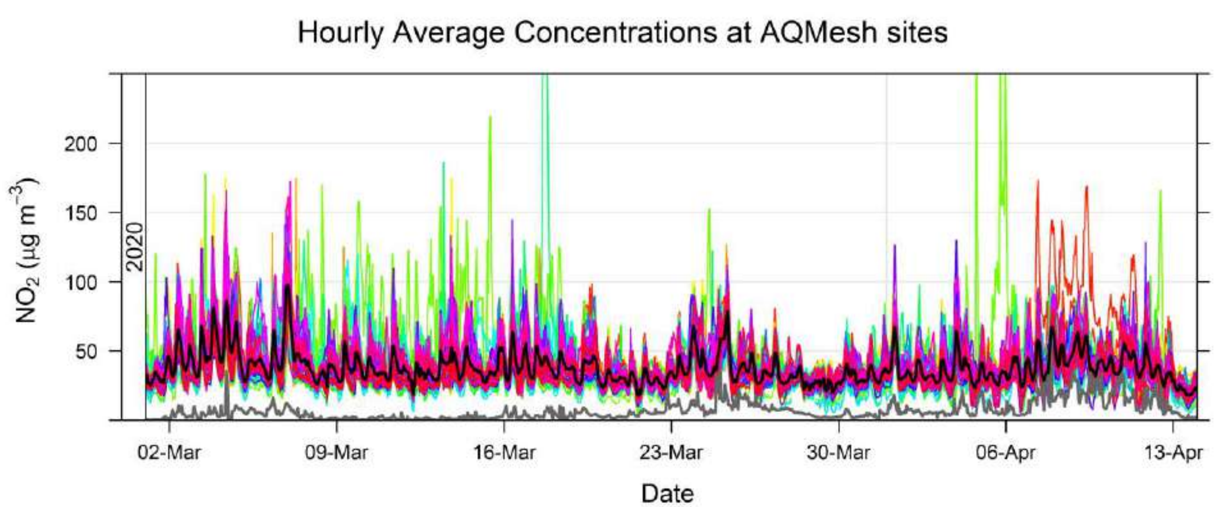
- The Breathe London network exhibits variability of $PM_{2.5}$ levels, but at this stage there is no clear reduction or evident association with the reduction in traffic.
  - London experienced pollution episodes [from 25 to 27 March](#) and [from 8 to 12 April](#) with elevated $PM_{2.5}$ levels, which have been captured by the Breathe London network. These increases were likely due to wind blowing in industrial and agricultural pollution from mainland Europe, as well as wood burning for the March episode.

**The measures and corresponding dates* include: 12 March – people with symptoms advised to stay home; 16 March – social distancing strongly encouraged; 20 March – cafes, pubs and restaurants ordered to close; nightclubs, theatres, cinemas, gyms and leisure centres told to close as soon as possible; 23 March – schools closed to most students, but open for vulnerable children and children of essential workers; ULEZ and other traffic charges suspended; 23 March (evening) – all citizens except essential workers advised to stay home.*

***Pre-confinement levels are based on Breathe London network data from December 2019, January 2020 and February 2020, with the exception of Christmas Day, Boxing Day, New Year's Eve and New Year's Day.*

****Local weather conditions favourable to pollution build-up may have occurred during a number of nights in the study period, which could explain some increases in pollution between 10:00pm to 6:00am.*

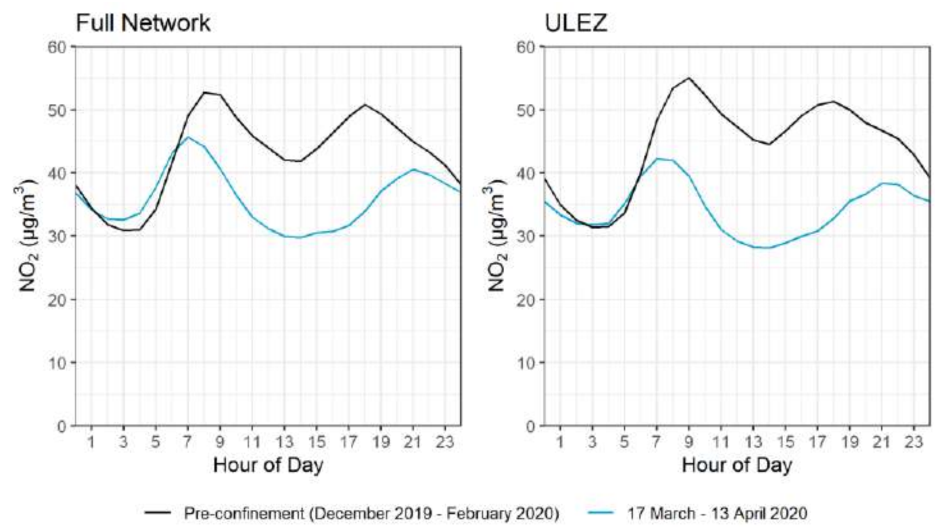
FIGURE 1: Breathe London network NO_2 measurements during 1 March to 13 April 2020.
Each trace is an individual monitor in the network ("AQMesh site"; 71 producing valid data in this period) and the thick black line represents the network average. The grey line represents upwind background from a rural site outside of London; elevated background readings indicate pollution from abroad.



Source: [Breathe London data](#)

FIGURE 2a: Breathe London network hourly mean NO_2 measurements during 17 March to 13 April 2020 compared to pre-confinement levels.

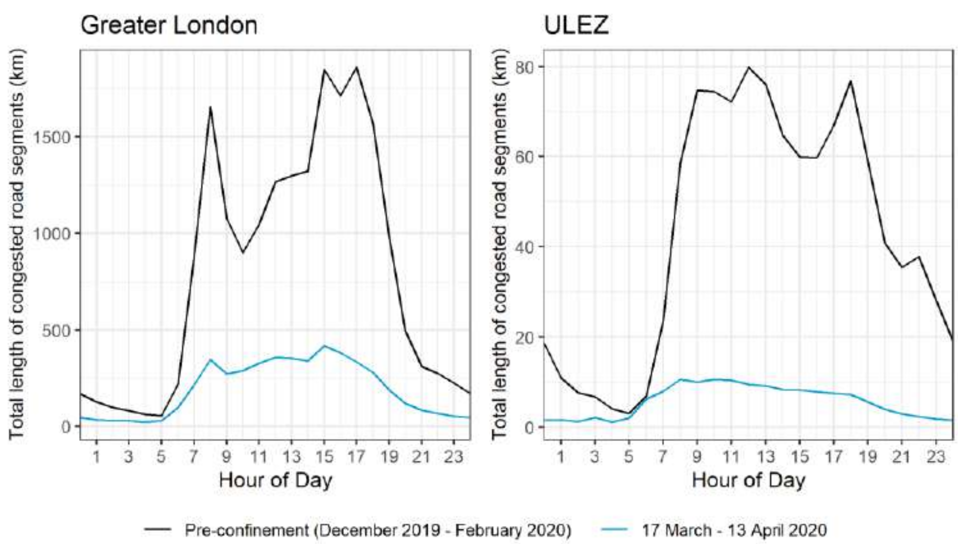
Levels shown for the full Breathe London network and a subset of monitors in Central London's Ultra Low Emission Zone (ULEZ). Both in Greater London and within the ULEZ, the greatest average decrease in NO_2 occurs during daytime hours.



Source: [Breathe London data]()

FIGURE 2b: Waze data – mean total length of congested roads by hour during 17 March to 13 April 2020 compared to pre-confinement levels.

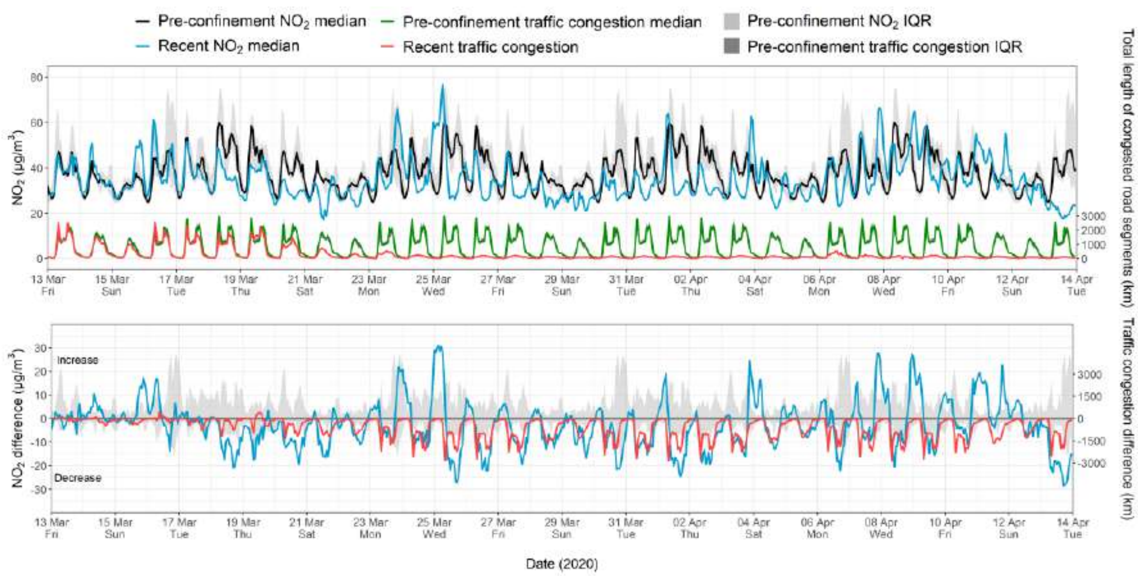
Levels shown for Greater London and within the ULEZ (scales are different).



Source: Waze data (by permission)

FIGURE 3a: Breathe London network NO_2 measurements during 13 March to 13 April 2020 in comparison to the typical hourly pre-confinement levels.

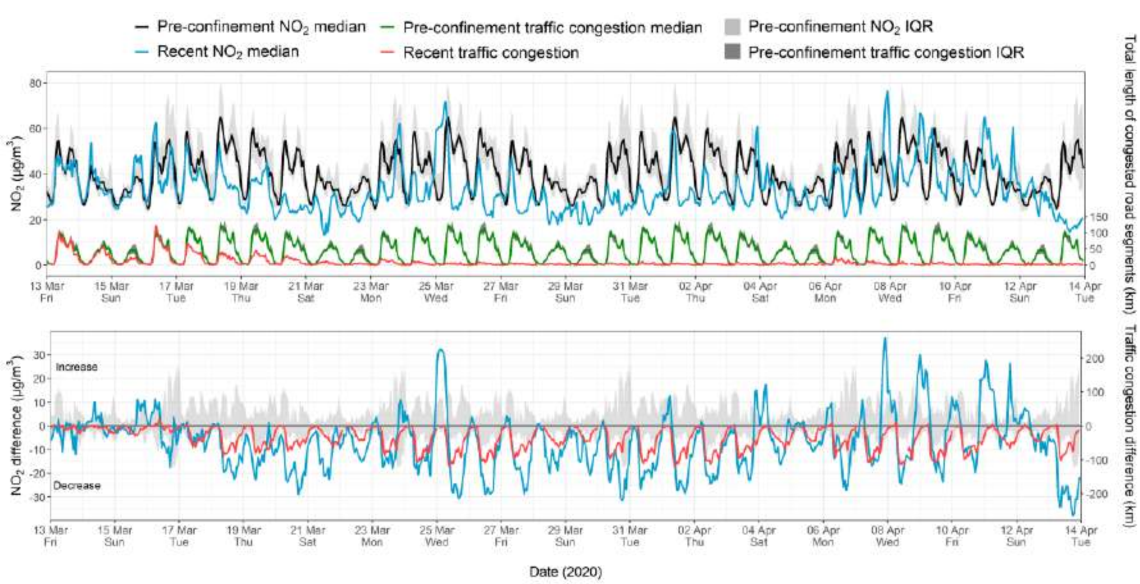
The blue line in the top half represents March/April measurements and the black line with grey shading represents the pre-confinement weekly average. The red line represents road congestion due to traffic in March 2020 and the green line represents the pre-confinement weekly congestion average. The bottom half shows magnitude of difference between the pre and post confinement measurements for both pollution (blue line) and traffic congestion (red line). Methods used to generate the figures are described below.



Source: [Breathe London data](#) and Waze data (by permission)

FIGURE 3b: Breathe London ULEZ monitors NO_2 measurements during 13 March to 13 April 2020 in comparison to the typical hourly pre-confinement levels.

Similar to Figure 3a above, but for the monitors in the ULEZ only.



Source: [Breathe London data](#) and Waze data (by permission)

Methodologies used to analyse data

Pre-confinement levels: To estimate the typical NO_2 concentrations during the pre-confinement period by weekday and hour (black lines and grey strips in Figure 3a and 3b), we assumed the behaviour across the network for the three months between December 2019 and February 2020 would be representative of March behaviour. This is not a perfect assumption because there is seasonal variation in monthly mean NO_2 concentrations across the year, with the magnitude varying from year to year. In future updates to this analysis, we may evaluate the uncertainty introduced with this assumption.

The pre-confinement concentrations in Figure 3a and 3b (black line) were obtained by first determining the median from all network sites for each date and hour and then pooling these network medians for each day-hour combination (i.e., pools of 13 values, collected across the 13-week December-February period). The median of each pool, or distribution, of network medians by day and hour is shown in Figure 3a and 3b. The interquartile range of these distributions is shown as the gray shaded area. Note that the day-hour time series repeats every seven days. We also conducted this analysis using the *mean* of the pool of network *means* (instead of the *median* of *medians* as described above), to constrain the upper bound in the method.

Traffic data (source: Waze For Cities Program by permission): The traffic data represented in the time series is a sum of road lengths with unique traffic congestion during a given hour based on Waze-generated anonymous incident and slow-down information, which we use as a proxy for transportation-related sources of pollution. We define a congested road segment as one exhibiting 60% or less of each road's free flow speed. We summed the length of congestion within two different areal extents:

1. the area within the ULEZ for comparison to the pods within the ULEZ and
2. the area of Greater London for the comparison with all pods.

Both the recent time series and historical pattern from the pre-confinement period (December 2019 through February 2020) are presented. Note that this data represents unique traffic congestion as identified by the Waze system, and not an estimate of actual emissions from on-road transport. This is a preliminary analysis and needs to be validated for robustness especially given the likely reduction in number of Waze users on London roads during the confinement period.

To derive the pre-confinement traffic data by weekday and hour, we calculated the interquartile range and median of all the Waze data by day of week and hour of day within the given time period (December 2019 to February 2020). The recent time series of congestion was similarly calculated for the same spatial domains as the pre-containment period.

Air Quality Expert Group
UK Air Quality During the Coronavirus Outbreak
Dr James Heydon (University of Nottingham) and Rohit Chakraborty (University of Sheffield)

The following response presents the findings from the first ever study into the composition of indoor emissions from DEFRA-certified multi-fuel stoves. We are currently in the 'write up' phase of the research but are sending our main findings early as a means of contributing to this call for information during the Covid-19 outbreak.

What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?

We anticipate indoor air quality during the Covid-19 pandemic to worsen for two reasons. First, there is an increased number of people at home all day with access to multi-fuel stoves. Second, our findings illustrate that DEFRA-certified stoves are releasing large quantities of particulate matter (pm1 and pm2.5) into the home during regular use. Considering the known links between particulate matter, lung function and susceptibility to respiratory illness[1] [2] [3], this is a cause for concern. Our research also indicates that these stoves are being used while vulnerable populations like children are not only in the home, but in the same room for extended periods of time. As these individuals are more sensitive to particulate matter the health risks posed are greater [4] [5]. While the 'burn season' for multi-fuel stoves is coming to a close now, the 'second wave' of Covid-19 cases anticipated to occur during winter will coincide with peak 'burn season'[6] and, in turn, increase exposure to indoor pm1 and pm2.5.

Our current research focuses on indoor emissions originating from DEFRA-certified multi-fuel stoves. We installed indoor and outdoor air quality sensors in the homes of 20 people over a four week period, asking them to record their stove use during this time. We also asked them to record any other emission-producing activity such as use of candles or cooking. This allowed us to co-locate stove use with sensor data and use modelling to exclude non-stove use emissions. 19 people used a DEFRA-certified stove and 1 person used an open fireplace in accordance with DEFRA's recommendations for their use in smoke control areas (smokeless coal only). Over the course of this research, three key findings are relevant to this call for information:

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[1] Schraufnagel, Dean E. et al. (2019) 'Air Pollution and Noncommunicable Diseases', CHEST, Volume 155, Issue 2, 409 - 416. https://doi.org/10.1016/j.chest.2018.10.042

[2] Wu, J. et al. (2018) Chronic Dis Transl Med. 2018 Jun; 4(2): 95–102. Published online Jun 8. 10.1016/j.cdtm.2018.04.001

[3] Chen, C., Wu, C., Chiang, H. et al. The effects of fine and coarse particulate matter on lung function among the elderly. Sci Rep 9, 14790 (2019). https://doi.org/10.1038/s41598-019-51307-5

[4] Oliviera, M. et al. (2019) 'Children environmental exposure to particulate matter and polycyclic aromatic hydrocarbons and biomonitoring in school environments: A review on indoor and outdoor exposure levels, major sources and health impacts', *Environment International*, Vol. 124, pp.180-204. https://doi.org/10.1016/j.envint.2018.12.052

[5] David Rojas-Rueda, Martine Vrijheid, Oliver Robinson, Aasvang Gunn Marit, Regina Gražulevičienė, Remy Slama, Mark Nieuwenhuijsen. Environmental Burden of Childhood Disease in Europe. International Journal of Environmental Research and Public Health, 2019; 16 (6): 1084 DOI: 10.3390/ijerph16061084

[6] Domestic Wood Use Survey (2016) https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/517572/Summary_results_of_the_domestic_wood_use_survey_.pdf

1) During regular use, average indoor pm1 and pm2.5 exposure are three times that encountered on days when stoves are not used (control). The average period of use is around four hours and there is no meaningful difference between weekdays and weekends.

2) During regular use, the rooms in which the stoves are situated (the living room being the most-common location) are being flooded with peaks of particulate matter for at least one hour. These levels are much higher than the average over the time the stove is lit. For example, most of the peaks are in the region of 15-45 ug/m3 for pm2.5 and 5-35 ug/m3 for pm1. This is significant. Emerging research demonstrates that intensity of PM exposure, in addition to the more traditional measure of exposure duration, has important links with respiratory illnesses and other health complications[7] [8].

3) These peaks of pm2.5 and pm1 correlate with the opening and closing of the stove doors. Put another way, the particulate matter coming into the house is occurring through normal stove use. As people consider their homes to be 'sanctuary spaces'[9], where air pollution does not occur, they are unaware that this is happening. Considering also that DEFRA-certified stoves do not test for indoor particulate matter emissions resulting from the opening of stove doors, this lack of awareness is informed by a certification scheme obfuscating the everyday risk posed by the units.

4) We found a significant increase in particle number concentration (PNC) with a small change in mass concentration for pm1 and pm2.5. This is important because there exists uncertainty about the toxicity and size measurement of such particulates[10]. As yet, there exists no UK regulation governing PNC.

Taken together, this analysis leads to the conclusion that DEFRA-certified solid fuel stoves project particulate matter indoors simply through the fuelling and refuelling process. As the stoves cannot be operated without this process, they must be recognised as normatively harmful when in use.

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[7] Lin et al (2017) 'Hourly peak concentration measuring the PM2.5-mortality association: Results from six cities in the Pearl River Delta study', *Atmospheric Environment*, 161, pp.27-33. https://doi.org/10.1016/j.atmosenv.2017.04.015

[8] Su, C., Breitner, S., Schneider, A. et al. Short-term effects of fine particulate air pollution on cardiovascular hospital emergency room visits: a time-series study in Beijing, China. Int Arch Occup Environ Health 89, 641–657 (2016). https://doi.org/10.1007/s00420-015-1102-6

[9] Heydon, J., Chakraborty, R. Can portable air quality monitors protect children from air pollution on the school run? An exploratory study. *Environ Monit Assess* 192, 195 (2020). https://doi.org/10.1007/s10661-020-8153-1

[10] Air Quality Expert Group (2018: 10) Ultrafine Particles in the UK, available at: https://uk-air.defra.gov.uk/assets/documents/reports/cat09/1807261113_180703_UFP_Report_FINAL_for_publication.pdf

Given the context of Covid-19, and the health risks associated with respiratory illness the virus engenders, we make the following recommendations:

a. In the near-term, or during subsequent periods, DEFRA or the relevant government department should issue recommendations for people not to use solid fuel stoves during lockdowns. Their use should be emphasised as particularly risky for vulnerable populations, such as the elderly or those with underlying respiratory conditions.
b. Over the longer-term, it is recommended that health warnings be issued with new solid fuel stoves. This will allow buyers to make an informed judgement as to whether they wish to purchase one or not.
c. Also over the longer-term, it is recommended that the testing at the basis of DEFRA's stove certification scheme includes indoor emissions tests for refuelling under real world conditions. At the least, these should account for pm1 and pm2.5 and, following further research at point d below, PNC.
d. Particle number count from domestic burning requires a major real-world study. Toxicity should be well defined for different types of particles.

Estimation of ambient NO2 and PM2.5 concentration change in Wales during COVID-19 outbreak

Professor Paul D. Lewis[1,2], Victoria Seller[1], Tom Price[3], Hamidreza Eskandari[1]

Centre for Health and Environmental Management Research and Innovation
[1]School of Management; [2]Medical School, Swansea University, SA2 8PP, UK; [3] Pollution Control & Private Sector Housing, Swansea Council

This brief summary provides a brief overview of data and results from a preliminary analysis carried out at Swansea University estimating the ambient reduction in ambient NO2 monitoring stations across Wales since the UK Government introduced restrictions through a 'lockdown' on March 23rd 2020. The full analysis to date with initial estimates for concentration reductions at 13 roadside, urban background, kerbside, urban centre and urban industrial sites, can be retrieved at: https://chemri.shinyapps.io/Wales_COVID19_Evidence/

This research is ongoing and will be updated as more data becomes available during the COVID-19 lockdown period. We have estimated NO2 concentration reductions using a multi-step process. Using trend analysis we are able to determine significant temporal changes in NO2 levels at sites since lockdown. Using random forest (RF) models we are able to predict the expected **meteorologically normalized NO2 levels** per day to compare **against measured daily average NO2 levels** at each location. This permits a comparison of **median residual ambient NO2 differences** (between predicted and measured levels) between pre- (01/01/2020 – 23/03/2020) and post-lockdown periods across monitoring stations. Datasets are described at the end of the summary.

We have developed an online tool, updated daily, using our models to track pollution change in Wales: https://chemri.shinyapps.io/upload/

The example plots in Figure 1 are for Swansea Roadside monitoring station and demonstrate the model outputs:

(i) Measured (red line) and RF model meteorological normalised predicted (blue line) average NO2 levels for each day in 2020. The start of COVID-19 lockdown on March 23rd 2020 is highlighted.

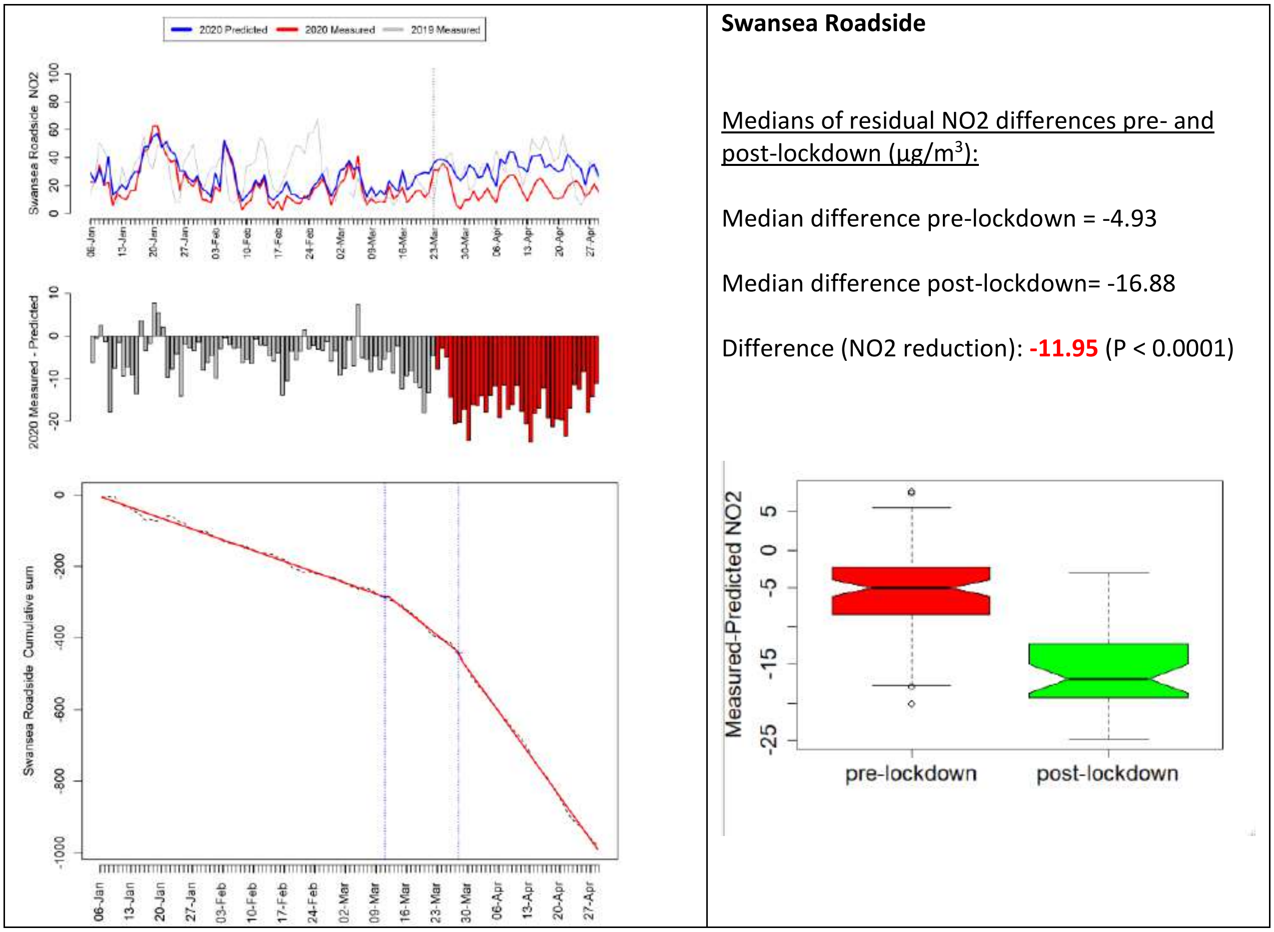
(ii) Daily residual differences between measured and RF weather normalized model predicted average NO2.

(iii) Estimated significant breakpoints identified by trend analysis (on cumulative sum of residual differences).

Medians of residual differences between model predicted and measured NO2 levels ($\mu g/m^3$) and boxplots of residual distributions are also shown alongside each plot for both pre- and COVID-19 post-lockdown. Differences between the median NO2 levels in both these periods are provided along with corresponding p-values. The following table summarises the decrease in median NO2 following lockdown on 23rd March relative to all days prior in 2020. The NO2 reductions at the urban industrial site at Port Talbot Margam and the urban background site at St Julian's Comprehensive School in Newport were not statistically significant. Excluding these sites, the mean difference in NO2 reduction across the remaining 11 sites is -8.96 $\mu g/m^3$ (SE=1.47).

	Swansea Roadside	**Swansea Morriston**	**Hafod DOAS**	**St Thomas DOAS**	**Cwm Level Park**	**Cardiff Centre**	**Newport St Julians**	**Newport M4**	**Hafod-Yr-Ynys**	**Port Talbot Margam**	**Chepstow A48**	**Rhondda Pontypridd**	**Wrexham**
NO2 reduction	-11.9	-7.5	-12.5	-8.8	-4.8	-12.6	-1.4	-7.9	-20.7	-3.4	-9.2	-6.5	-3.4

Figure 1.

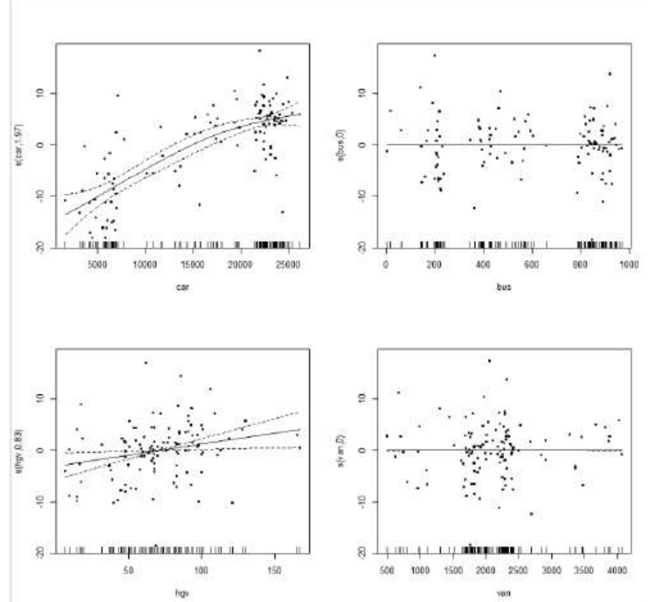


We have further extended our analysis by using generalized additive models to predict the impact of traffic volume change different classes of traffic (car, van, bus and HGV) on the daily average predicted NO2 change at four roadside monitoring stations in the Swansea urban area during this period.

To estimate the relationship between traffic reduction and ambient NO2 levels since COVID-19 lockdown, we calculated the NO2 residual differences between RF model predicted meteorological normalized and measured NO2 data at Swansea Roadside, Swansea Morriston, Swansea Hafod DOAS and Swansea St Thomas DOAS over a ten year period and used as a dependent variable to train GAMS models per site with total daily traffic count data for cars, light vans, HGV vehicles and buses as independent variables. Thus, each GAMS model predicts daily average residual NO2 levels when using daily traffic count data from 2020 as input. Each GAMS model could also reveal whether the change in volume of any vehicle type was significantly associated with change in NO2 levels since COVID-19 lockdown. Non-significant terms were retained in the optimal models to control for those vehicle types when predicting associations of cars with NO2 reduction since COVID-19 lockdown.

The plots in Figure 2 show an example for Swansea Roadside of the linear and non-linear relationships between vehicle type volume and predicted residual change in NO2 models. For all sites, there was a positive and significant linear, or near linear, relationship between the volume of cars and NO2 residual values. There was no similar relationship observed for cars, HGV vehicles or buses at three sites but HGV had a significant relationship at Swansea Roadside.

Figure 2.



Swansea Roadside

Vehicle	Estimated df	p-value
s(car)	2	< 2e-16 ***
s(bus)	0	0.97449
s(hgv)	0	0.00719 **
s(van)	0	0.30709

R-sq.(adj) = 0.638
Deviance explained = 64.7%

The following tables show the daily median counts of vehicle type for pre- and post-lockdown at each of the four monitoring sites. The tables also show the predicted reduction in NO2 from the GAMS models when the volume of cars is entered at median values observed for either the pre- or post-lockdown periods (controlling for vans, HGV and bus volumes). Thus, the models permit the local authority to estimate the median daily reduction in NO2 that could occur by a unit reduction of daily median cars. For example, at Neath Road (corresponding to the Swansea Hafod DOAS) within an existing Air Quality Management Area (AQMA), the model predicts that a 10% reduction in cars (1100) without reducing other vehicle types would lead to approximately a median daily reduction of 2 μg/m^3 in NO2.

Swansea Roadside							
	Daily median volumes				Predicted reduction in NO2 (μg/m^3)		
	Pre-lockdown	Post-lockdown	Difference	P value	Car volume at Pre-lockdown	Car volume at Post-lockdown	Difference
Cars	22440	6017	16423	<0.0001	-4.23	-13.34	**-9.11**
Light vans	2280	1694	586	<0.0001			
HGV	82	46	36	<0.0001			
Bus	836	214	623	<0.0001			

Swansea Morriston Roadside							
	Daily median volumes				Predicted reduction in NO2 (μg/m^3)		
	Pre-lockdown	Post-lockdown	Difference	P value	Car volume at Pre-lockdown	Car volume at Post-lockdown	Difference
Cars	32431	10310	22121	<0.0001	-2.35	-8.40	**-6.05**
Light vans	2280	1694	586	<0.0001			
HGV	65	24	41	<0.0001			
Bus	110	74	36	<0.0001			

Swansea Hafod DOAS							
	Daily median volumes				Predicted reduction in NO2 (µg/m^3)		
	Pre-lockdown	Post-lockdown	Difference	P value	Car volume at Pre-lockdown	Car volume at Post-lockdown	Difference
Cars	15225	4225	11000	<0.0001	-4.63	-24.49	**-19.86**
Light vans	1316	534	782	<0.0001			
HGV	77	12	65	<0.0001			
Bus	108	38	70	<0.0001			

Swansea St Thomas DOAS							
	Daily median volumes				Predicted reduction in NO2 (µg/m^3)		
	Pre-lockdown	Post-lockdown	Difference	P value	Car volume at Pre-lockdown	Car volume at Post-lockdown	Difference
Cars	22774	6046	16728	<0.0001	8.56	-1.54	**-10.10**
Light vans	1102	545	557	<0.0001			
HGV	70	31	39	<0.0001			
Bus	97	43	54	<0.0001			

Finally, we have also carried out an initial assessment to determine whether ambient reduction had occurred in PM2.5 levels in South Wales. PM2.5 have however been increased substantially since lockdown, relative to pre-lockdown levels, at all monitoring stations on four occasions for periods of days. We have further explored whether these regional increases were due to transboundary effects and non-local sources. Using hourly satellite and ensemble modelled PM2.5 data retrieved from the Copernicus Atmosphere Monitoring Service (CAMS) for North Western Europe we were able to determine that all four temporal episodes of increased PM2.5 levels were occurring across a wide geographical area extending across the South of England and into France.

Figure 3.

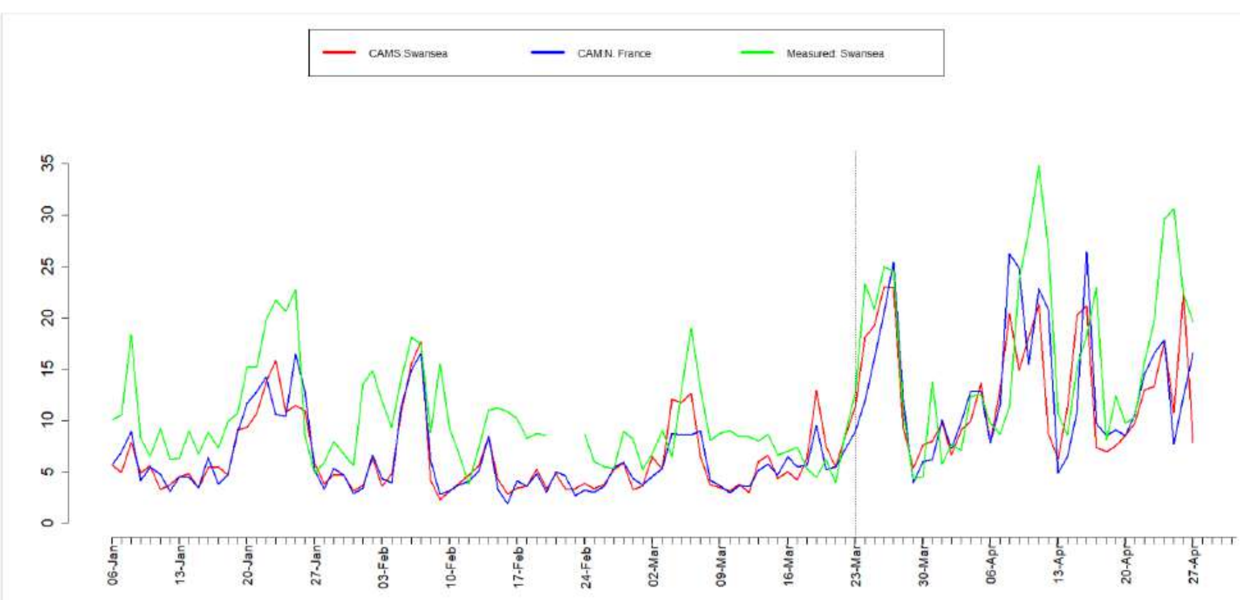


Figure 3 shows daily levels of PM2.5 throughout 2020 for measured data at Swansea Port Tennant monitoring station and CAMS modelled data for a location in the North of France (30km South East from Calais) and Swansea. The CAMS data for Swansea can be considered as urban background over a 10km by 10km area. Whereas, with these transboundary effects, it is difficult to determine change in ambient PM2.5 levels in Wales, the data indicates the need to establish whether the elevated levels of PM2.5 since lockdown have had a detrimental impact on vulnerable people during this period.

We are working with colleagues at the Farr Institute in Swansea University Medical School and SAIL databank to assess potential impacts of increased PM2.5 on vulnerable groups during this period, particularly respiratory and cardiovascular patients, within the Welsh population.

Datasets used:

Air pollution and modelled meteorological data: Hourly measurements for NO2 (µg/m^3), modelled wind speed (m/s), modelled wind direction (°) and modelled temperature (°C) from 01/01/2010 (or 2011) to the current day are retrieved daily for 13 AURN monitoring stations from Air Quality Wales through functions in the R statistical environment 'openair' package. The sites include: Cardiff Centre (Urban centre), Swansea Roadside (Roadside), Swansea Morriston Roadside (Roadside), Swansea Hafod DOAS (Roadside), Swansea St Thomas DOAS (Roadside), Swansea Cwm Level Park (Urban background), Newport St Julians Comp School (Urban background), Newport M4 Junction 25 (Roadside) , Hafod-Yr-Ynys (Kerbside), Port Talbot Margam (Urban industrial), Chepstow A48 (Roadside), Rhondda Pontypridd Gelliwastad Road (Roadside), Wrexham (Roadside).

Hourly measurements for PM2.5 (µg/m^3), modelled wind speed, modelled wind direction and modelled temperature (µg/m^3), from 01/01/2019 to the current day are retrieved daily in the same manner for 9 monitoring stations at Cardiff Centre, Swansea Roadside, Swansea Port Tennant Roadside, Newport St Julians Comp School, Port Talbot Margam, Chepstow A48, Wrehxam, Caerphilly Fochriw (Roadside), Anglesey Brynteg (Other).

Measured meteorological data: Hourly average measured data for wind speed (m/s), wind direction (°), temperature (°C) and relative humidity (%) from 01/01/2010 to present day for Cwm Level Park 30m mast (Swansea Council).

Traffic count data: Hourly traffic count data from 01/01/2010 to present day (Swansea Council) for 4 automatic traffic counters within the city: Carmarthen Road (4, lanes, adjacent to Swansea Roadside AURN), Fford Cwm Tawe (4 lanes, adjacent to Swansea Morriston Roadside AURN), Neath Road (2 lanes, adjacent to Swansea Hafod DOAS) and Pentreguinea Road (2 lanes, adjacent to Swansea St Thomas DOAS).

Technical Note

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Project title	AQEG COVID-19 response	Job number	
cc		File reference	
Prepared by	Katie Fawcett, Angie Chan	Date	29 April 2020
Subject	Can you provide estimates for how emissions and ambient concentrations of NOx, NO2, PM, O3, VOC, NH3 etc may have changed since the COVID outbreak?		

1 Introduction

The UK has introduced severe travel restrictions and social distancing to slow the spread of the COVID-19 disease since 23rd March 2020. These include the suspension of non-essential business, advice to remain at home and the introduction of curfews. The consequence has resulted a decrease in commercial, industrial and transport activity and associated emissions of local air pollutants. However, movement of people and goods persists to sustain the provision of essential services, including hospitals, supermarkets, emergency services and refuse collection. This technical note investigates how emissions and ambient concentrations of the most concerning pollutants in the UK may have changed since the COVID outbreak.

2 Changes in emissions

Traffic emissions

The current traffic volume data have been obtained from the coronavirus press conferences[1] (26th April 2020, presented in Appendix A). The number of motor vehicles usage in Great Britain is 62% lower than the first week of February 2020, before the outbreak.

Measured traffic data across Newcastle have been provided by Arup's Transport Team[2], the data indicate a decrease in traffic volume after implementation of severe traffic restrictions. After the implementation since 23rd March 2020, median daily traffic volumes across the roads in Newcastle have fallen by 60% (and 59% for weekend). Figures are presented in Appendix A.

[1] Slides and datasets to accompany coronavirus press conference: https://www.gov.uk/government/publications/slides-and-datasets-to-accompany-coronavirus-press-conference-26-april-2020 . [Accessed: April 2020]
[2] Traffic counters reporting to the Urban Observatory

Aviation emissions

Due to the travel restriction and border closures, usage of aircraft has been substantially reduced. There has been a considerable decrease in flight departures recorded from the 10 biggest UK airports since the lock down. This decrease can be seen in Appendix B.

Other emissions

Commercial emissions are also anticipated to be reduced due to office and restaurant/ café closures, while domestic emissions, conversely are expected to increase as the general public is advised to remain at home. The impact of COVID-19 to other sectors such as agriculture and industrial activities are yet to be seen, further review into these sources is required when the data are available.

3 Changes in ambient concentrations

The changes in ambient concentration of a variety of the most concerning pollutants are presented below and the methodology is presented in Appendix C. The supporting air quality data are presented as graphs in Appendix D. The monitoring sites examined are:

- London Marylebone Road (roadside);
- London Bloomsbury (urban background);
- London Eltham (suburban background);
- Rochester Stoke (rural background);
- Scunthorpe Town (urban industrial); and
- Horley (suburban industrial/airport).

NOx

Roadside

- NOx concentrations measured in 2020, during the implementation of social distancing measures, are lower at the roadside location studied (Marylebone Road), compared to the average NOx concentration in 2015-2019.

Background

- NOx concentrations are marginally lower at the rural background site since the COVID outbreak. Changes cannot be observed at the urban and suburban background sites.

Industrial/airport

- Lower concentrations can be observed at the Horley since the COVID outbreak but not at the Scunthorpe Town site.

NO_2

Roadside

- NO_2 concentrations at the roadside location are lower, compared to the averaged NO_2 concentration in 2015-2019. Reduction in NO_2 concentrations can also be observed since the COVID outbreak.

Background

- Background sites don't show the same scale of reduction, and multiple peaks can be seen at the London Eltham site.

Industrial/airport

- Some marginal reductions can be observed at both industrial sites, with the greatest reductions seen at the Horley site.

Particulate Matter (PM)

Roadside

- No obvious reduction in PM_{10} concentrations at the roadside location can be seen across the examined period.

Background

- Several peaks in PM_{10} concentrations are observed across all the background sites, and clear reductions cannot be seen.

Industrial/ airport

- The industrial site does not show any reduction in PM_{10} concentrations since the COVID outbreak.

There are clear peaks in the data at all sites and they are caused by a particulate pollution episode in mid-April[3]. This data shows that PM_{10} is heavily impacted by meteorological conditions and can be considered as being affected by regional transboundary sources.

Ozone (O_3)

Roadside

- A notable increase in O_3 concentration is observed at Marylebone roadside location since the COVID outbreak, measured concentrations are also higher than the averaged O_3 concentration in 2015 – 2019 persistently.

[3] London Air Quality Network - Moderate PM_{10}, $PM_{2.5}$ and O_3 pollution episode in London and Southeast England recorded on 8th to 12th April. Available at: https://www.londonair.org.uk/london/asp/publicepisodes.asp?species=All&region=0&site=&postcode=&la_id=&level=All&bulletindate=08%2F04%2F2020&MapType=Google&zoom=9&lat=51.4750&lon=-0.119824&VenueCode=&bulletin=explanation&episodeID=ModPM10PM25O3MidApril2020&cm-djitdk-djitdk= . [Accessed: April 2020]

Background

- Increases in O_3 concentrations are observed across all the background sites after the implementation of social distancing. Measured O_3 concentrations are also higher than the averaged concentrations recorded in 2015 – 2019.

Industrial

- The examined industrial sites do not measure O_3 concentrations.

The increase in O_3 concentrations across all sites was caused by a pollution episode recorded in mid-April[3]. This high O_3 concentrations could be due to the increased sunshine this year over April, which has been above average.

In addition, O_3 is a secondary pollutant and its concentration is predominantly related to reactions with other pollutants in the atmosphere. In local setting, O_3 concentration is reduced by exhaust emissions. Given the characteristic of O_3 and the decrease in NOx/NO_2 concentrations after the COVID outbreak, the general increase in O_3 concentrations is considered to be expected.

VOC (benzene)

Roadside

- Reductions in benzene concentrations can be seen during this assessed period at the roadside location,

Background

- The measured benzene concentrations do not show any obvious reductions. There are also some peaks in the assessed period, the cause of these are unknown at the time of writing.

Industrial

- The examined industrial sites do not measure benzene concentrations.

Summary

Since the COVID outbreak, the most noticeable changes in emissions are evident in the transport and aviation sectors and this due to the severe travel restrictions. Emissions associated with other sectors will require further evidence to support.

$NOx/\ NO_2$ concentrations are directly affected by the volume of traffic and the resulting exhaust emissions. $NOx/\ NO_2$ concentrations at roadside location responds most swiftly to the traffic reduction where concentrations have recorded decreases. Reductions can also be observed at industrial sites and no prominent changes are observed across the background sites.

O_3 concentrations are higher across all the sites, it is likely due to the increased sunshine over April. It is also important to note that O_3 is a secondary pollutant were its concentration is reduced by exhaust emissions. Given the general decrease occurred in NOx/NO_2 concentrations since the COVID outbreak, the increment in O_3 concentration is considered to be expected.

Relationship between PM concentrations and the COVID outbreak cannot be determined at the time of preparing this technical note, as obvious changes cannot be observed in the assessed period.

Some reductions in benzene concentrations can be observed at the roadside location, but no obvious changes at the background site.

The above findings are only representative for the examined period, further data will be required to inform the relationship between air quality and the COVID outbreak.

Technical Note

29 April 2020

Appendix A Change in Transport Use – Great Britain

Figure A1: Change in Transport Use – Great Britain (25th April 2020)

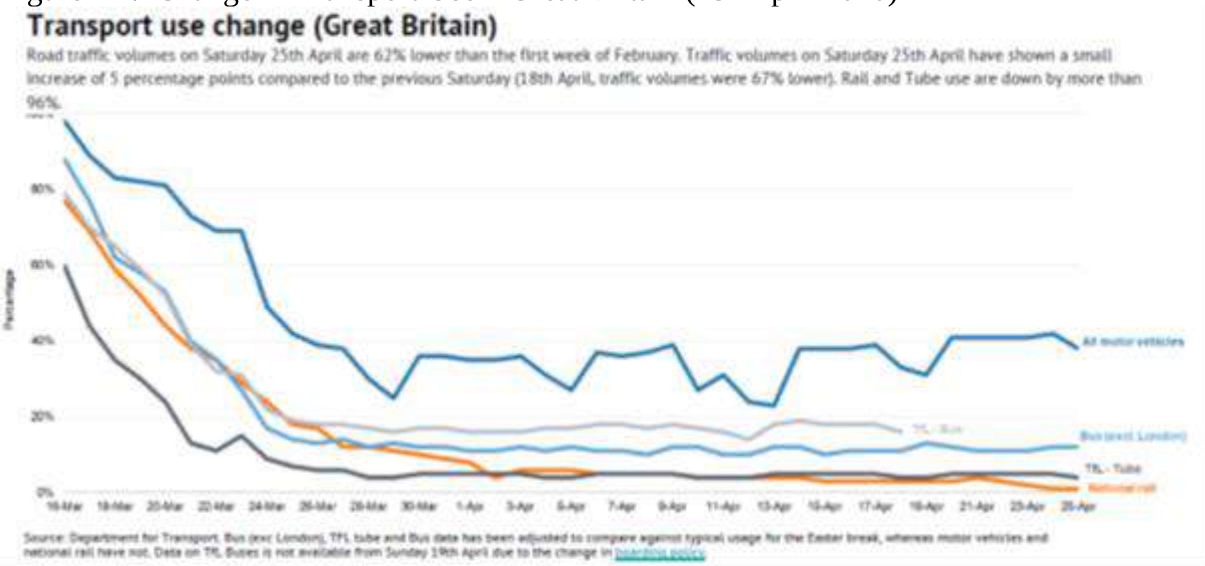


Figure A2: Total vehicle count across Newcastle

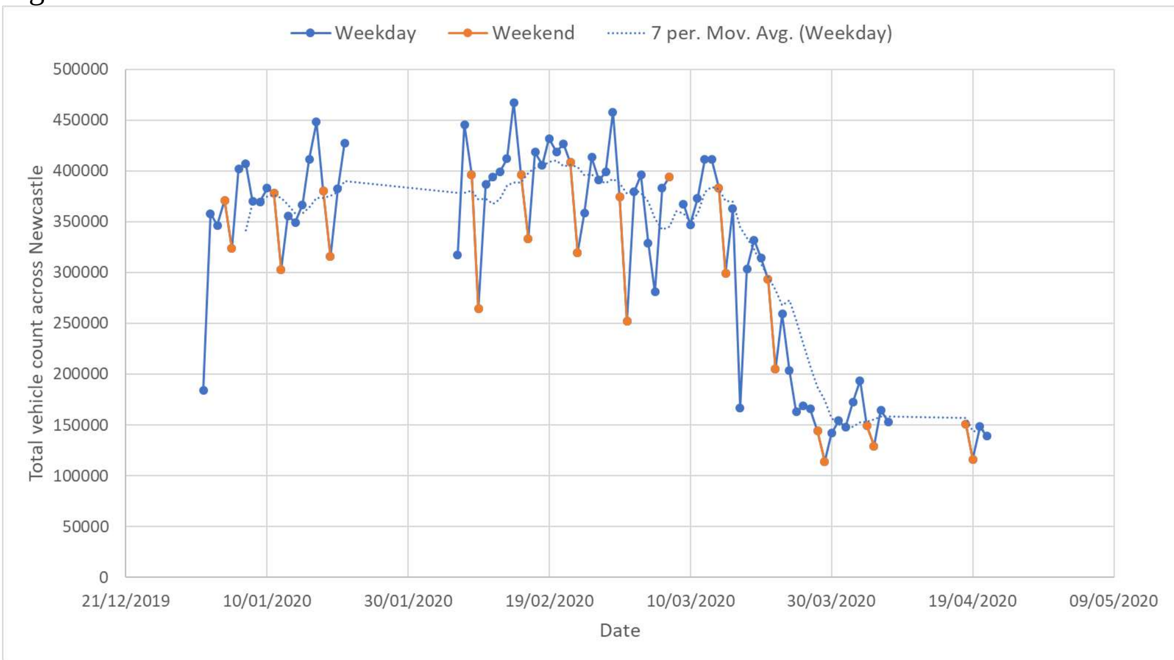
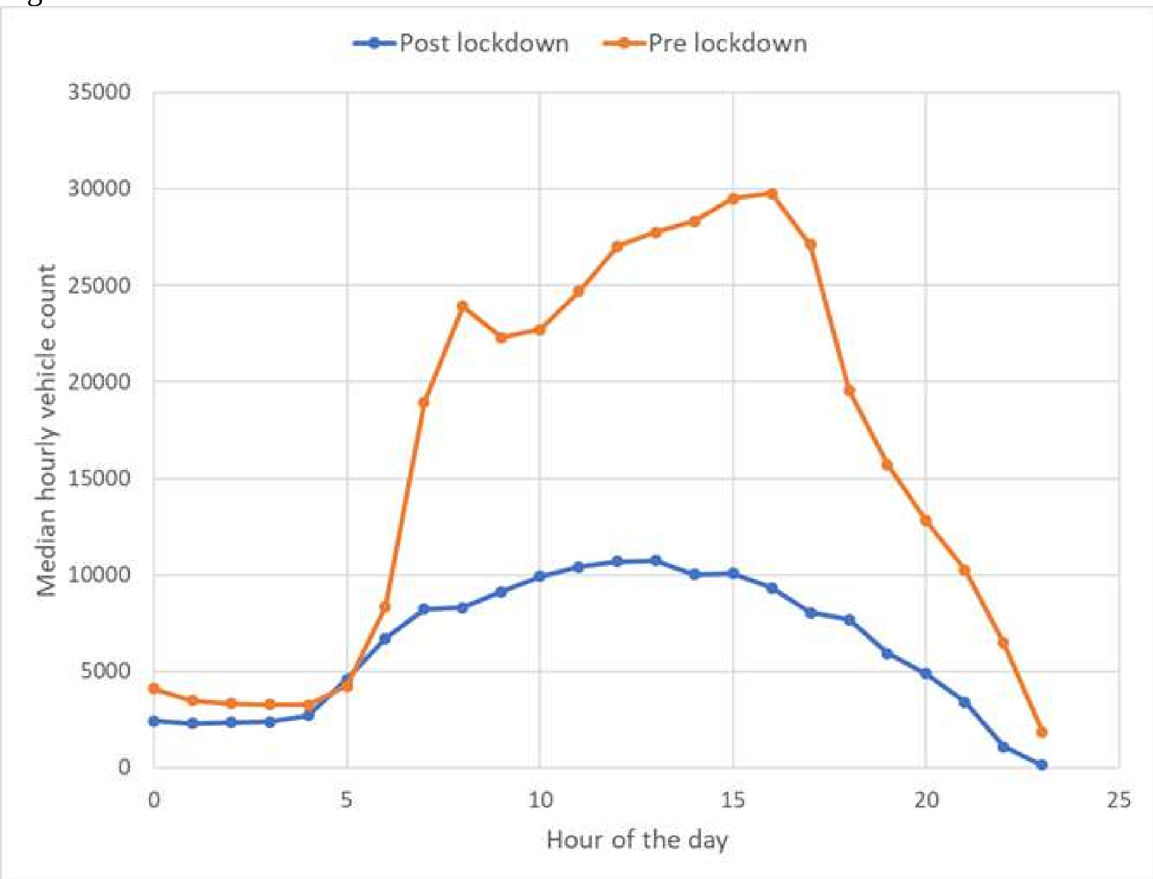
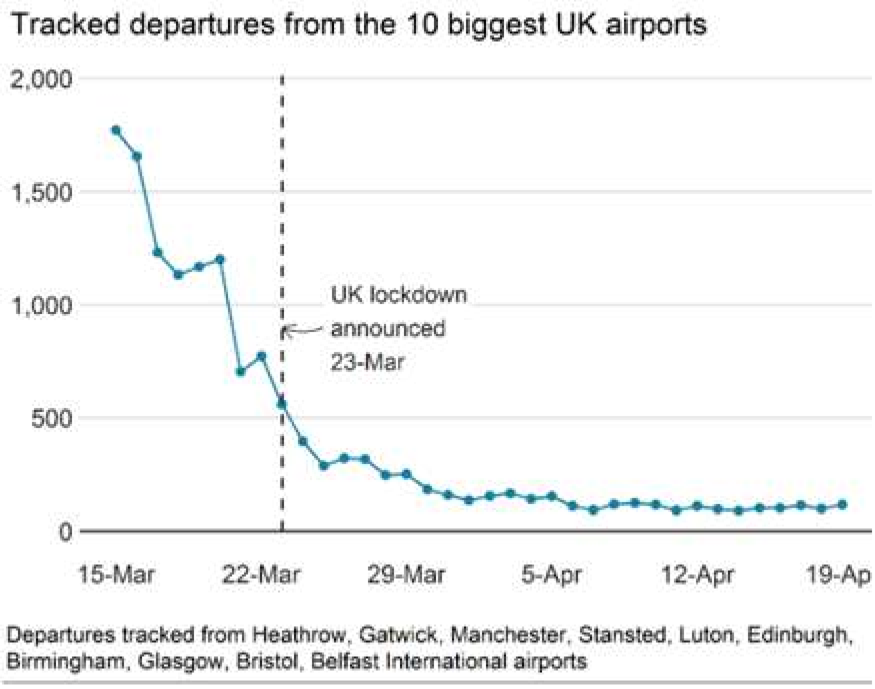


Figure A3: Total vehicle count across Newcastle



Appendix B Decline in Flight Use due to Coronavirus[4]



4

Appendix C Methodology

All air quality data were obtained from Defra's Automatic Urban and Rural Network (AURN)[5]. The period considered in this analysis is from 20th March to 26th April 2015 – 2020. The analyses focus on comparing the differences before and after 23th March 2020 where severe travel restrictions imposed by the UK Government.

The air quality data are obtained from the following sites:

- London Marylebone Road (roadside);
- London Bloomsbury (urban background);
- London Eltham (suburban background);
- Rochester Stoke (rural background);
- Scunthorpe Town (urban industrial); and
- Horley (suburban industrial/airport).

Hourly averaged data were used, and the pollutants considered in the analyses are NOx, NO_2, PM_{10}, O_3 and VOC (benzene). Ratified data are not available for the examined period in 2020.

[5] Defra – Automatic Urban and Rural Network. Available at: https://uk-air.defra.gov.uk/networks/network-info?view=aurn . [Accessed: April 2020]

Appendix D Air Quality Data

NOx

Figure D1: Monitored NOx Concentration at Marylebone Road (roadside location)

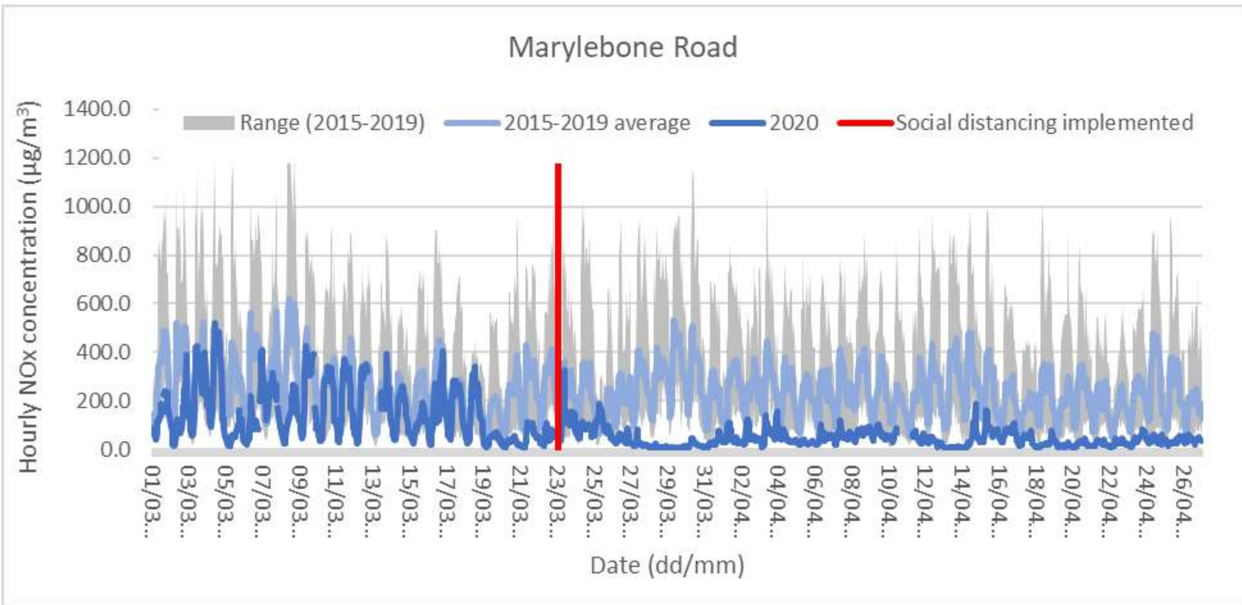


Figure D2: Monitored NOx Concentration at London Bloomsbury (urban background location)

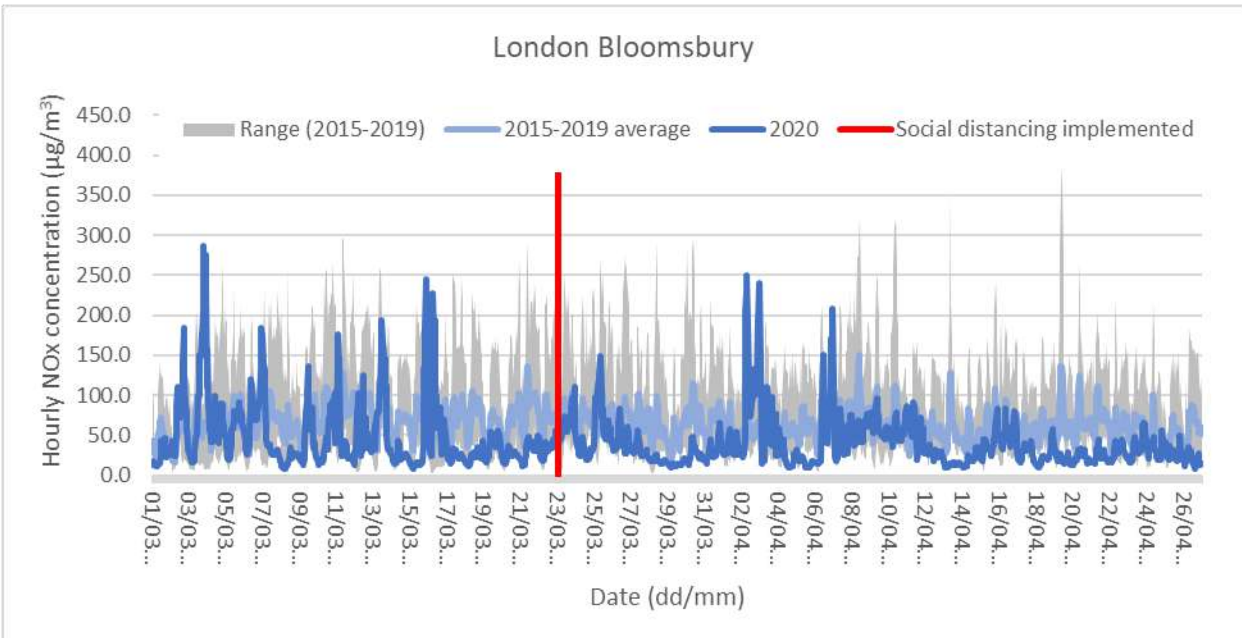


Figure D3: Monitored NOx Concentration at London Eltham (suburban background location)

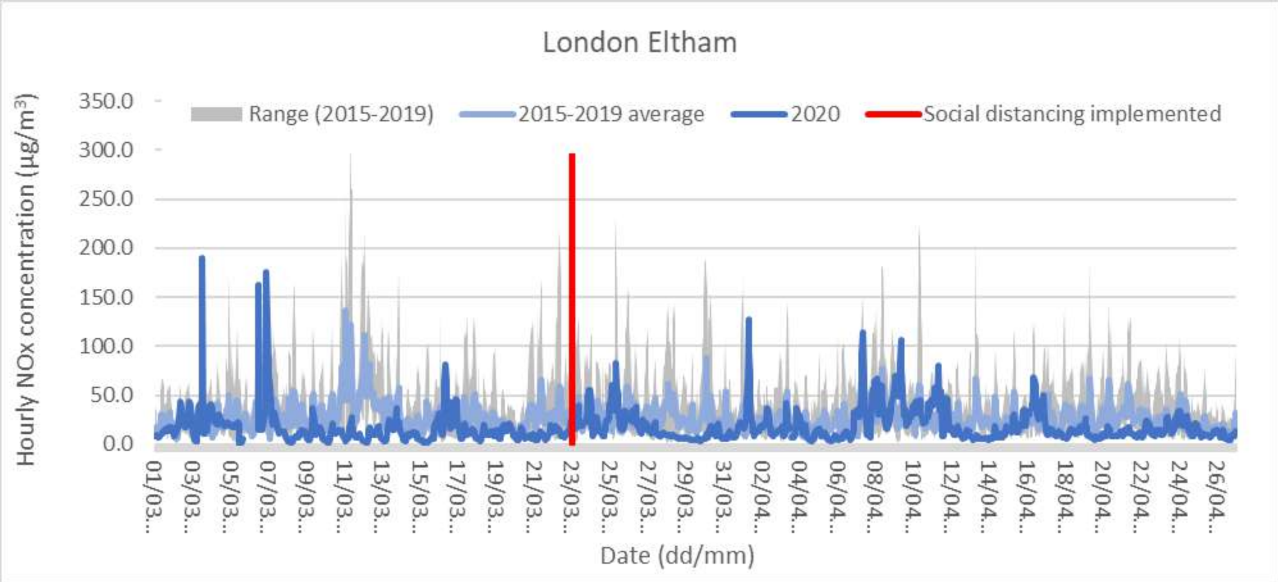


Figure D4: Monitored NOx Concentration at Rochester Stoke (rural background location)

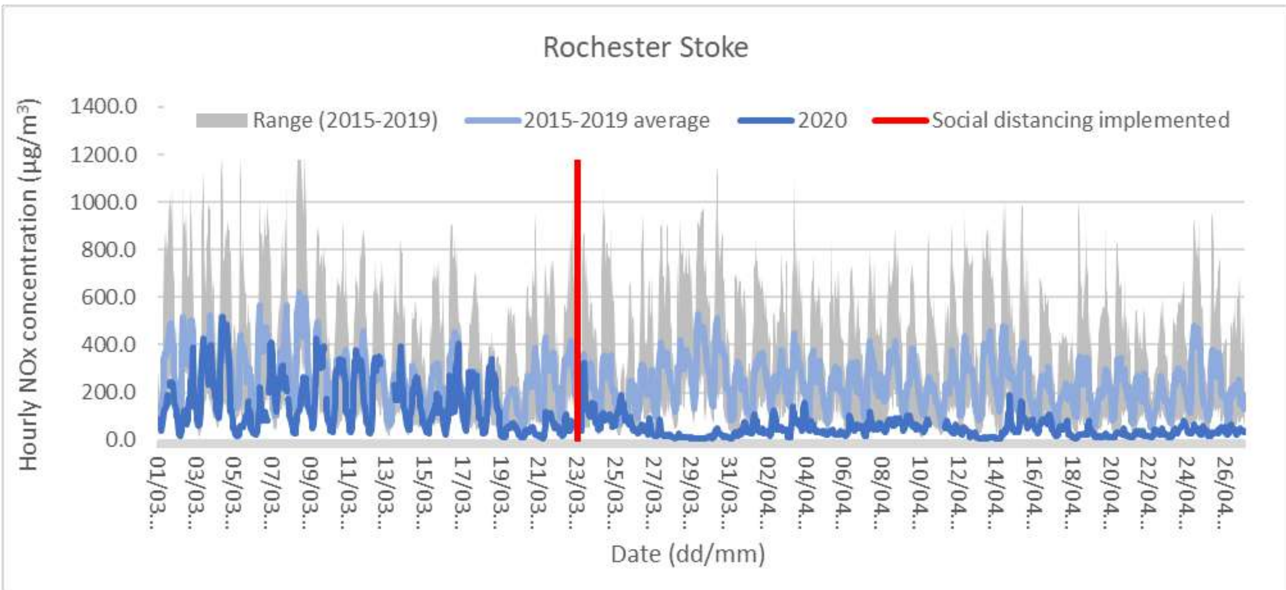


Figure D5: Monitored NOx Concentration at Horley (suburban industrial/airport location)

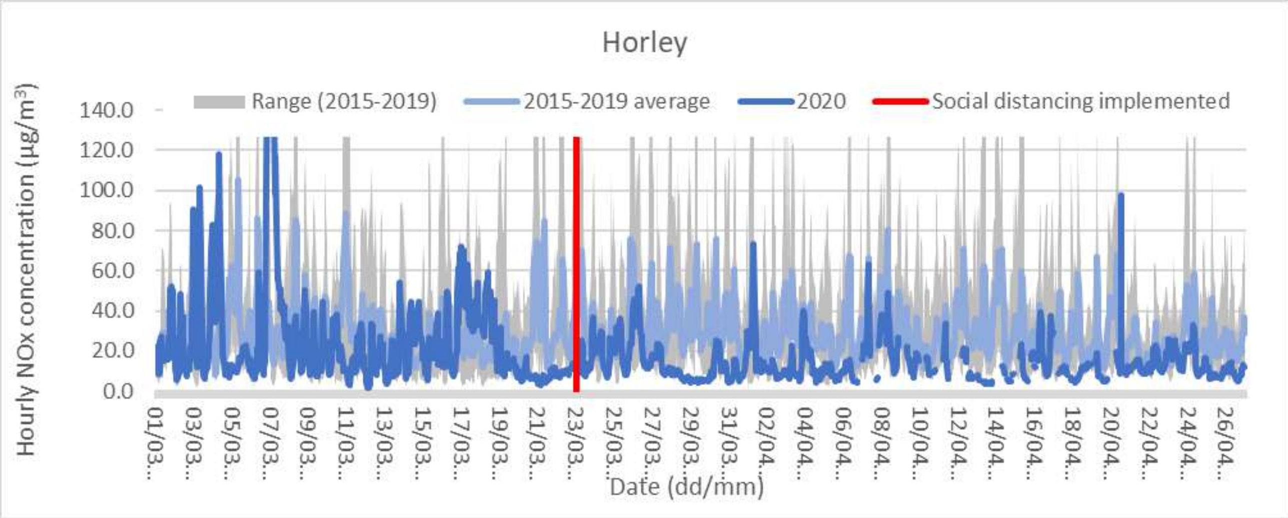
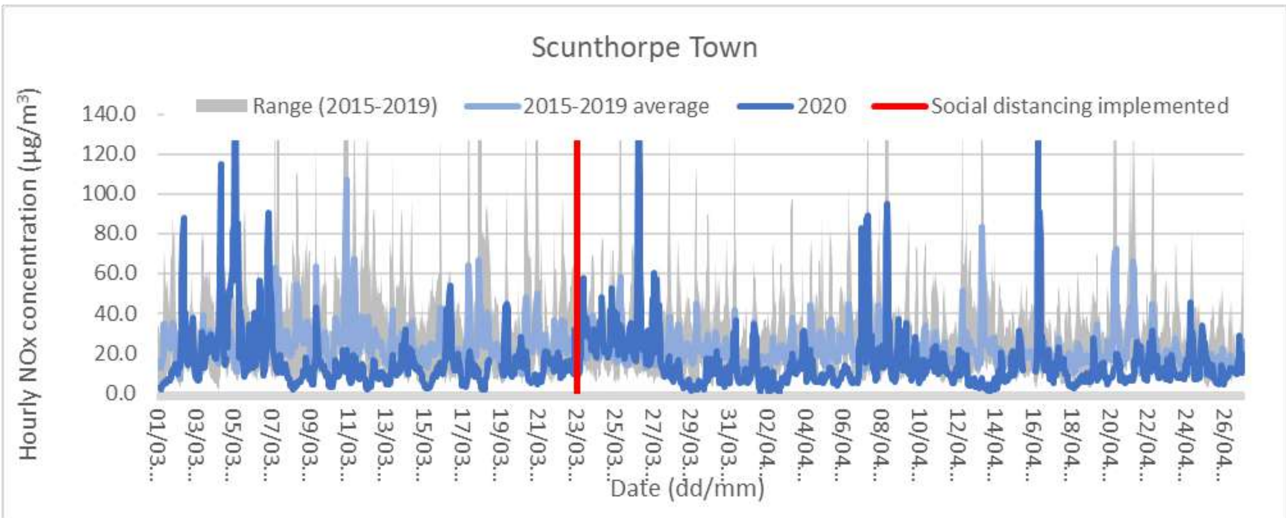


Figure D6: Monitored NOx Concentration at Scunthorpe Town (industrial location)



NO_2

Figure D7: Monitored NO_2 concentrations at Marylebone Road (roadside location)

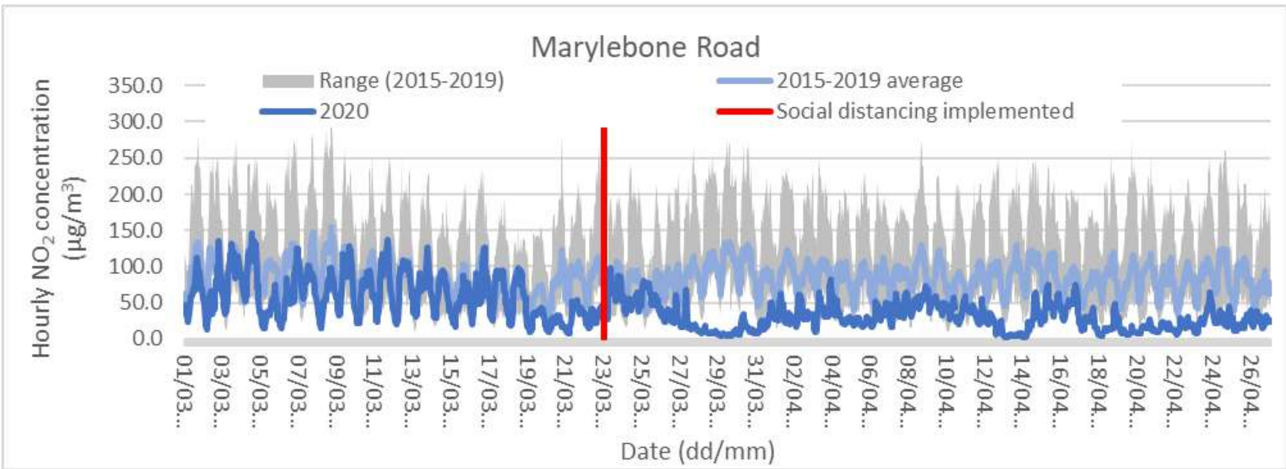


Figure D8: Monitored NO_2 concentrations at London Bloomsbury (urban background location)

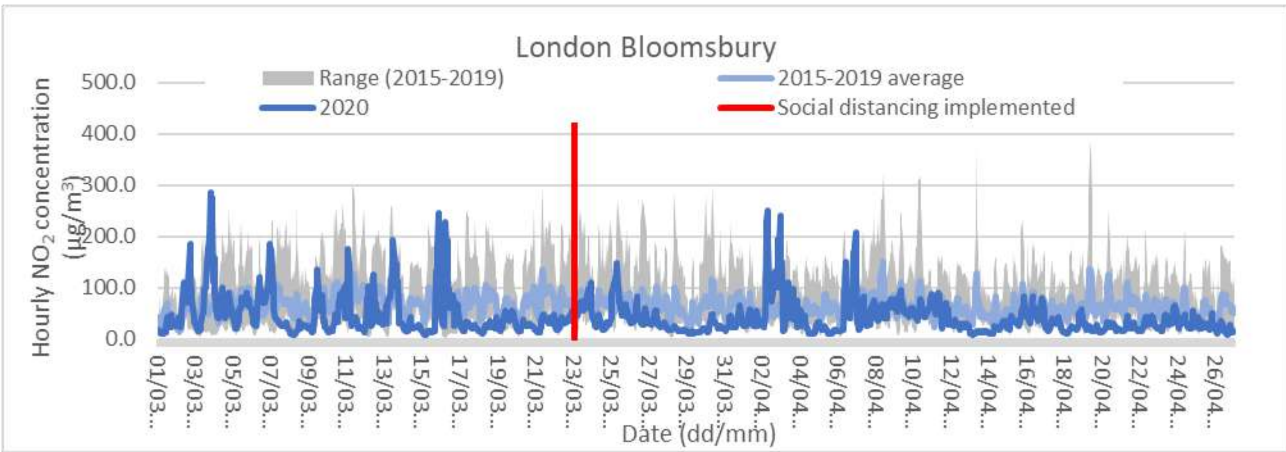


Figure D9: Monitored NO_2 concentrations at London Eltham (suburban background location)

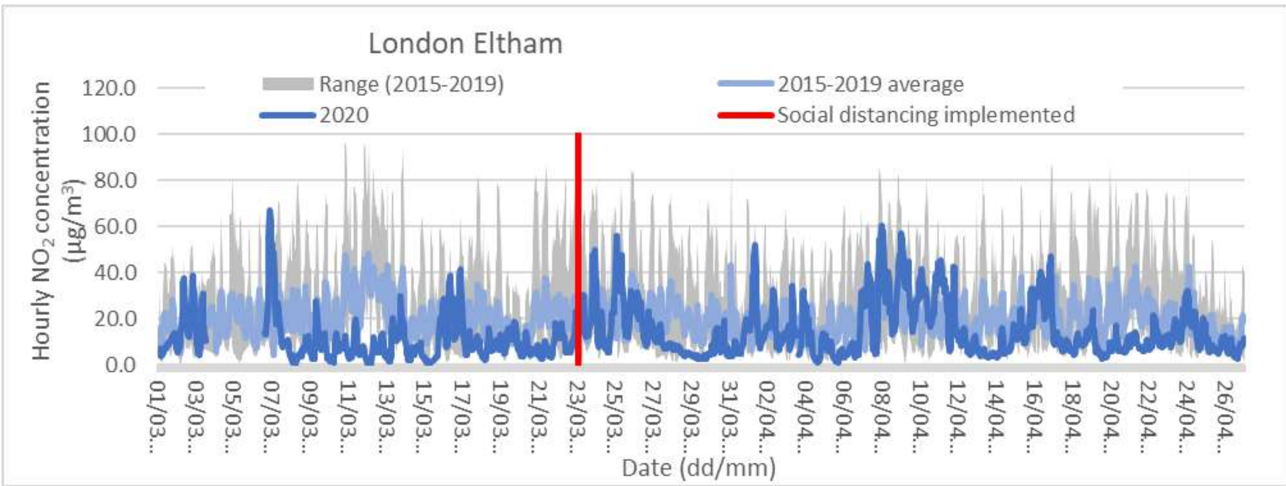


Figure D10: Monitored NO_2 concentrations at Rochester Stoke (rural background location)

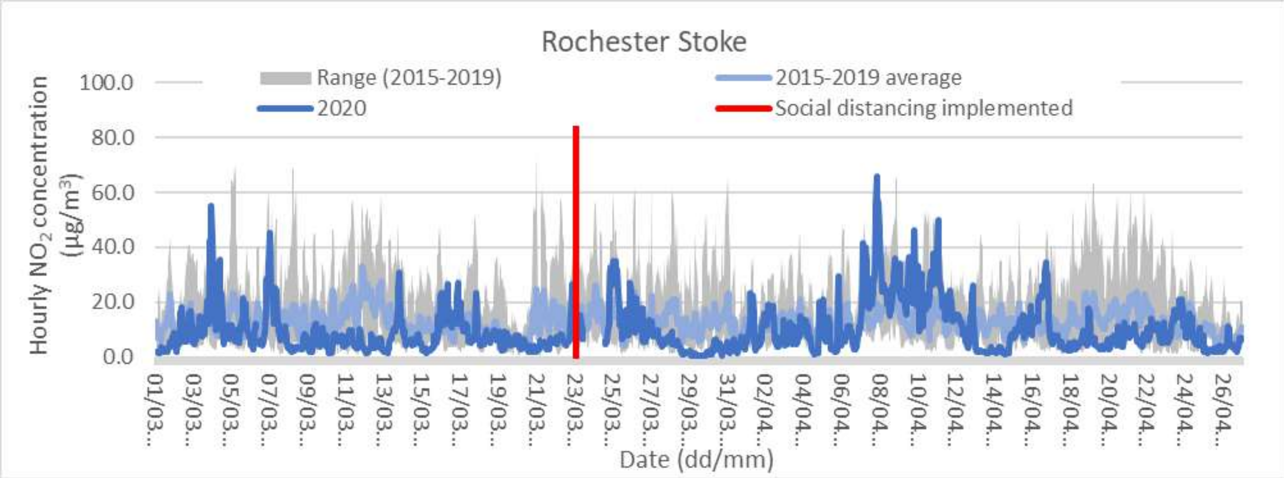


Figure D11: Monitored NO_2 concentrations at Horley (suburban industrial/airport location)

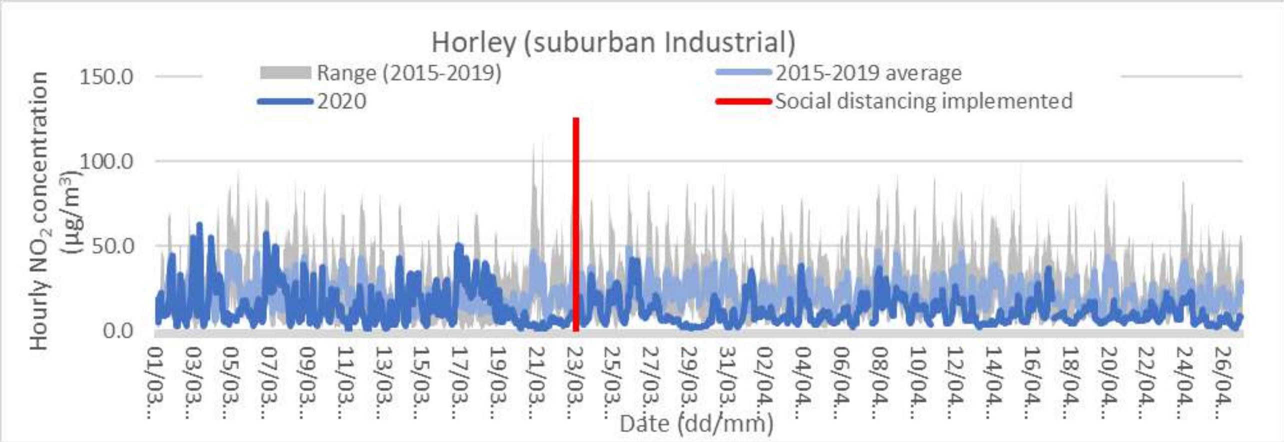
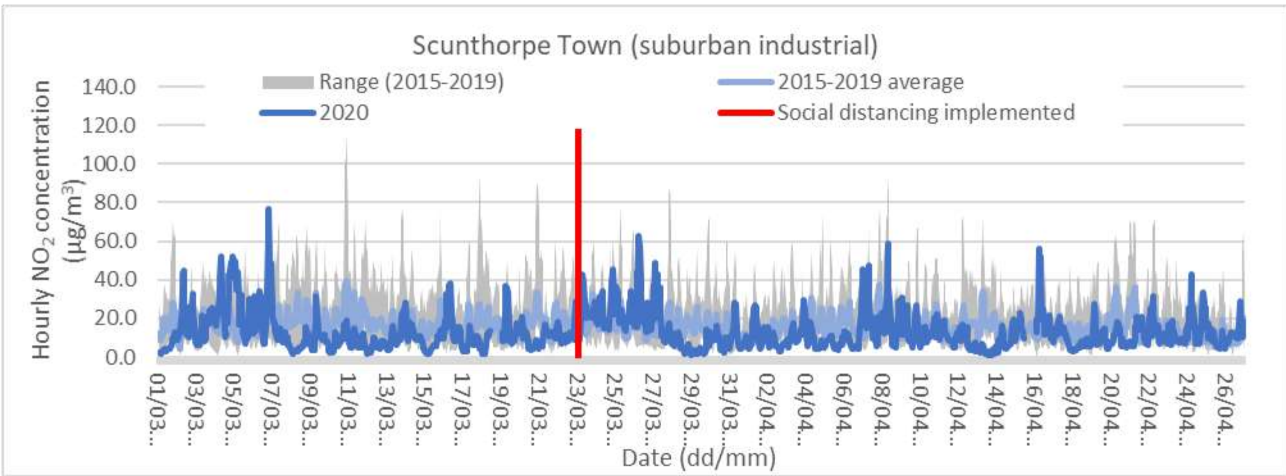


Figure D12: Monitored NO_2 concentrations at Scunthorpe Town (suburban industrial location)



PM

Figure D13: Monitored PM_{10} concentrations at Marylebone Road (roadside location)

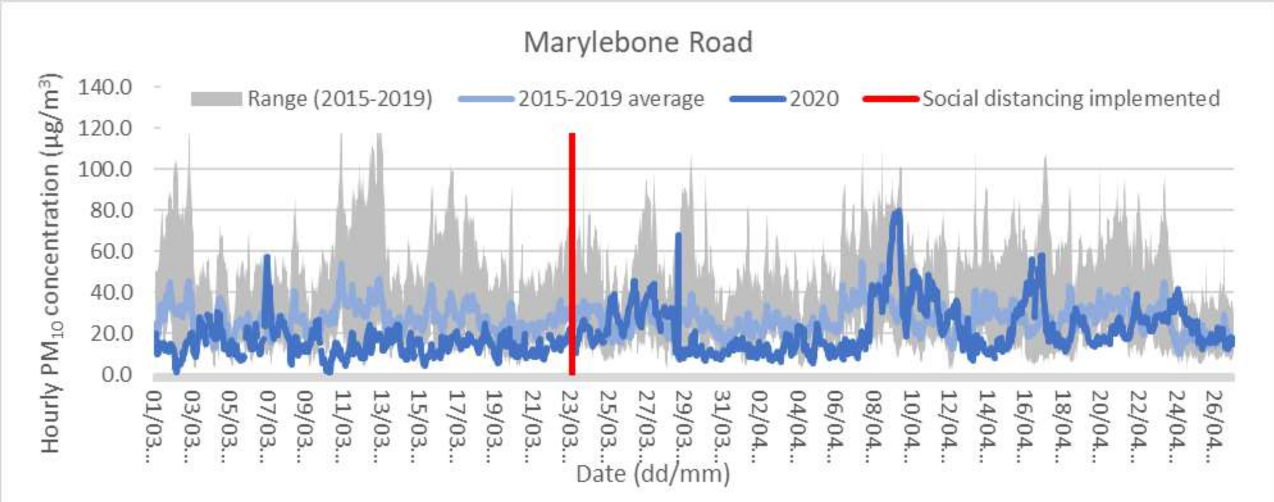


Figure D14: Monitored PM_{10} concentrations at London Bloomsbury (urban background location)

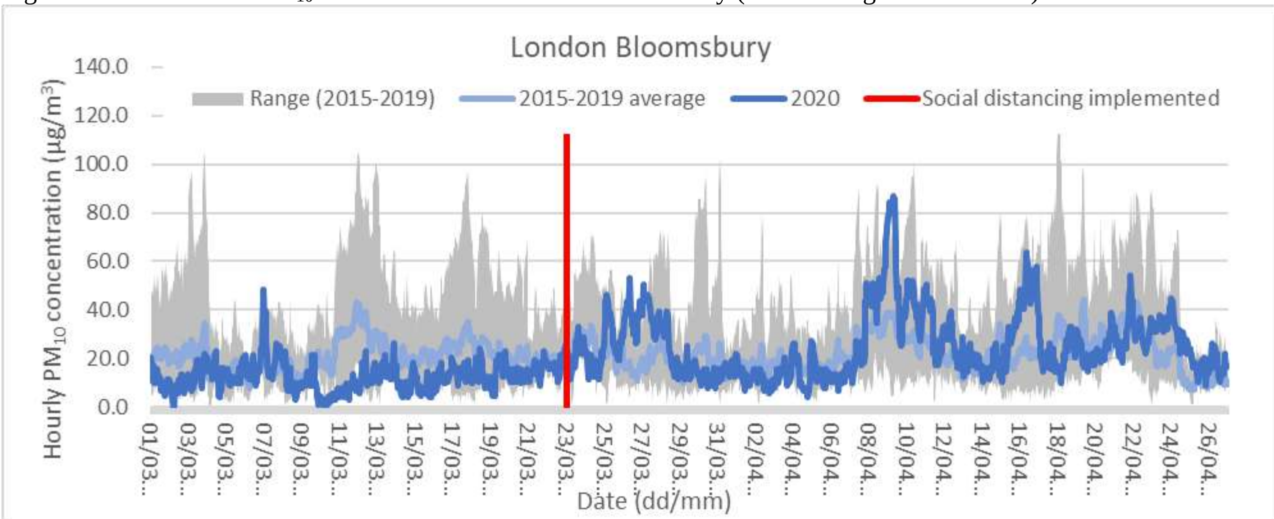


Figure D15: Monitored $PM_{2.5}$ concentrations at London Eltham (suburban background location)

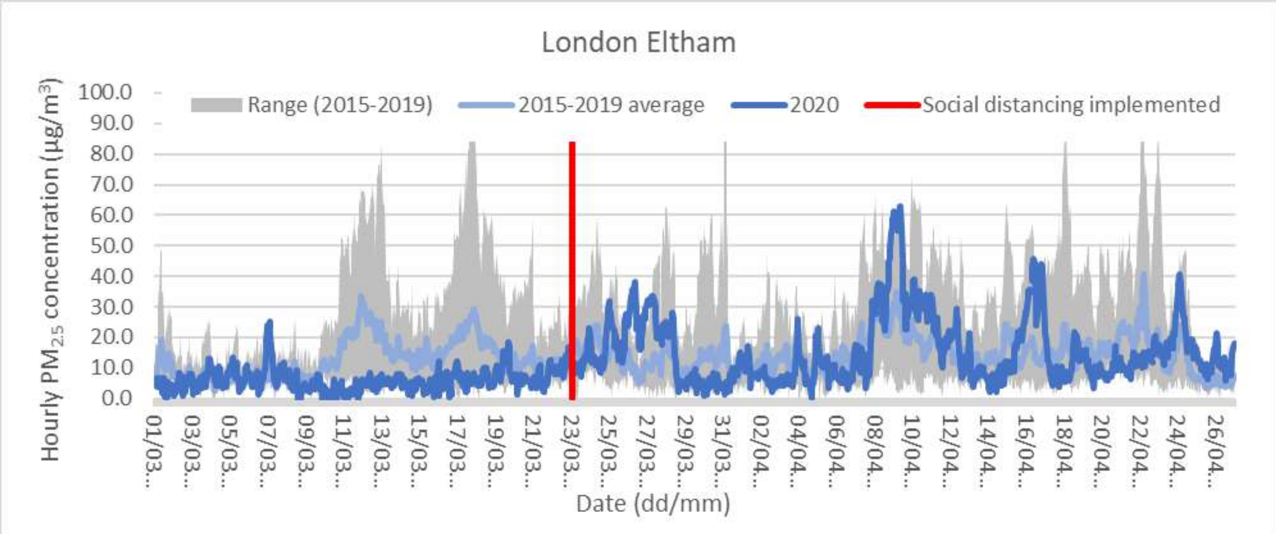


Figure D16: Monitored PM_{10} concentrations at Rochester Stoke (rural background location)

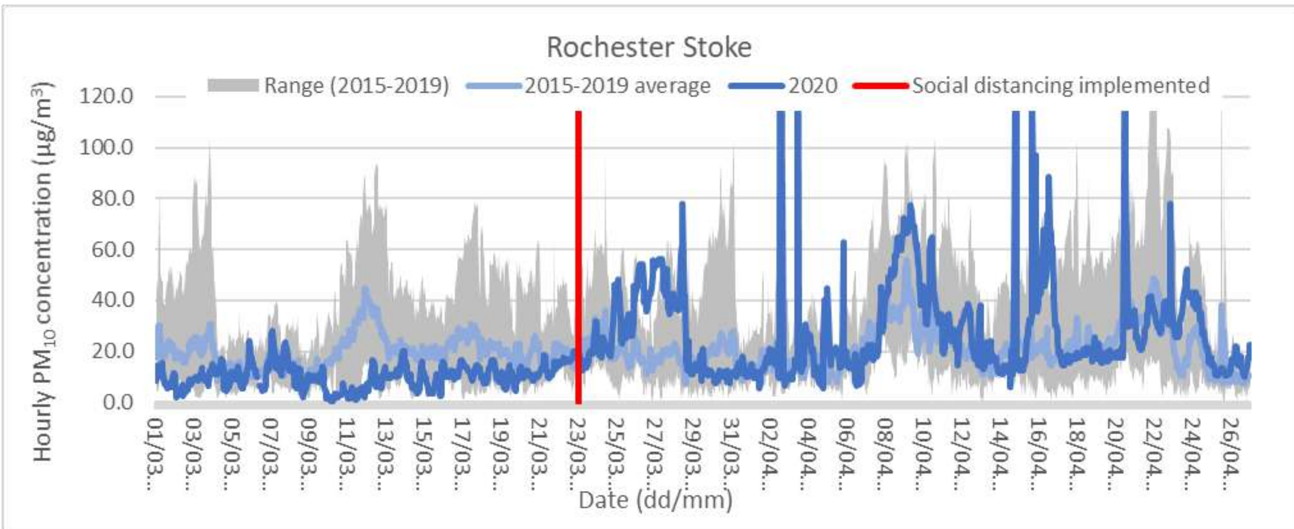
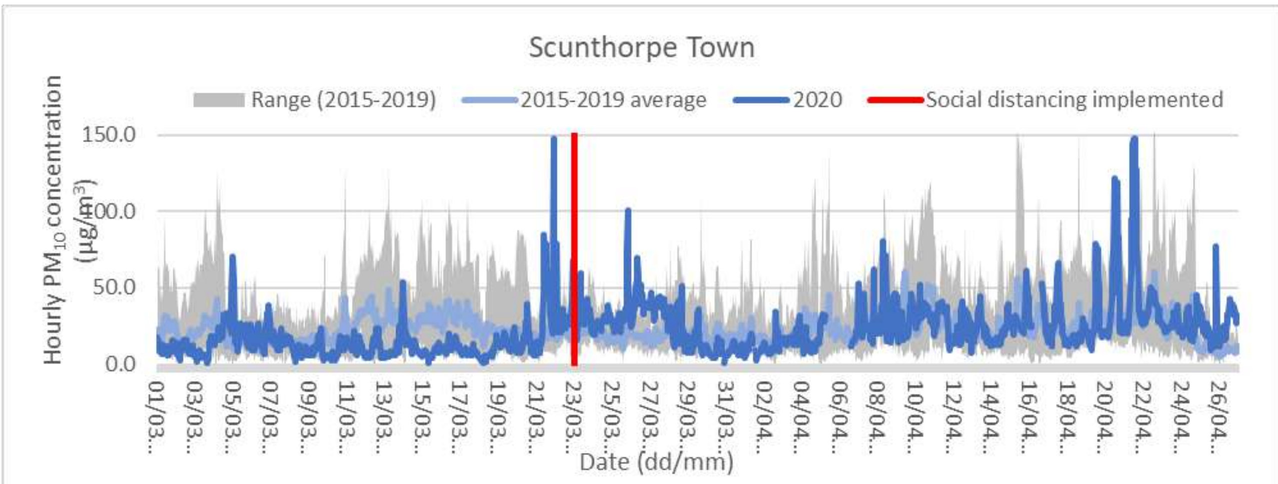


Figure D17: Monitored PM_{10} concentrations at Scunthorpe Town (industrial location)



O_3

Figure D18: Monitored O_3 concentrations at Marylebone Road (roadside location)

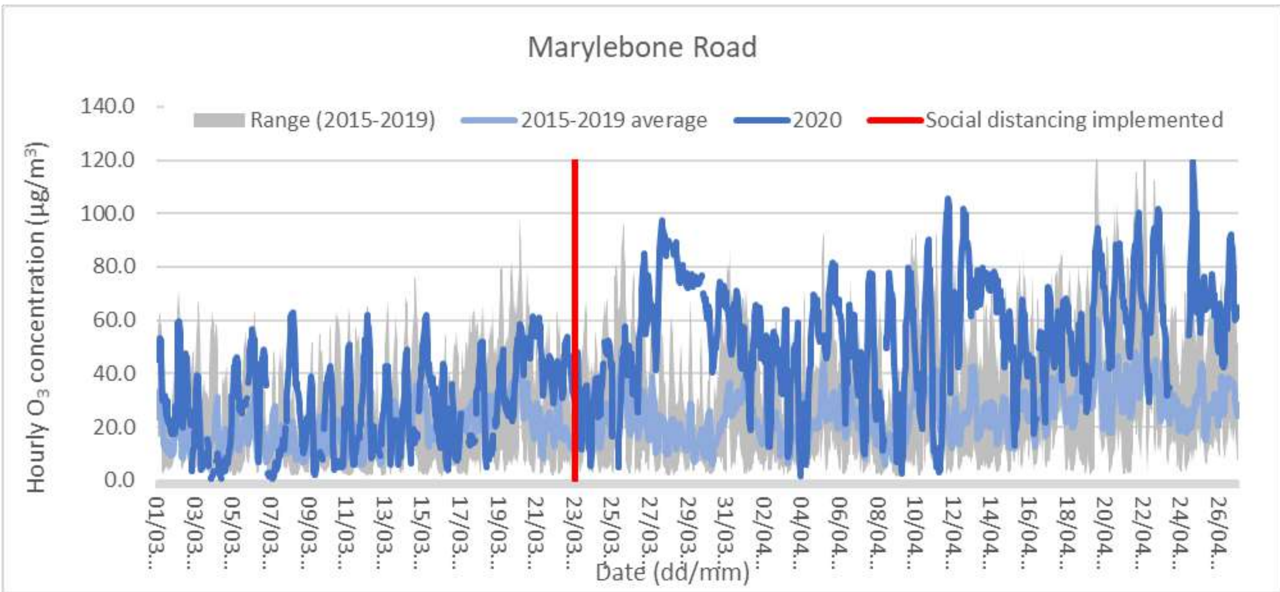


Figure D19: Monitored O_3 concentrations at London Bloomsbury (urban background location)

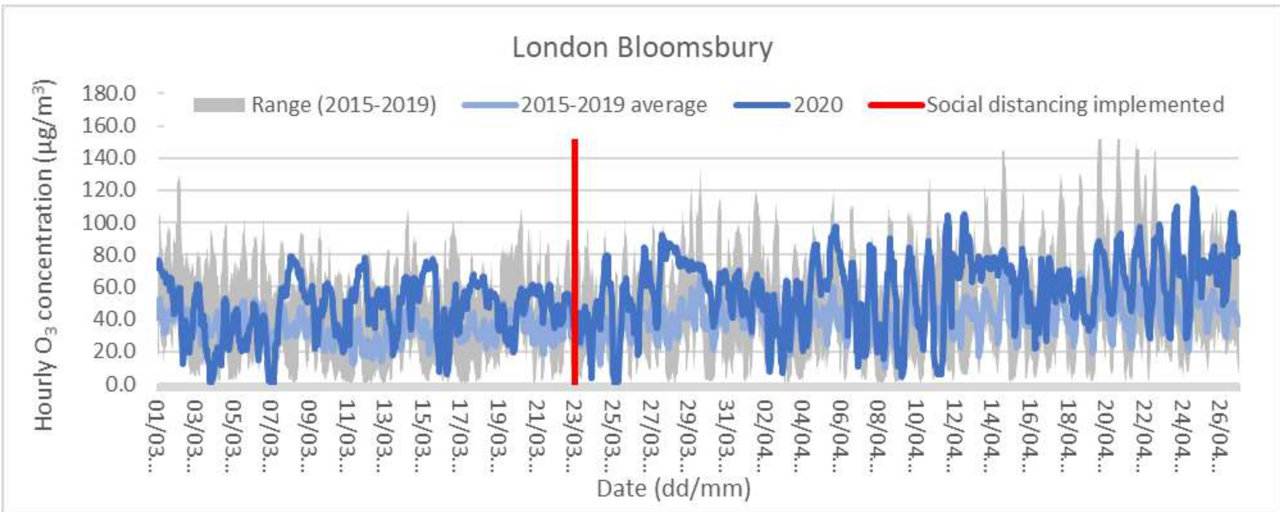


Figure D20: Monitored O_3 concentrations at London Eltham (suburban background location)

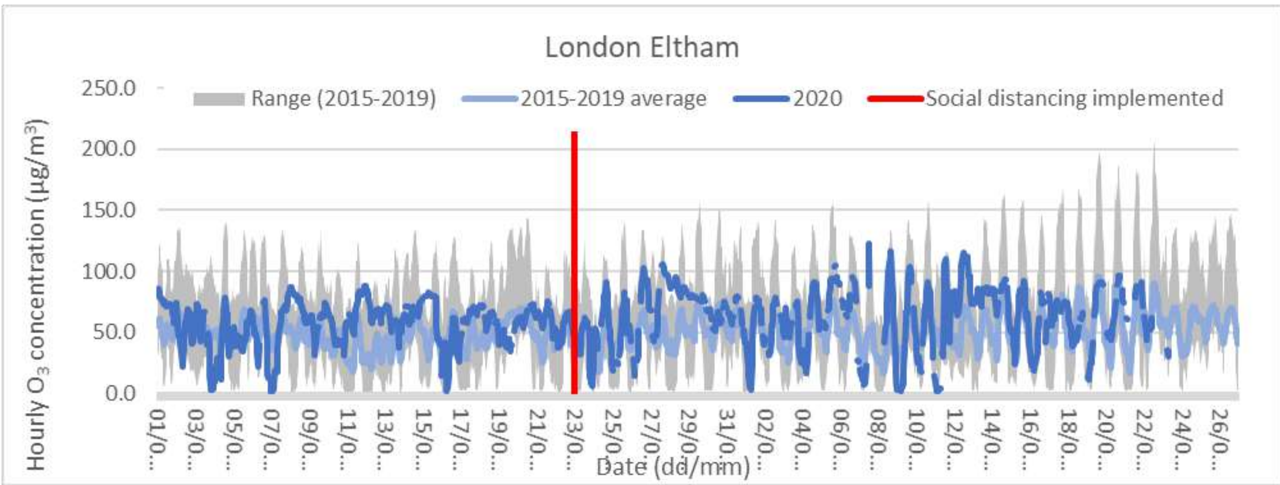
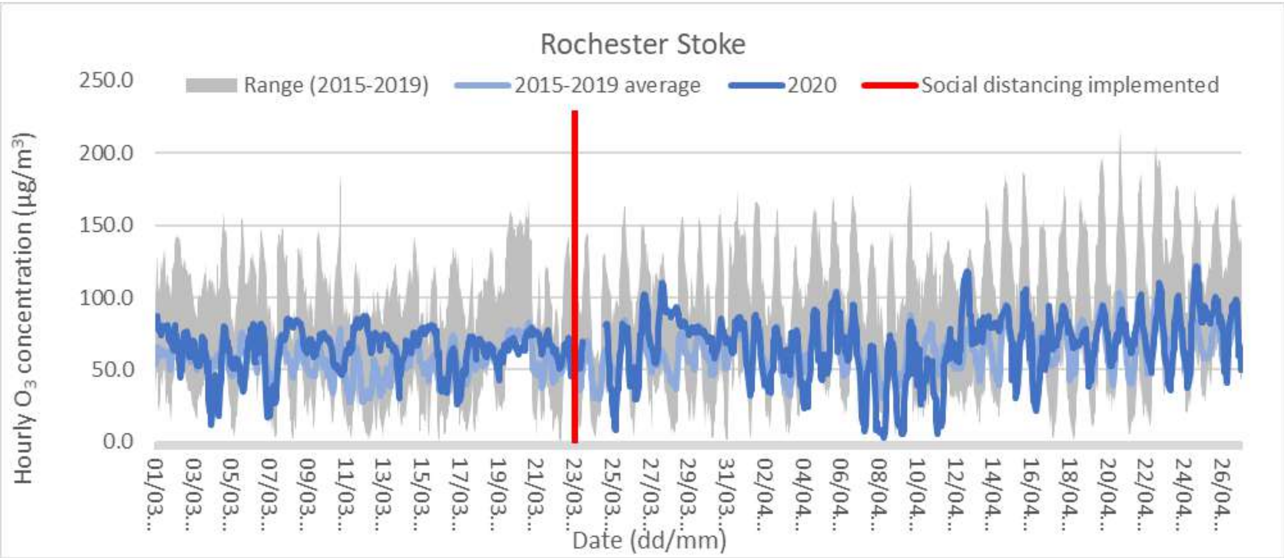


Figure D21: Monitored O_3 concentrations at Rochester Stoke (rural background location)



VOC (benzene)

Figure D22: Monitored benzene concentrations at Marylebone Road (roadside location)

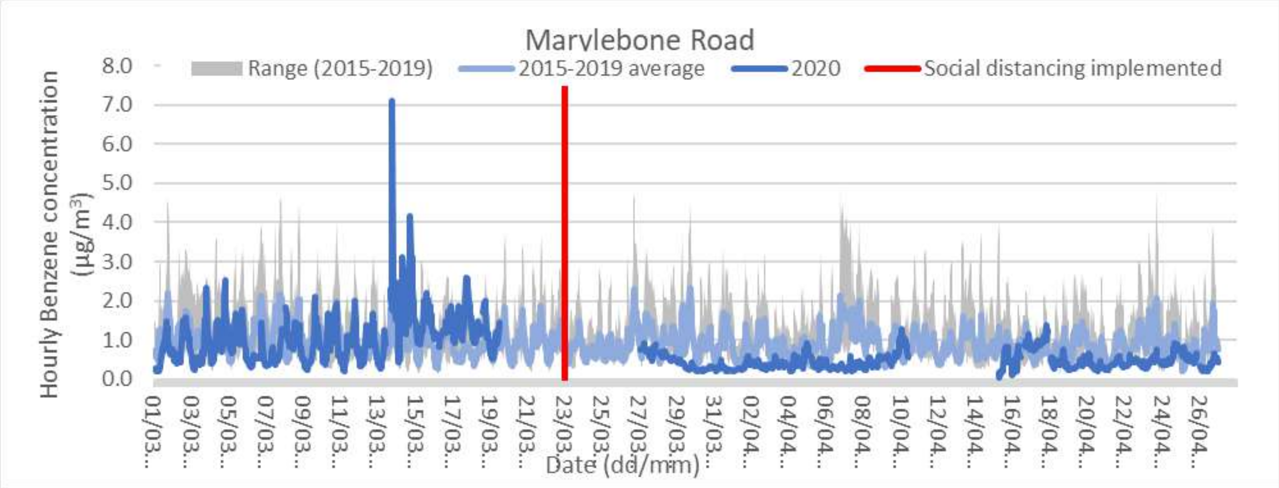
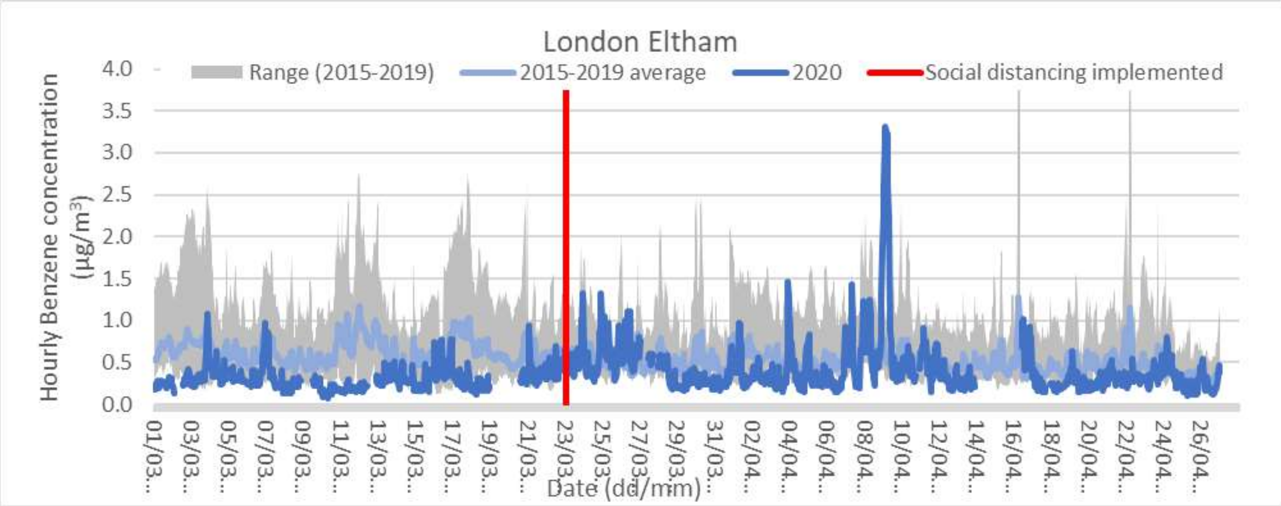


Figure D23: Monitored benzene concentrations at London Eltham (urban background location)



DOCUMENT CHECKING (not mandatory for File Note)

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Response to Defra call for evidence "*Estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK*" on point "*How might altered emissions of air pollutants over the next three months affect UK summertime air quality?*"

Predicting the influence COVID-19 restriction on air quality pollutants for summer 2020 with an atmospheric chemistry transport model.

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Summary
In order to explore the potential impact of COVID-19 restrictions on air pollution this summer, the GEOS-Chem chemistry transport model has been run in a regional configuration over the UK to simulate an air pollution event last summer (24th to 28th August 2019). To understand how these restrictions might impact, the model has also been run with emissions of anthropogenic NO reduced by 10%, 30% and 50%, and the impact on NO_2, $PM_{2.5}$, O_3 and O_x were analysed. COVID-19 restrictions are however likely to influence a wider set of species than just NO emissions. CO, VOCs, dust etc are all likely to be reduced by reduced travel. However, there is also the potential for other species to increase. Domestic burning for example is likely to have increased in this period as people burn gardening waste and enjoy cooking outdoors. At this resolution, the model results are reflective of regional-scale responses to change in emissions rather than road-side changes.

We find:

- reductions in anthropogenic NO emissions lead to roughly similar reductions in NO_2 concentrations
- small changes in O_3 concentrations from reduced anthropogenic NO emissions. Changes reflected both a repartioning of O_x into O_3 due to reduced NO emissions, and the potential for both increased and decreased O_3 photochemical production due to reduced NOx concentrations.
- small changes in PM2.5 concentrations due to reduced anthropogenic NO emissions, as aerosol phase nitrate is a relatively small component of summer time PM2.5 and it is likely limited by available ammonia in most places.

Introduction

In order to explore the potential impact of COVID-19 restrictions on 2020 summer time UK air quality, a 2019 event is explored. Between the 24th and 28th of August 2019, a widely dispersed air quality event occurred over the UK with increased concentrations seen for NO_2, O_3 and $PM_{2.5}$. We use this event as a proxy for what might happen during a 2020 summer time air quality event in the UK if emissions (notably transport emissions) are reduced due to COVID-19 restrictions. Figure 1 shows back trajectories ending over the UK at noon on the 25th August 2019. This shows the role of long-range transport in this event.

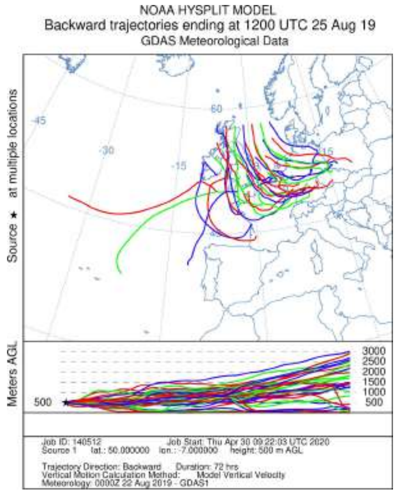


Figure 1. *Back trajectories ending over the UK at noon on the 25th August. This shows the long range transport of pollutants from the continent to the UK and then flow over the UK north / north westward.*

We focus solely on changes in anthropogenic NO emissions and run additional simulations with total anthropogenic NO reduced by 10, 30 and 50%. In order to perform these simulations we use the GEOS-Chem model run in a regional configuration for this event. We start simulations on the 9th of August, and consider the first 6 days the spin up period. The first 9 days (15th to 24th) analysed represent background conditions, with the pollution event occurring from the 24th to the 28th of August. Details of the model, and a comparison with the observations are provided in the Modelling and Evaluation section later in this report. Given the spatial resolution of the model (~30km) results are indicative of changes to the regional concentration of pollutants rather.

Results

Figure 2 shows the average model concentrations of pollutants (NO_2, O_3, O_x and PM2.5) calculated for the location of the AURN urban background stations (see Appendix for sites used) for simulations with unchanged NO emissions and with 10, 30 and 50% reductions in NO emissions. Figure 3 shows these same concentrations as a fractional change from the base simulation. Figure 4 shows the change in surface concentrations for the 4 pollutants across the UK at noon on the 25th August 2019 , and Figure 5 shows the equivalent fractional change.

Impact on NO_2

With decreased NO_x emissions, NO_2 concentrations at the sites reduce (Figures 2). We find the NO_2 concentration responds with roughly the same fractional decrease as the NO_x emissions decrease - a 10% reduction in NOx emissions leads to a roughly a 10% reduction in NO_2 concentrations (Figure 3). There is some non-linearity in the response. A 50% reduction in NO_x emission leads to a slightly smaller than 50% reduction in the NO_2 concentrations (Figure 3). This probably reflects non-linearities in the chemistry and non-anthropogenic source of NO_x within the UK model domain, together with transport of NO_x species from outside of the domain. These reductions are felt across the UK (Figures 4 and 5). Remote locations show smaller changes as the influence of the boundary conditions are larger here. Overall, it would seem likely that a reduction in NO emissions due to COVID-19 restriction would result in a fractionally consistent reduction of NO_2 concentrations for most locations in the UK. .

Impact on O_3 and O_x

During the background period (15-23rd), reduced NO emissions leads to small average increases in O_3 concentrations during the night time, and small decreases at day concentrations (Figures 2 and 3). Changes in O_x (NO_2+O_3) are even smaller, with almost no change in O_x during the nighttime and small changes during the day (a 50% drop in NOx emissions leads to a 4% drop in day time O_x). The small decreases in daytime O_3 and O_x, likely reflects small reductions in net O_3 production due to lower NO concentrations. The increase in night time O_3 reflects a re-partitioning of O_x for NO_2 to O_3 under lower NO emissions.

During the polluted period, O_3 and O_x concentrations increase. During the night time O_3 concentrations increase with reduced NO emissions, whereas O_x concentrations remain essentially unchanged. We interpret this again as a rebalancing of O_x from NO_2 to O_3 as NO emissions lower. There is little change in peak O_3 concentrations under the different NO emissions but if anything there is a reduction in mean O_x and O_3 concentrations with reduced NO emissions. The model simulations used here use the same boundary conditions regardless of the change in NO in the domain. The model domain (see modelled section below) contains much of France, Netherlands, Ireland and Belgium but does not contain Germany or other parts of industrial Europe. These results highlight the significance of trans-boundary pollution as mean O_3 and O_x concentrations barely change despite significant changes in UK (and French, Belgium and Netherlands) emissions. It appears that much of the O3 seen in this period originates from outside of the domain (See Figure 1).

Some of this lack of change may also reflect regional differences in the model response cancelling out. Figures 4 and 5 show the change in O_3 and O_x concentrations under reduced NO emissions on the 25th August at noon. Here we see reductions of a few ppbv (10%) in noon time O_3 concentration in some places (London, South East, Scotland) but increases of a similar magnitude in others (Midlands, Yorkshire, South Wales coastal regions with heavy shipping). Very large changes are seen around Dover due to reduction in NO emissions associated with shipping. Changes in O_x are more muted. These changes in O_3 (and O_x) reflect the differing influence of a reduced NO emissions. If O_x is conserved, reducing NO emissions will lead to higher O_3 concentrations as less O_x is composed of $NO_{2.}$ In VOC sensitive regions decreasing NO emissions will decrease O_3 concentrations as less NO will be available to react with $RO_{2,}$ in NO_x sensitive regions, decreasing NO emissions will increase O_3 concentrations as less NO_2 is available to react with OH. Different days (not shown) during the pollution event show different geographical patterns of O_3 increases and decreases reflecting the interplay of meteorology, emissions and chemistry.

It would seem likely that any response to changing NO emissions due to COVID restrictions would likely show regional differences in the magnitude of response. It would also depend on the time of year, the prevailing weather conditions and also likely whether emissions of compounds other than NO_x (CO, VOCs etc) also changed. Thus making predictions about the likely influence of these restrictions on O_3 concentrations during the summer months is difficult.

Impact on $PM_{2.5}$

Model concentrations of during the background period are essentially unchanged by reductions of NO emissions (Figures 2 and 3). The most direct mechanism for NO emissions to influence PM2.5 is through changing aerosol nitrate concentrations. Nitrate consists of roughly 20% of the aerosol mass at this time of year (for London based on ClearFlo results) , and so any change in the emissions of NO will likely lead to a much smaller overall change in the PM2.5 mass. Ammonia emissions will also limit the amount of nitrate that can be in the particle which will again dampen the response of a change in emission.

During the polluted period the model shows more sensitivity. At noon time there are relatively small increases in PM2.5 which maximize at 5% or less. However, significant changes are seen during the evening and early morning periods, with lower NO emissions leading to lower PM2.5 production. This may reflect reduction in the nighttime N_2O_5 route to produce nitrate, and the lower temperatures at night which may change the ammonia nitrate partitioning into the aerosol phase. However, further work is necessary to explore this.

Over most of the UK, day time PM2.5 does not change significantly with reduced NO emissions (Figure 4 and 5). There is some suggestion that away from urban centres there is a larger change which may indicate ammonia control in some locations (close to ammonia sources) and nitrate (HNO_3+NO_3^-) control (far from ammonia sources notably over the oceans).

In general reduced NO_x emissions would appear to lead to only small changes in PM2.5 concentrations for most of the UK due to nitrate only playing a relatively small role in total PM2.5 at this time of year, and the limits on nitrate concentrations provided by ammonia emissions.

Organic aerosol concentrations may change under decreased NO emissions due to changes in oxidants (OH, NO_3, O_3), with increased oxidants leading to increased SOA production. This has been suggested in China, in a preprint which suggests increases in secondary aerosols due to increased oxidant concentrations [Huang et al., 2020].

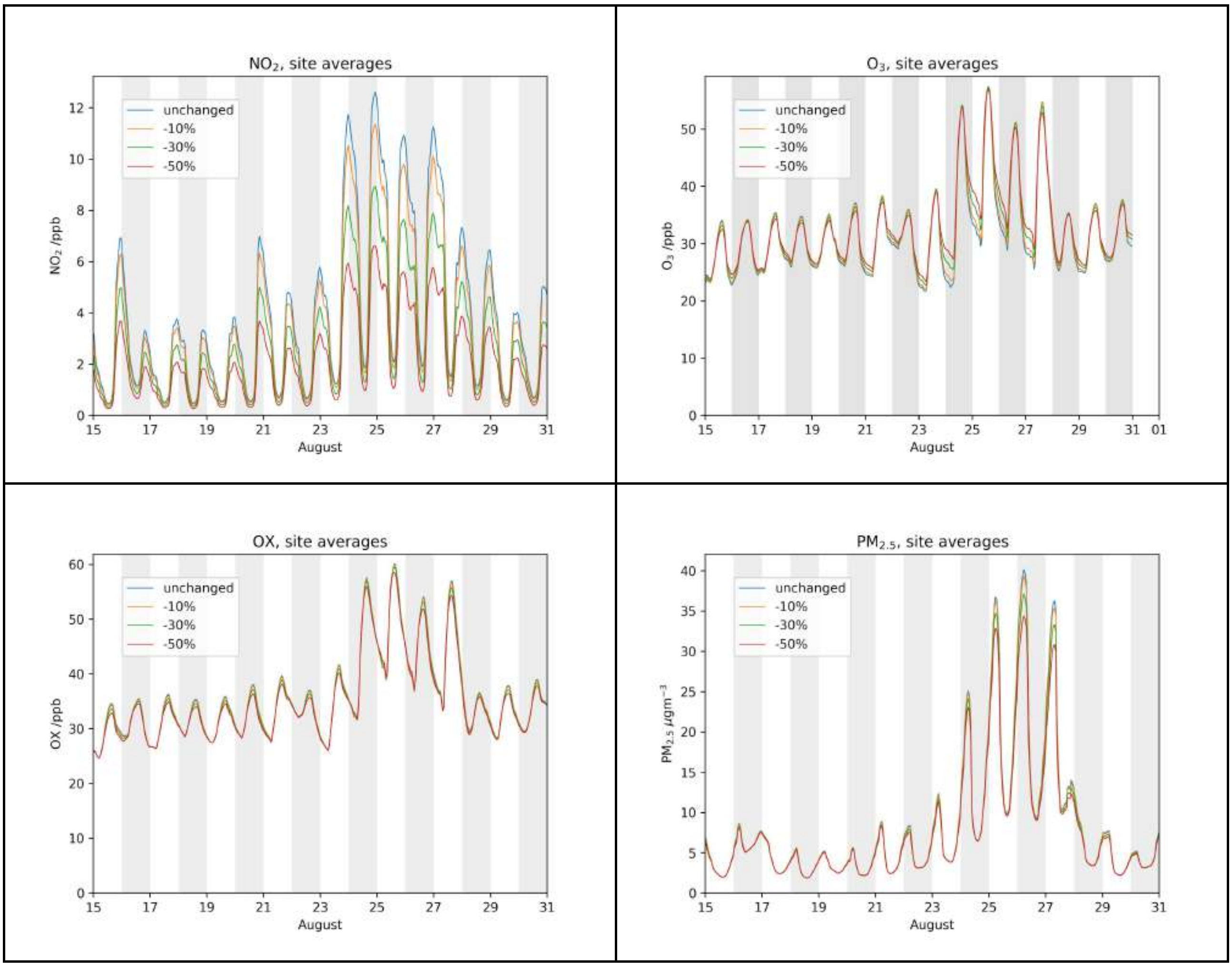


Figure 2. Model response for NO_2, O_3, O_x and PM2.5 under different NO_x emissions reductions averaged for all UK urban background sites (see Appendix 1 for list of sites). The lines show the mean concentration at those sites for the standard emissions (blue) and reduction of 10 (yellow), 30 (green) and 50 (red) % respectively.

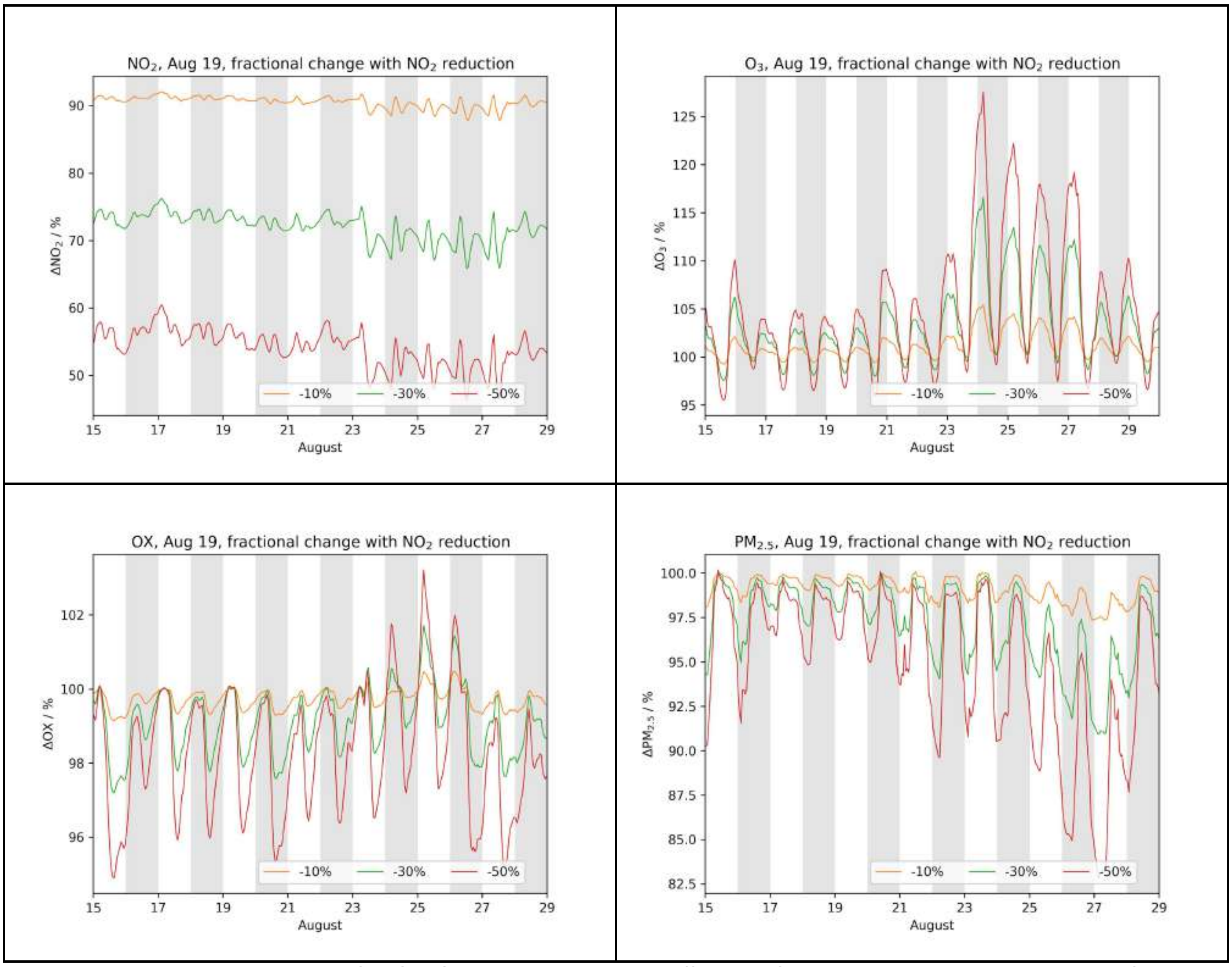


Figure 3. Fractional change in NO_2, O_3, O_x and PM2.5 under different NO_x emissions reductions averaged for all UK urban background sites (see Appendix 1 for list of sites). The lines show the fractional change in the mean concentration at those sites reductions of 10 (yellow), 30 (green) and 50 (red) % respectively.

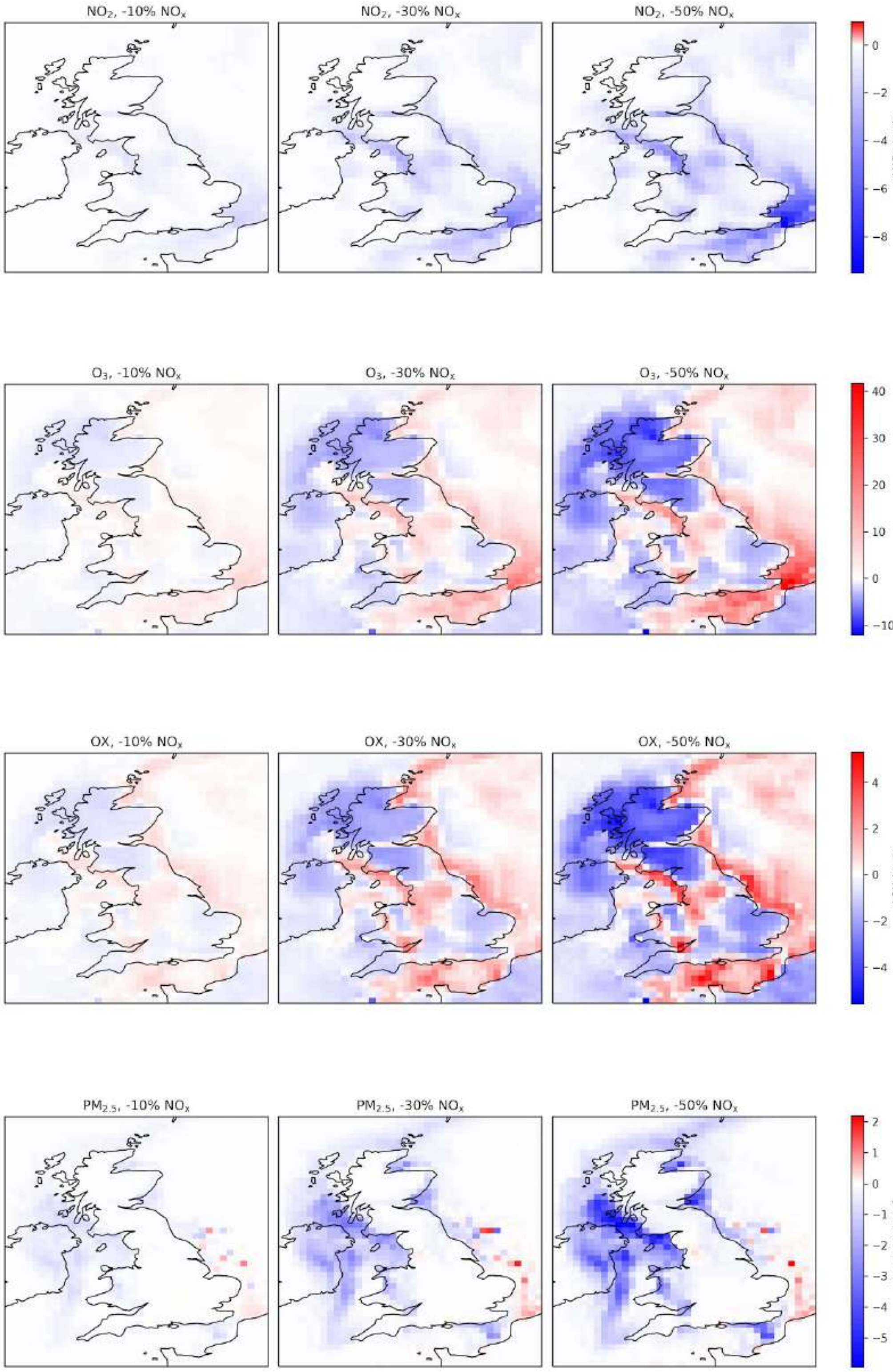


Figure 4. *Maps of the changes in the surface concentration of pollutants with 10, 30 and 50% reduction in the emissions of anthropogenic NO at noon time on the 25th of August.*

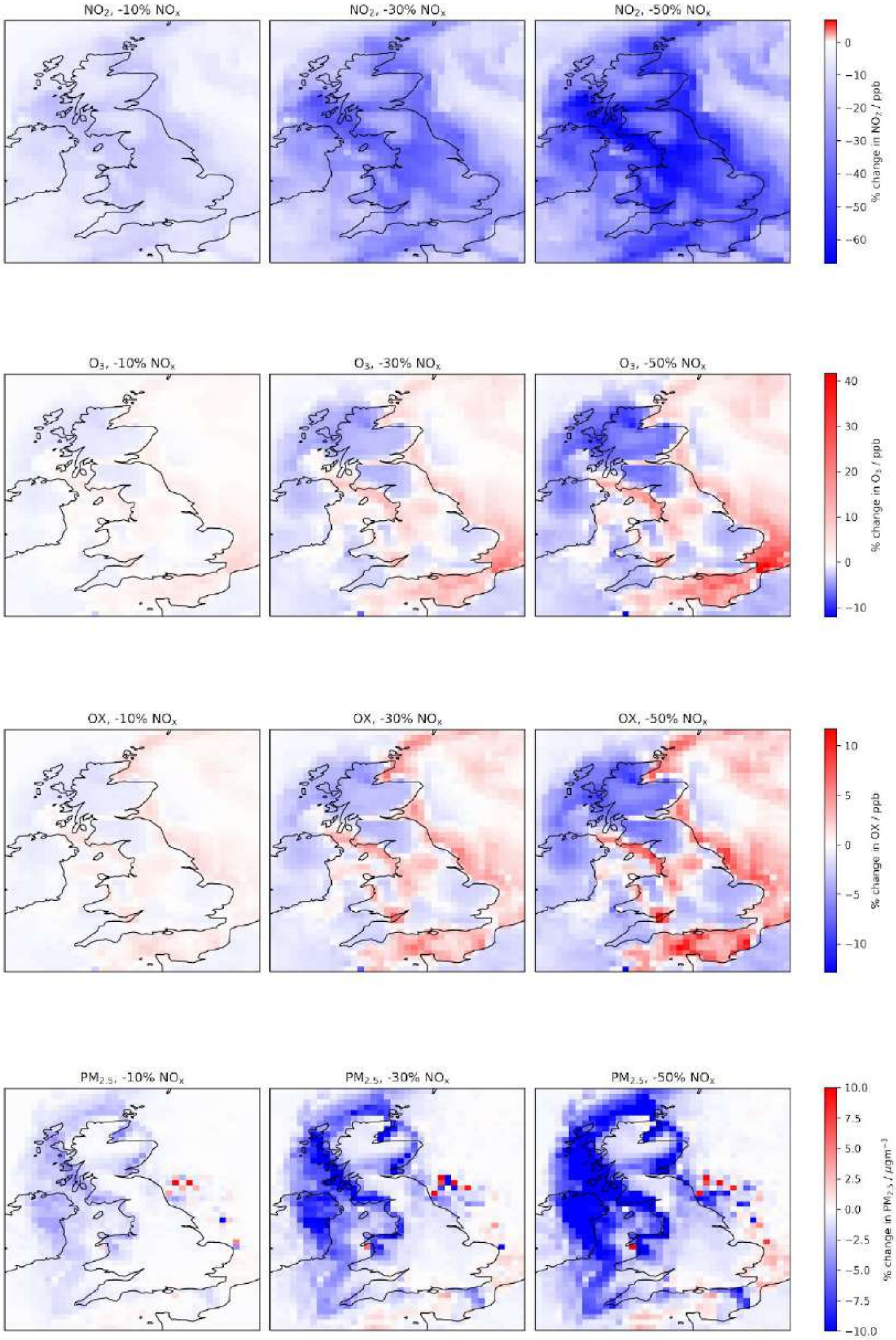


Figure 5 *Maps of the fractional change in the surface concentration of pollutants with 10, 30 and 50% reductions in the emissions of anthropogenic NO at noon time on the 25th of August.*

The model

We have used version 12.6.0 of the GEOS-Chem model (www.geos-chem.org), a well used open source offline (uses prescribed meteorology rather than calculating it itself) atmospheric chemistry modelling tool, to evaluate a number of simulation exploring the impact of COVID-19 restrictions in the UK.

Domain

The model has been run over a domain (from 45°N,15°W to 65°N,5°E) covering the UK, Northern France, Benelux (Figure 6). The grid resolution is 0.25° in the N-S direction (~27km) and 0.3125° in the E-W direction (~35km).

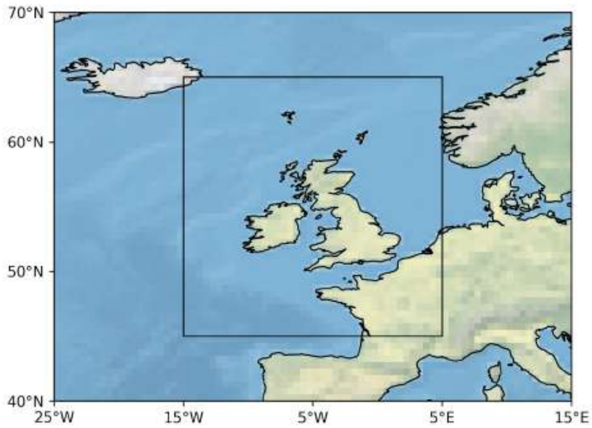


Figure 6*. Inner box indicated the model domain for the higher resolution (0.25°x0.3125°) modelling.*

Boundary conditions from the modelling have come from a coarse resolution (4°x5°) GEOS-Chem simulation with the same emissions over Europe as the fine resolution, and the default GEOS-Chem emissions for the rest of the globe. The same boundary conditions are used for all simulations.

Emissions

The model uses the EMEP spatially resolved emissions for the anthropogenic emissions within the European domain. These incorporate the NAEI estimates through the EMEP process. The last year of available data is 2017. Seasonal, weekly and diurnal variability of emissions is represented through factors sourced from EMEP. More details of other emissions used in the model can be found here.

Transport

The model uses offline assimilated meteorology from NASA Goddard's Global Modelling and Data Assimilation (GMAO) GEOS-5 system.

Chemistry and aerosol processes

The model chemistry contains a complete description of the atmosphere's O_x-HO_x-NO_x inorganic chemistry. It also contains a simplified chemistry of the atmosphere's organic chemistry (Ethane, Propane, Alkenes, Butane and greater alkanes, isoprene) and a representation of tropospheric halogen chemistry.

The model aerosol considers sulfate, nitrate, ammonium aerosol (aerosol thermodynamics provided by the ISOROPIA-II scheme), sea-salt (coarse and accumulation modes), organic and black carbon (hygroscopic and hygrophobic) and dust (4 bin sizes),

Deposition

Both wet and dry deposition of atmospheric constituents is considered.

Simulation period.

The model is run from 9th of August to the 31th August with the first 5 days considered spin up.

Model performance

The GEOS-Chem mode has been used previously to explore UK air quality [Dunmore et al., 2016, Sherwen et al., 2017] but its main application in the past has been to North American or Chinese air quality issues. Further work is necessary to assess the model's performance for the UK and this will be conducted as Luke Fake's PhD project. Some initial work validation work for the August period used in this case study is presented here.

Figure 6 shows a comparison of the base model simulation and AURN urban background sites for O_3, NO_2, and PM2.5. All AURN background urban sites reporting observations are included in the analysis (see appendix 1). The model is sampled at the urban background sites and then averaged. Future work will need to explore the performance of the model more and assess against a wider suite of observations.

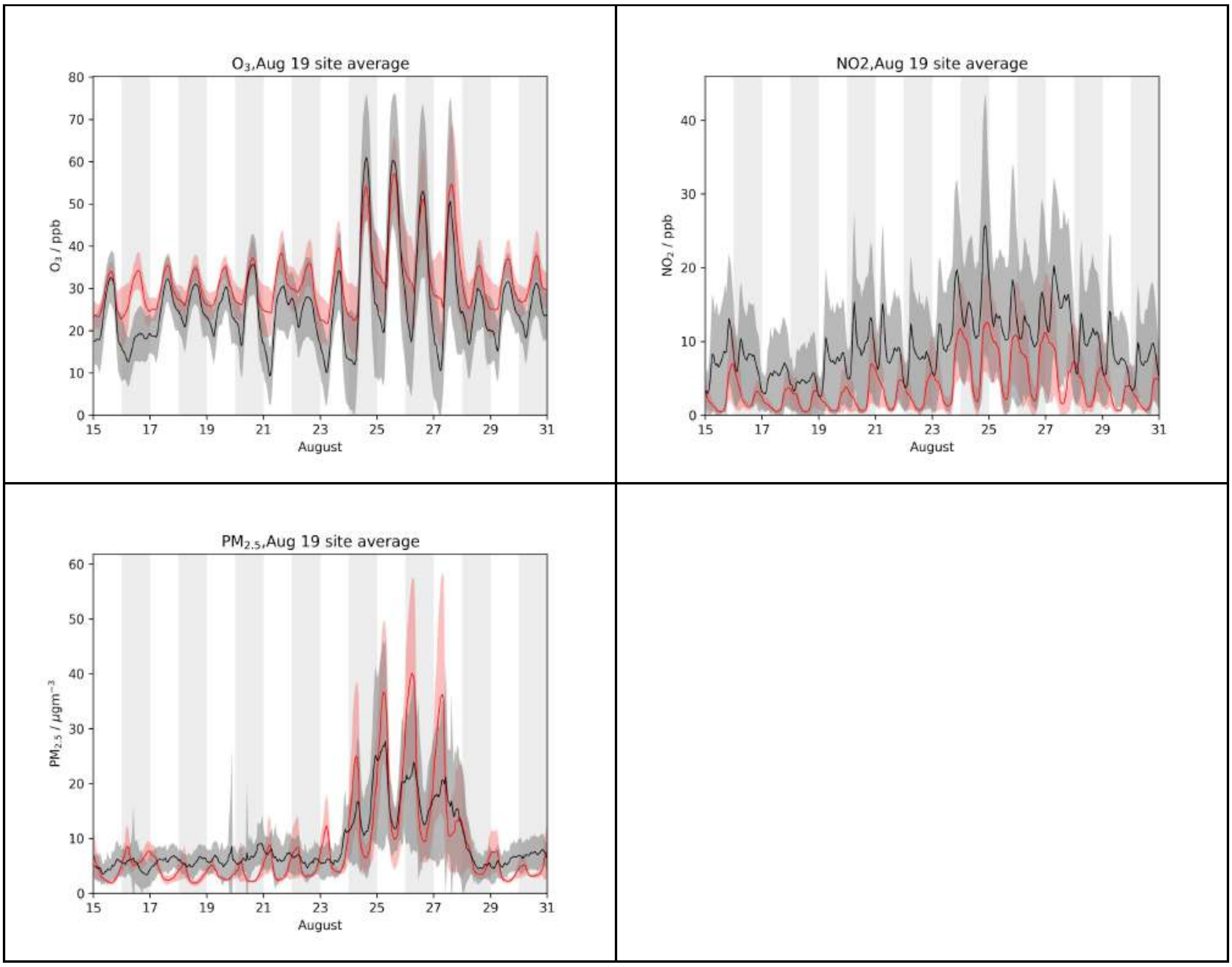


Figure 6. *Comparisons between the base model performance and observations for O_3, NO_2 and PM2.5. Observations (black) show the mean AURN value for all UK urban background sites (see Appendix 1). Model (red) shows the equivalent means for model output at all of the urban background sites. The shaded areas represent the* ± 1 standard deviation from the mean *of the measurements or models across the sites.*

O_3 concentrations follow the general pattern of the observations. During the background period they are slightly overestimated but follow the trend during the polluted period reasonably well. The model fails to capture the drop of O_3 during the nights. This may be indicative of an underestimate of NO_x emissions (see discussion of NO_2) at night which would tend to convert O_3 into NO_2.

Both modelled and measured NO_2 concentrations show increases during the event which is qualitatively captured by the model. However, NO_2 concentrations are significantly underestimated by the model especially during the day. There are a number of potential explanations for this. The molybdenum converters used in the measurement of NO_2 often cause a high bias in observations as they also lead to the measurement of a fraction of other NOy compounds (HNO_3, PAN etc). However, given the relatively long lifetime of NO_y this doesn't provide a strong explanation for the model failure which is predominantly based around an exaggerated diurnal cycle. Urban background sites may be unrepresentative of the model grid-boxes they are being compared against. At a resolution of ~25km there is significant heterogeneity in a background urban gridbox, and these sites may be sampling more polluted air than the model gridbox average This may reflect errors in the diurnal emissions profile of NO in the model which may be too low in the middle of the day. An underestimate in the NO emissions would help to explain the over-estimate in O_3 during the background period.

Simulated PM2.5 concentrations follow the same trend as the observations with higher concentrations during the pollution event. They tend to show an enhanced diurnal cycle compared to the observations with higher concentrations at night than observed. Further work looking at the change in the individual speciation of the PM2.5 is necessary to understand what is occurring here.

Appendix.

Urban Background sites used in the analysis:

Aberdeen, Belfast Centre, Birmingham Acocks Green, Blackpool Marton, Bournemouth, Brighton Preston Park, Canterbury, Cardiff Centre, Coventry Allesley, Cwmbran, Derry Rosemount, Edinburgh St Leonards, Glasgow Townhead, Hull Freetown, Leamington Spa, Leeds Centre, Leicester University, London Bloomsbury, London Haringey Priory Park South, London Hillingdon, London N. Kensington, Manchester Piccadilly, Newcastle Centre, Norwich Lakenfields, Nottingham Centre, Peebles, Plymouth Centre, Portsmouth, Preston, Reading New Town, Sheffield Devonshire Green, Southampton Centre, Southend-on-Sea, Stoke-on-Trent Centre, Sunderland Silksworth, Thurrock, Walsall Woodlands, Wigan Centre, Wirral Tranmere

THE EFFECT OF COVID-19 LOCKDOWN MEASURES ON AIR QUALITY IN LONDON IN 2020

A note from the Environmental Research Group, King's College London

16 May 2020

KEY MESSAGES

- NO_2 concentrations reduced significantly at busy roadside sites due to reductions in traffic flows of ~53% across London and over 60% in the central area; reductions in average NO_2 concentrations at two busy roadside sites (Marylebone Road and Euston Road) were 55% and 36% respectively. Overall, the mean reduction in hourly NO_2 concentrations were 21.5% across the London roads. The reductions are the difference between the average concentration from 1 January to 12 March and that from 24 March to 22 April.
- Reductions in NO_2 were smaller at outlying roadside sites and at urban background sites. The reduction in average NO_2 at North Kensington was 22% and the mean reduction across all urban background sites was 14%.
- PM_{10} and $PM_{2.5}$ concentrations were higher after lockdown than at any time in 2020 to date, due to several pollution episodes driven by anticyclonic easterly flows suggestive of long-range transport.
- These high PM concentrations are a clear warning that if the UK is to achieve the current WHO $PM_{2.5}$ guideline then as well as actions in the UK, other European countries will need to achieve their emission reduction targets.
- Ozone concentrations were higher post lockdown, partly due to reductions in NO_x but mainly as a result of pollution episodes in easterly anticyclonic air flows. The highest hourly ozone concentration at North Kensington was 129µg/m^3 (~65 ppb) on 24 April.
- Wood burning made a contribution to ambient PM concentrations before and during the lockdown periods. During the lockdown period the evening peak occurred later than in the winter, perhaps reflecting longer daylight hours.
- During the pre-lockdown period (from 12 to 23 March) roadside increments in NO_x, NO_2 had begun to reduce, compared to the January to March average, and continued to reduce in the full lockdown period. This was also true of roadside increments in $PM_{2.5}$ despite the increase in overall $PM_{2.5}$ concentrations.
- Traffic activity as indicated by increases in weekday/weekend ratios of NO_x and NO_2 suggest traffic activity reduced at weekends during the lockdown period. Increases in weekday/weekend ratios of $PM_{2.5}$ and PM_{10} however are probably determined by the occurrence of high PM concentrations mainly on weekdays during the lockdown period.
- The total gaseous oxidative potential (OP) of London's atmosphere, as measured by the sum of ozone and $NO_{2,}$ increased after lockdown. This was due largely to the incidence of ozone episodes post-lockdown; without these, the gaseous oxidative potential would still probably have increased. Decreases in global ozone as well as decreases in regional ozone episodes and in NO_2 concentrations would be required to reduce the gaseous OP of London's atmosphere.
- One would speculate that the overall OP of London's atmosphere (incorporating pro-oxidant particle associated components) will be heavily influenced by the changes in traffic flow post-lockdown, as many of the key drivers of this activity are metals derived from brake and mechanical wear processes. These concentrations are measured at KCL and could be analysed further.
- Modelling the effects of traffic reductions in London, together with changes in travel behaviours due to the lockdown measures, we estimate reductions in [annual average] personal exposures to NO_2 and $PM_{2.5}$ of 18-27% (NO_2) and 5-24% ($PM_{2.5}$) for children, tube users, professional drivers and hospital staff. The largest benefits were for those who reduced their travel, e.g. tube users and the highest exposure to NO_2 was for professional

drivers who continued to work. Spending an extra 1 hour per day in kitchen environments increased exposure to NO_2 by 9-18% and $PM_{2.5}$ by ~19% for everyone.

- These modelled changes arise purely from the model assumptions about traffic, travel and indoor activities in London and could be modified by changes in these pollutants arising from other sources such as transboundary transport of PM. A more comprehensive analysis for the specific lockdown period, combining changes to European, UK and London emissions as well as a wide range of population subgroups could be undertaken as part of a larger study.

1. Introduction

This note summarises the effects of the social distancing measures introduced in mid-March 2020 to inhibit the spread of the Covid-19 virus. The data for sites in London have been analysed, focusing on particulate matter (PM_{10} and $PM_{2.5}$), nitrogen oxides (NO_x) and nitrogen dioxide (NO_2) and Ozone (O_3). It should be noted that since the analyses have been done urgently and the data is not in its final ratified status. Nevertheless, a high degree of confidence is justified as a range of automatic and manual quality assurance procedures have been applied and all data is scaled to standards traceable to national and international standards. Moreover we should stress here that this analysis is a preliminary assessment and a considerable amount of further work cold be done using the large amount of data collected by King's College London.

The lockdown period has been split into two sections, namely the 'pre-lockdown' period from 12 to 23 March (inclusive) when the public were strongly urged to observe social distancing, to limit their movements etc; and the 'post-lockdown' period from 24 March onwards when the measures on social distancing, travel etc., were strengthened.

The effects of the measures in both parts of the lockdown period have been complicated by the long periods of anticyclonic weather and relatively high temperatures (particularly in April). This has resulted in easterly air flows and consequent import of PM from other European countries. Temperatures in April have often exceeded 20ºC and these, combined with easterly flows picking up precursor pollutants, have resulted in photochemical production of ozone with hourly values in the region of 55 – 65 ppb.

The analyses below present time series and other data for sites in the London Air Quality Network (LAQN) where the selection criteria were sites with at least 4 of the 5 pollutants.

2. Nitrogen oxides ($NO_X = NO + NO_2$) and NO_2

The data are analysed by roadside and urban background sites, and we deal with both NO and NO_2 together in each category. NO_x concentrations are dominated by traffic sources in London, clearly at roadside sites but also at background locations. NO_x concentration changes are therefore a direct indicator of the change in emissions from traffic sources during the lockdown periods. Changes in NO_2 concentrations would not be expected necessarily to be as large as those in NO_x because of the non-linear chemistry which governs the NO_x to NO_2 conversion. As can be seen from the plots in Figure 1 at some busy roadside sites in Central London (e.g. Marylebone Road, Euston Road) reductions in NO_x are large. The plots show hourly average concentrations together with the LOESS trend line superimposed. In the period up to and including 11 March (i.e. before the 'pre-lockdown' period) the average NO_x concentrations were 177.7 μg/m^3 and 162.5 μg/m^3 at these two sites respectively, while after the lockdown, from 24 March onwards the average NO_x concentrations were 42.9 µg/m^3 and 62.3 µg/m^3, representing reductions of 76% and 62% respectively. Corresponding reductions in NO_2 at these two sites were 55% and 36% respectively.

One would expect a seasonal reduction in NO_x and NO_2 concentrations as improved dispersion conditions become more frequent in spring and summer, and bearing this in mind, the observed reductions in NO_x at these two sites are broadly consistent with reported traffic reductions in Central London of the order of 60%.

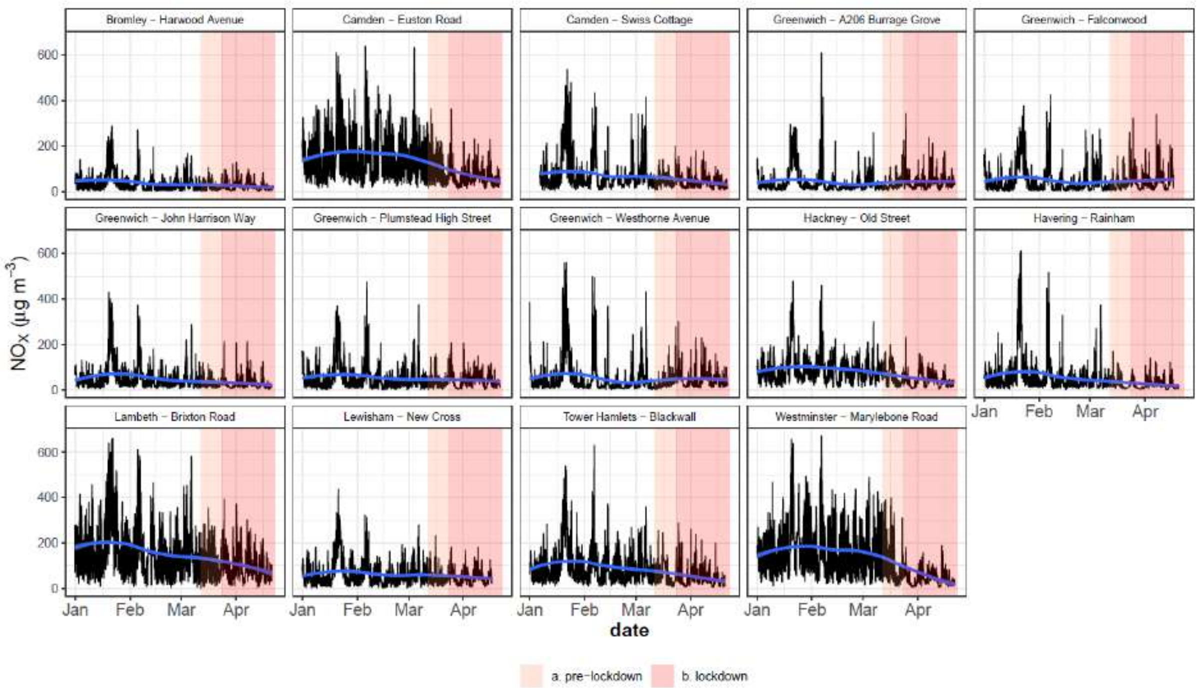
Some sites in outer areas show smaller reductions (the Greenwich sites for example) but overall there appears to have been a widespread reduction in NO_x concentrations across London, albeit to varying degrees. Changes in NO_2 however are even more variable. At most of the sites near busier roads there have been clear reductions in NO_2, at Marylebone Road, Euston Road, Swiss Cottage, Brixton Road, Tower Hamlets and Old Street for example. But at some sites – the Greenwich sites at Westhorne Avenue and the A206 for example – concentrations may even have increased in the few weeks since the lockdown. The extent to which this is due to the predominantly easterly wind flows in the past few weeks will need further investigation.

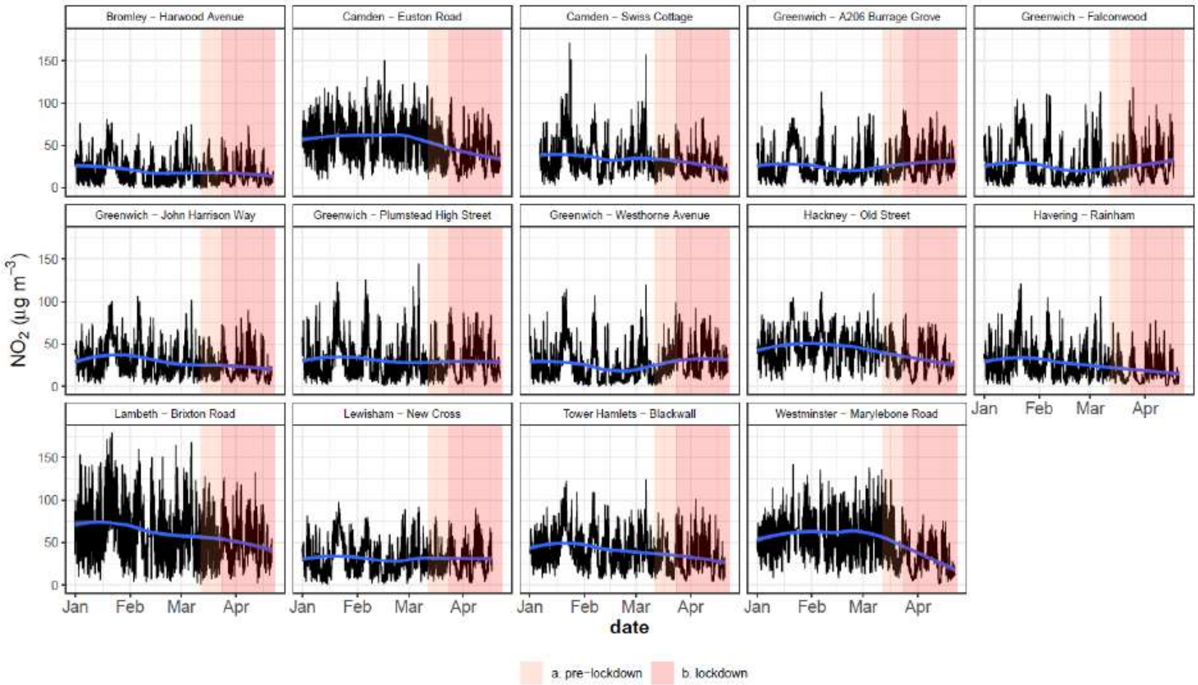
Overall, the mean reduction in hourly NO_X concentrations across the London roads was -44% (ranging from 5% to –75%); for NO_2, the mean decrease was 21.5% (range: 32% to –55%).

The reduction at background locations was smaller compared with that observed at roadside locations. Overall, the background sites observed a decrease of 14% in their NO_2 hourly concentrations, ranging from 30% to –38%.

Figure 1 NO_x and NO_2 concentrations at Roadside and Background sites in London

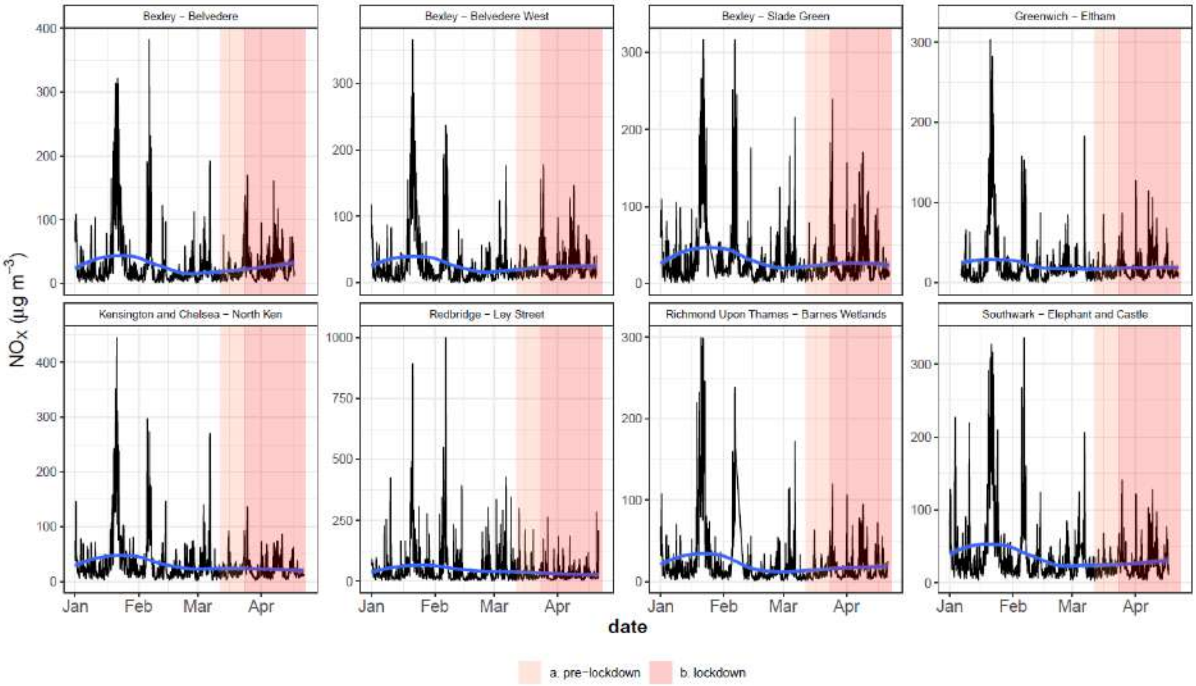
Roadside sites

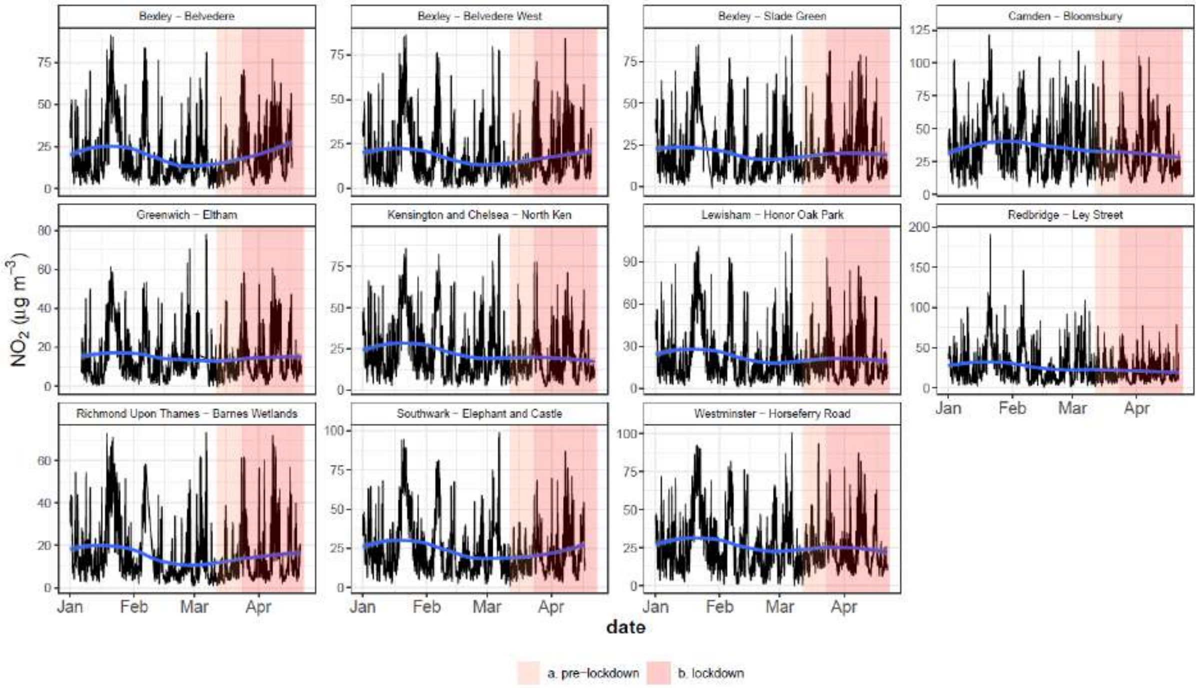




Bromley – Harwood Avenue
Camden – Euston Road
Camden – Swiss Cottage
Greenwich – A206 Burrage Grove
Greenwich – Falconwood
Greenwich – John Harrison Way
Greenwich – Plumstead High Street
Greenwich – Westhorne Avenue
Hackney – Old Street
Havering – Rainham
Lambeth – Brixton Road
Lewisham – New Cross
Tower Hamlets – Blackwall
Westminster – Marylebone Road
NO_2 ($\mu g\ m^{-3}$)
date
Jan
Feb
Mar
Apr
0
50
100
150
a. pre-lockdown
b. lockdown

Background sites



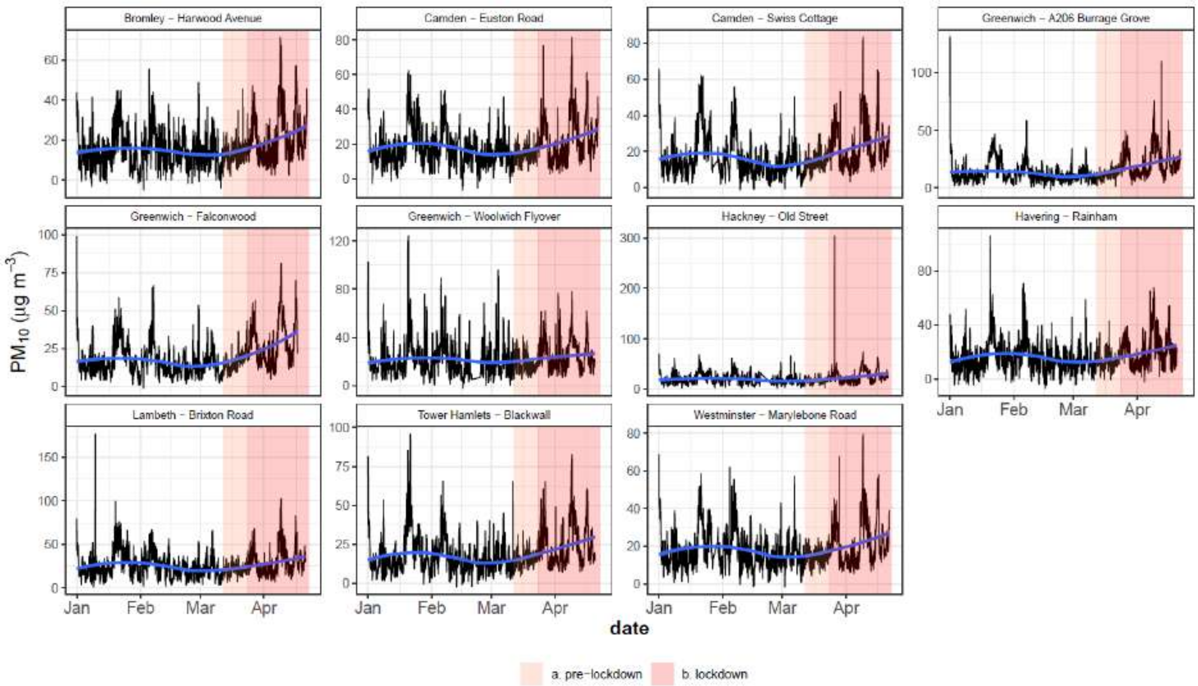

Bexley – Belvedere
Bexley – Belvedere West
Bexley – Slade Green
Camden – Bloomsbury
Greenwich – Eltham
Kensington and Chelsea – North Ken
Lewisham – Honor Oak Park
Redbridge – Ley Street
Richmond Upon Thames – Barnes Wetlands
Southwark – Elephant and Castle
Westminster – Horseferry Road
NO2 (µg m-3)
date
Jan
Feb
Mar
Apr
a. pre-lockdown
b. lockdown

3. Particulate Matter 10 µm (PM_{10})

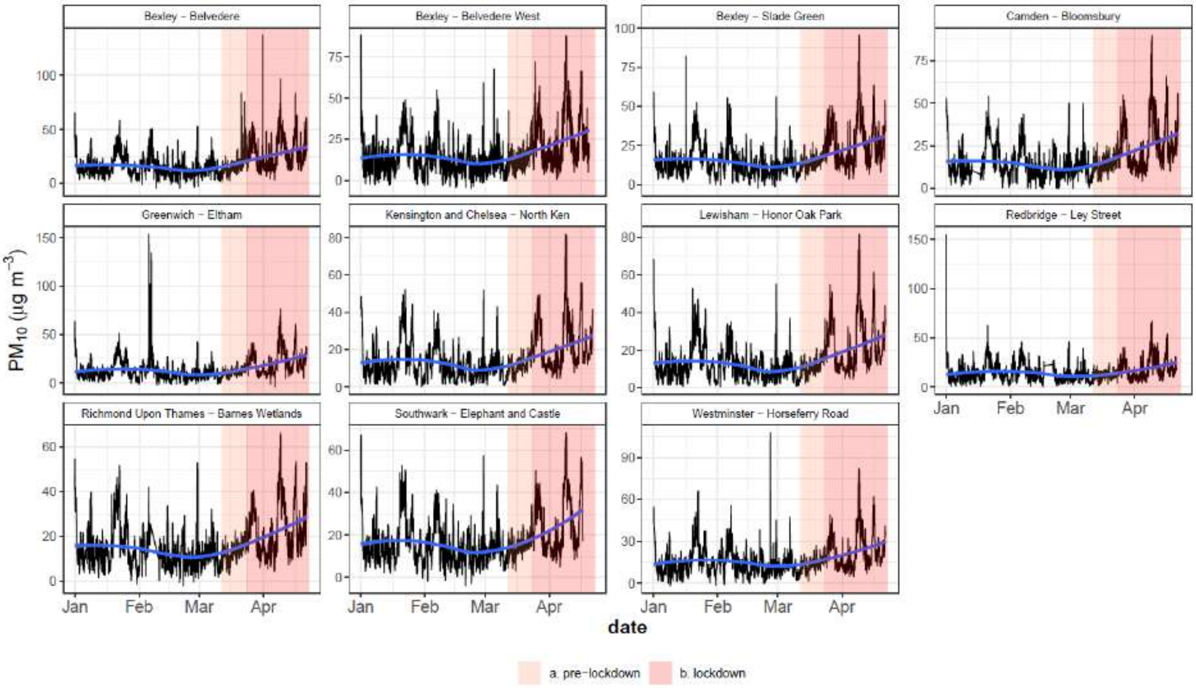
Time series plots of hourly PM_{10} concentrations are shown in Figure 2 below. The effect of easterly winds is immediately apparent with the time series showing the clear effect of a series of pollution episodes near the end of March and in early April, when concentrations higher than any so far seen in 2020 were observed at most sites, at both roadside and background sites. These increased PM_{10} concentrations are a result of the relatively small contribution of road traffic to total mass concentrations of PM_{10}, coupled with the increased contribution from sources outside London and outside the UK. A brief discussion of the more detailed data on the speciation of PM concentrations which King's College London collects is given below.

Figure 2. Hourly PM_{10} concentrations at roadside and background sites in London

Roadside sites



Background sites

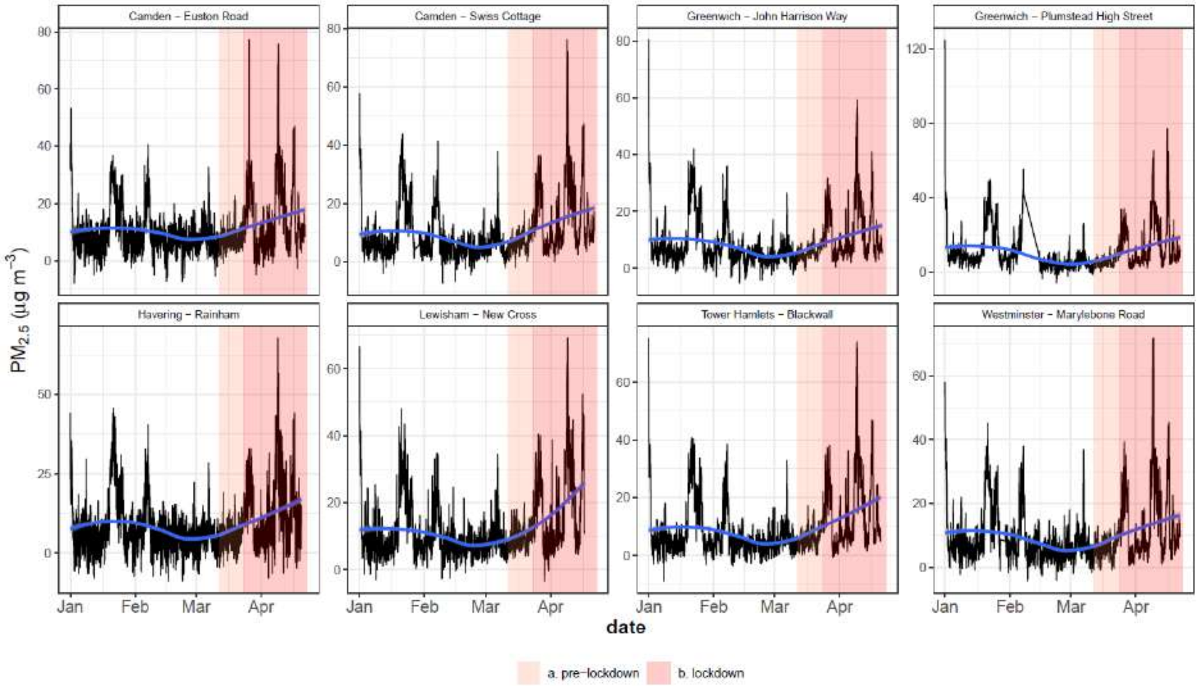

Bexley - Belvedere
Bexley - Belvedere West
Bexley - Slade Green
Camden - Bloomsbury
Greenwich - Eltham
Kensington and Chelsea - North Ken
Lewisham - Honor Oak Park
Redbridge - Ley Street
Richmond Upon Thames - Barnes Wetlands
Southwark - Elephant and Castle
Westminster - Horseferry Road
PM10 (µg m-3)
Jan
Feb
Mar
Apr
date
a. pre-lockdown
b. lockdown

4. Particulate Matter 2.5 µm ($PM_{2.5}$)

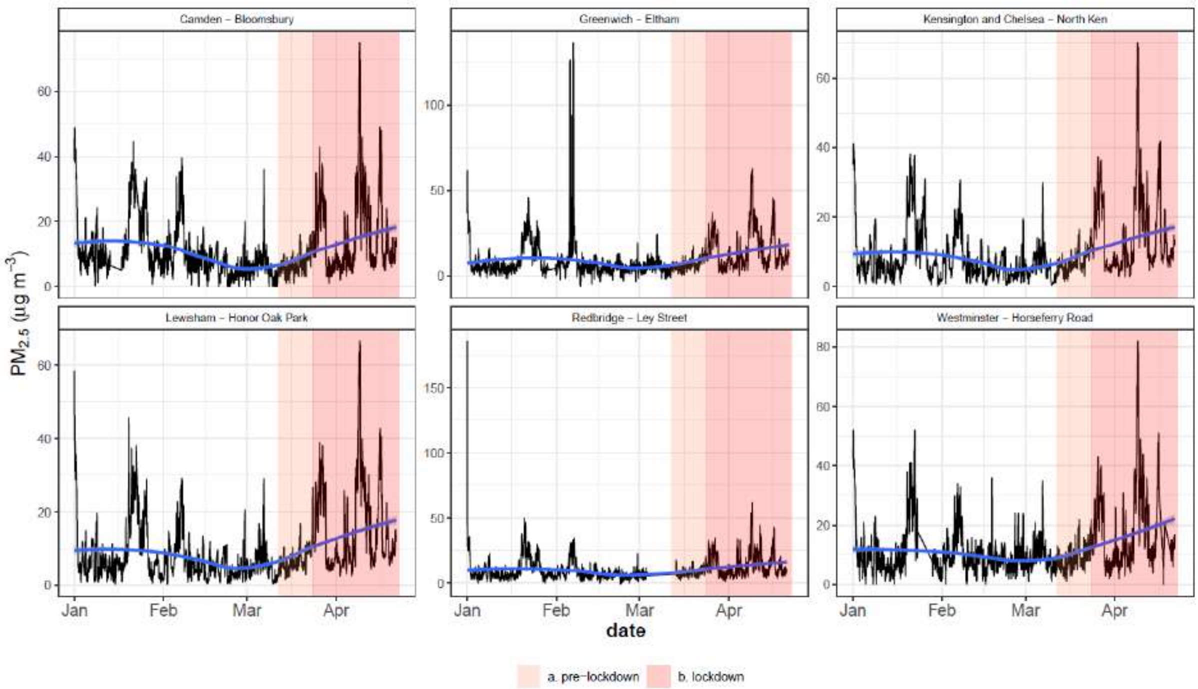
A similar picture also applies for $PM_{2.5}$ where concentrations post-lockdown demonstrated little evidence of an impact from the local reductions in traffic, due to the episodes of easterly winds, with the observed concentrations higher than that at any period during the current year.

Figure 3. Hourly $PM_{2.5}$ concentrations at roadside and background sites in London

Roadside sites

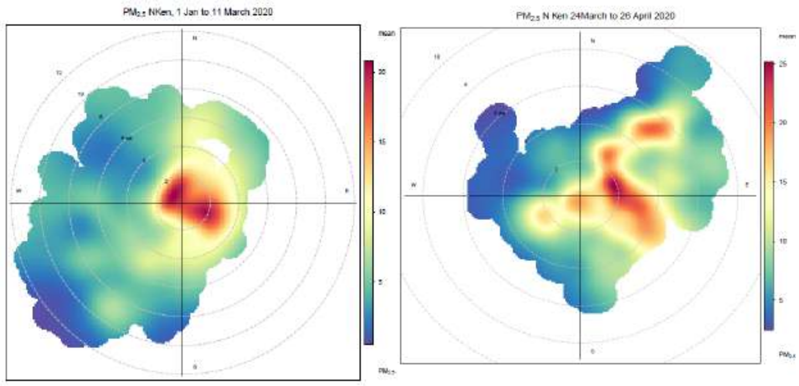


Background sites



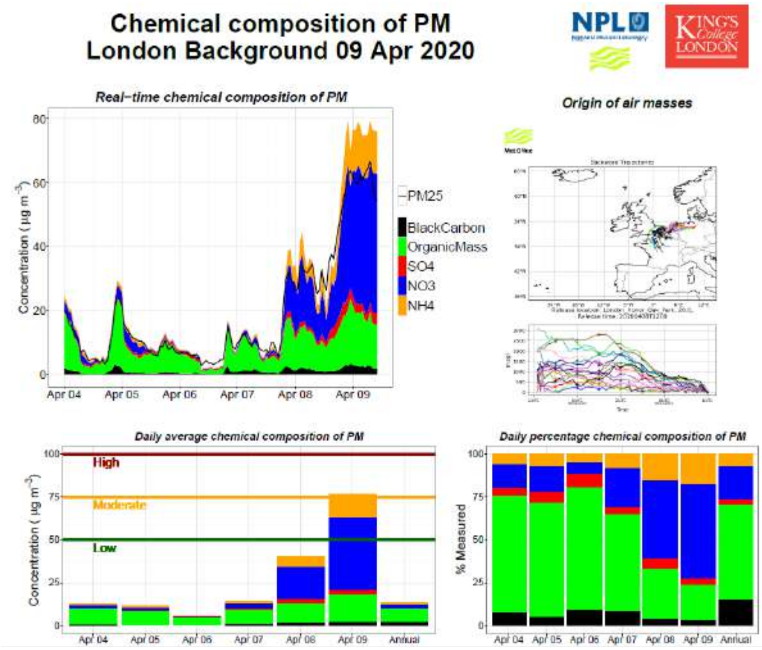
The influence of easterly flows and potential long-range transport can be seen in the two polar plots in Figure 4 below. The plot for the period before the pre-lockdown period shows highest concentrations on relatively low wind speeds with a more uniform directional dependence than the plot for the post lockdown period (from 24 March to 26 April). The low wind speed dependence in the pre-lockdown period is consistent with elevated contributions from local sources. More detailed chemical analysis could shed more light on these contributions. The plot for the post-lockdown period shows a much stronger influence of easterly winds and on higher wind speeds, both consistent with long range transport of particles and their precursors into the UK.

Figure 4. Polar plots of hourly $PM_{2.5}$ at North Kensington, left plot data from 1 Jan to 11 March and right plot from 24 March to 26 April 2020.



Further detail on the composition of $PM_{2.5}$ is afforded by the NERC-funded 'supersite' operated by King's College London at Honor Oak Park. Data from the aerosol mass spectrometer and aethalometer along with backtrack trajectories are shown in Figure 5 for 9 April when concentrations of $PM_{2.5}$ were elevated. The influence of nitrate and organic aerosol on the elevated concentrations observed is clear.

Figure 5. Chemical composition in $PM_{2.5}$ measured at Honor Oak Park with King's pollution forecast and backtrajectory information from the Met office-NAME model.

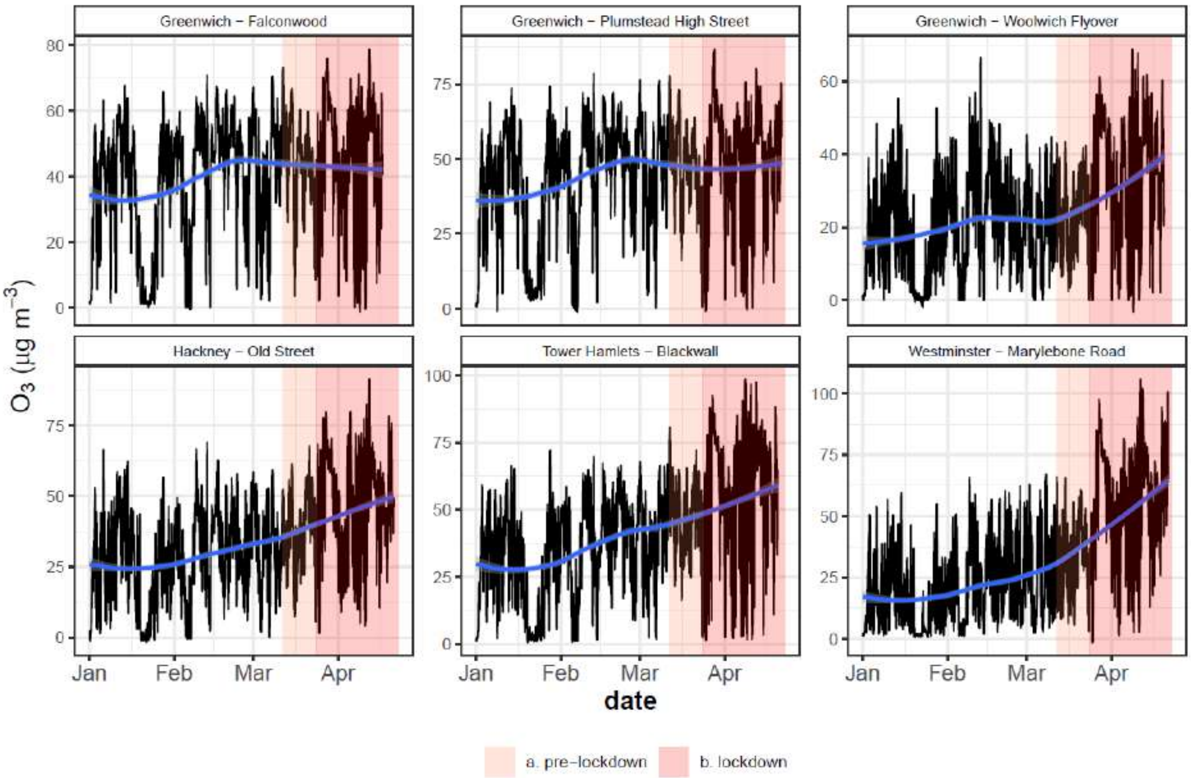


5. Ozone (O_3)

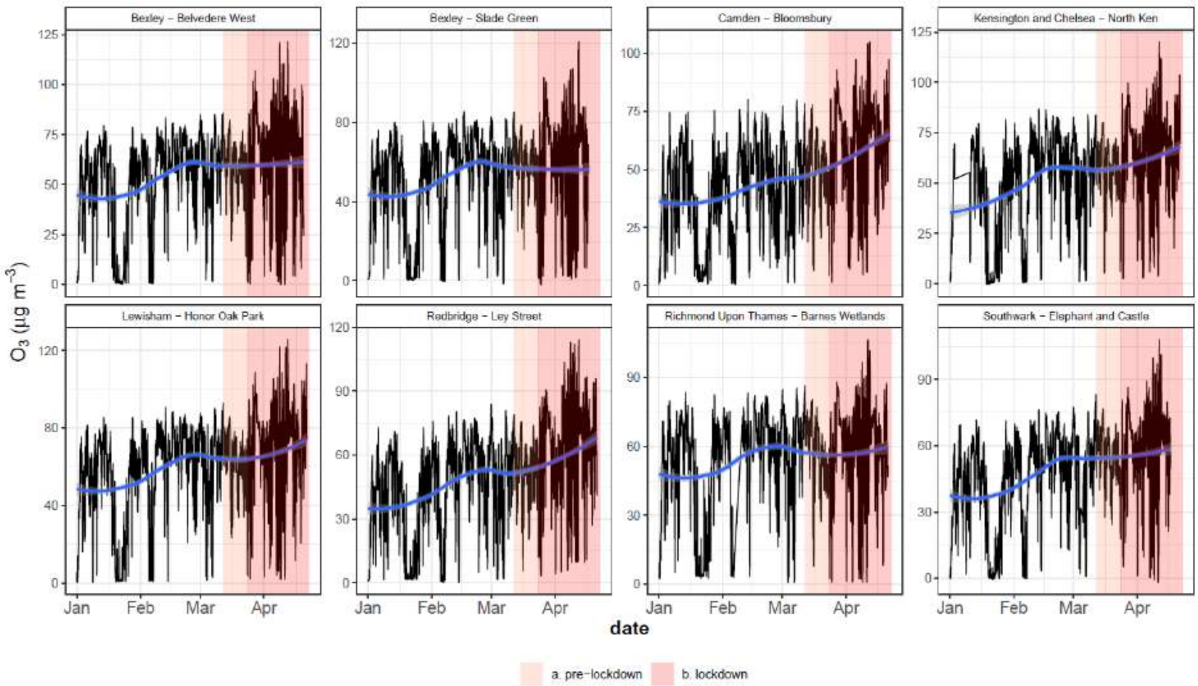
During the post-lockdown period meteorological conditions – elevated temperatures, easterly flows – were conducive to increased photochemical activity, leading to hourly concentrations occasionally exceeding 60 ppb at some urban background sites, with rural concentrations of a similar magnitude, see Figure 6 below.

Figure 6. Hourly ozone concentrations in London

Roadside sites

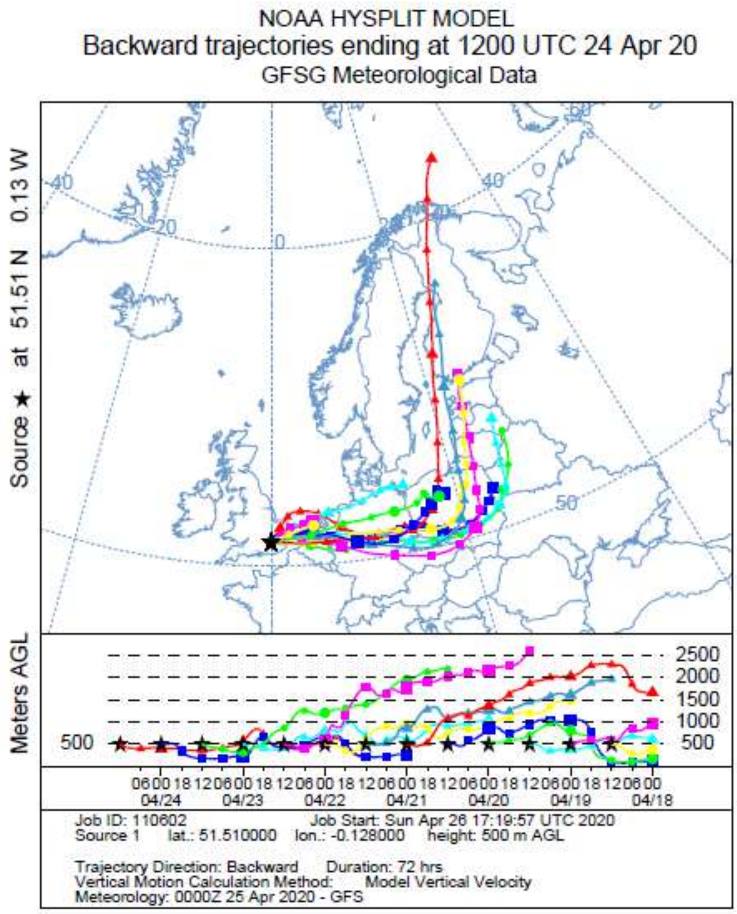


Background sites



As noted for $PM_{2.5}$ and PM_{10}, if the UK is to achieve reductions in ozone concentrations then both the UK and the rest of Europe will need to honour their emission reduction obligations within the EU and the UNECE/CLRTAP, see Figure 7 below. This shows a backtrack trajectory for noon on 24 April when hourly ozone concentrations at North Kensington reached 129 µg/m^3 (65 ppb) at 3 pm.

Figure 7. 72-hour backtrajectories for London, 24 April 2020, from the NOAA Hysplit model (courtesy of the Met Office).



6. Roadside increments in NO_x, NO_2 and $PM_{2.5}$

Quantifying the impact of the measures is not straightforward as London has benefited from significant reductions in traffic emissions due to the accelerated adoption of emission reduction technology through the Ultra Low Emission Zone in 2019 as well as cleaning up London's bus and taxi fleet. Simply comparing 2020 to previous years to account for seasonal and meteorological variability would therefore lead to an overestimation of the impact of the lockdown. Where appropriate, we have attempted to control for meteorology by subtracting the measurements made as a representative London urban background station from the measurements at the roadside station.

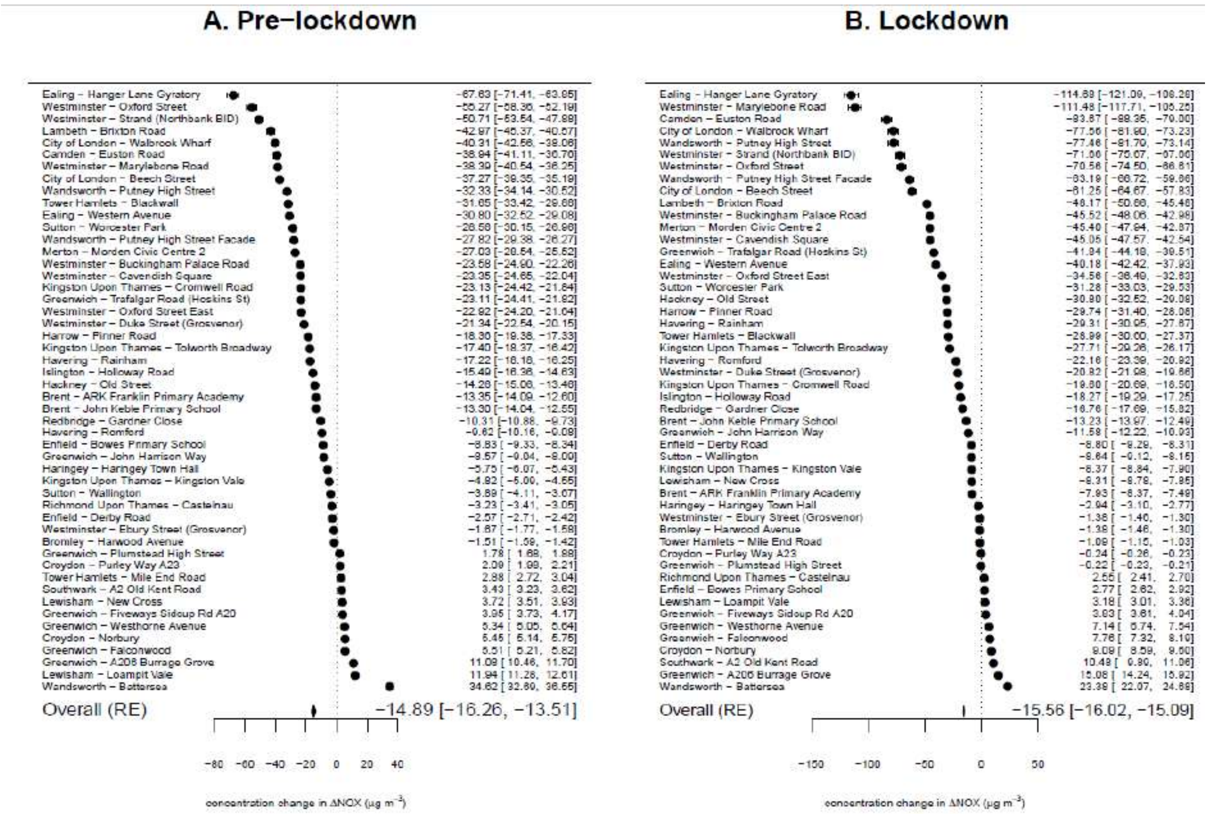
The forest plots in Figure 8 show the change in the roadside increment (concentration measured at the roadside minus the concentration measured at urban background – Kensington and Chelsea North Kensington) measured during the pre-lockdown period (12 to 23 March) and lockdown period (24 March onwards) compared to the mean concentrations measured Jan to March.

2020.Confidence intervals in the plots were calculated using the uncertainty based on the measurement method on the annual concentration. Sites with lower increments might have more weight in the meta-analysis result.

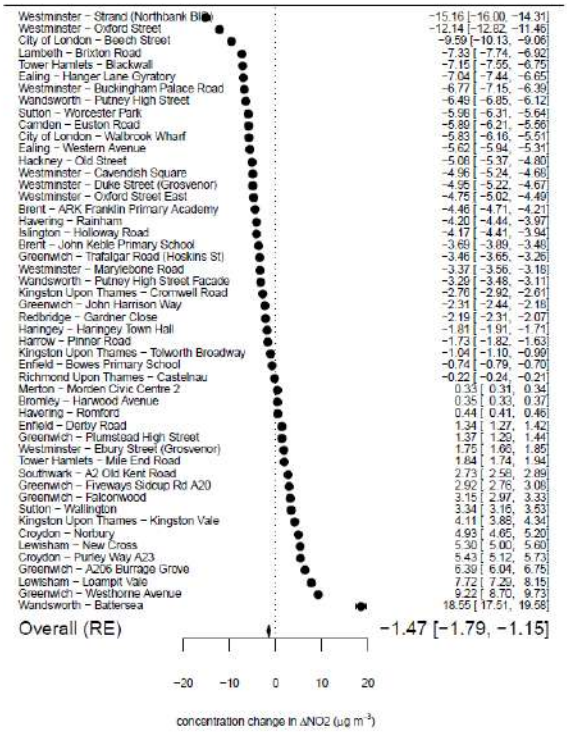
The plots for all three pollutants suggest that there was a reduction in traffic activity in London in the pre-lockdown phase (when measures were 'recommended'), but that this reduced further when the full lockdown measures were strengthened. However not all roadside sites showed decreases in NO_x and NO_2. Two factors could affect this finding; first there may well have been an increase in traffic at some sites, but also the predominance of easterly winds in both periods may have meant that the contribution of the road emissions was less during lockdown because of the relative orientation of the wind direction, the road and the monitoring site location.

The plots focusing on the roadside increment of $PM_{2.5}$ demonstrate the effect of local traffic measures in reducing concentrations, despite the overall increase in $PM_{2.5}$ during the lockdown period, largely due to long range transport.

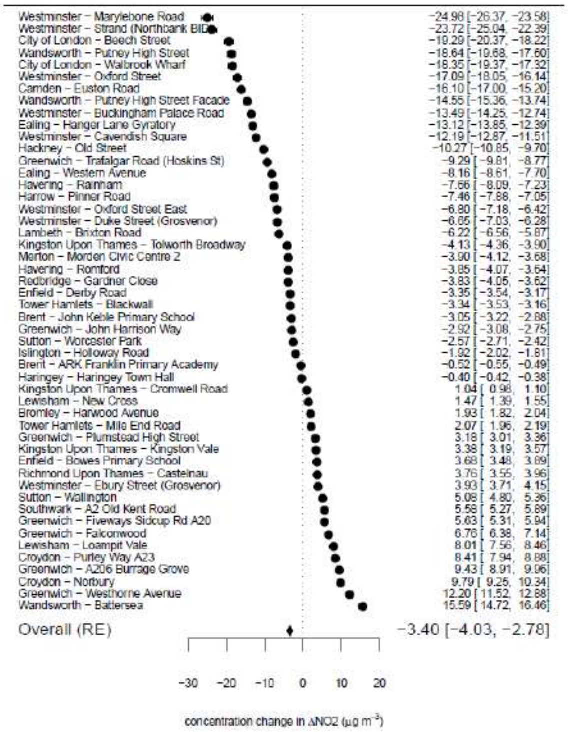
Figure 8. Forest plots of changes in roadside increments (Δ) of NO_x, NO_2 and $PM_{2.5}$ at London roadside sites.



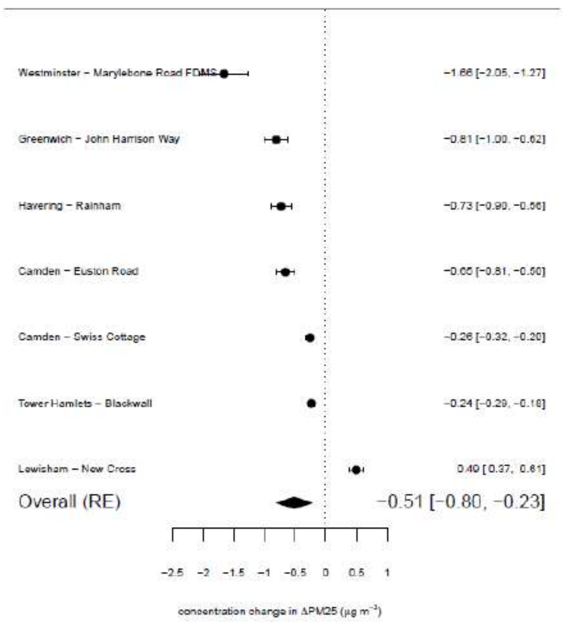
A. Pre−lockdown



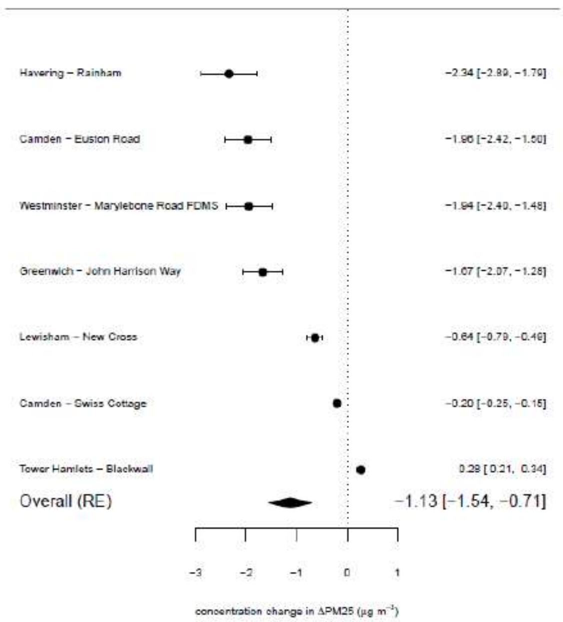
B. Lockdown



A. Pre−lockdown



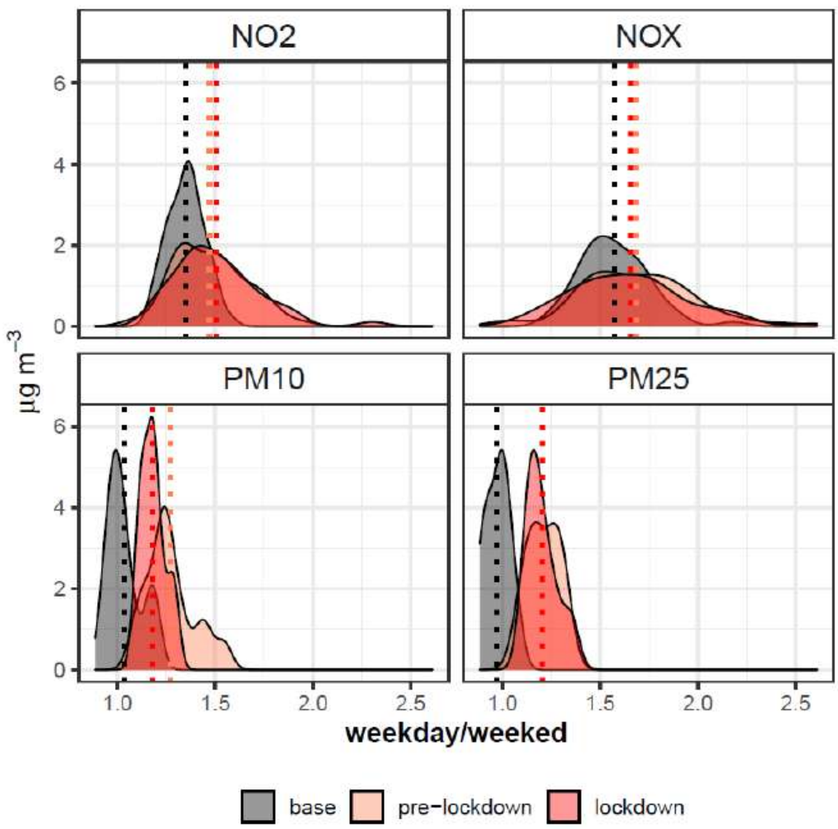
B. Lockdown



7. Weekend/weekday differences

Figure 9 shows the distribution of weekdays/weekend ratios for NO_2, NO_x, PM_{10} and $PM_{2.5}$ for the three periods: base (January to mid-March), pre- and post-lockdown. For NO_2 and NO_x, the ratios >1 indicate higher concentrations over the weekdays compared to weekends. When the lockdown measures were implemented, there is a shift to higher ratios. That might indicate that traffic emissions during weekends (and bank holidays) have probably decreased; and some work-related traffic is still taking place Monday to Friday. For PM_{10} and $PM_{2.5}$ a clear shift in the ratio means were observed, from 1 during the base period, to 1.2-1.3 during the lockdown. This change in the ratio is probably due to the PM episodes taking place predominantly during weekdays.

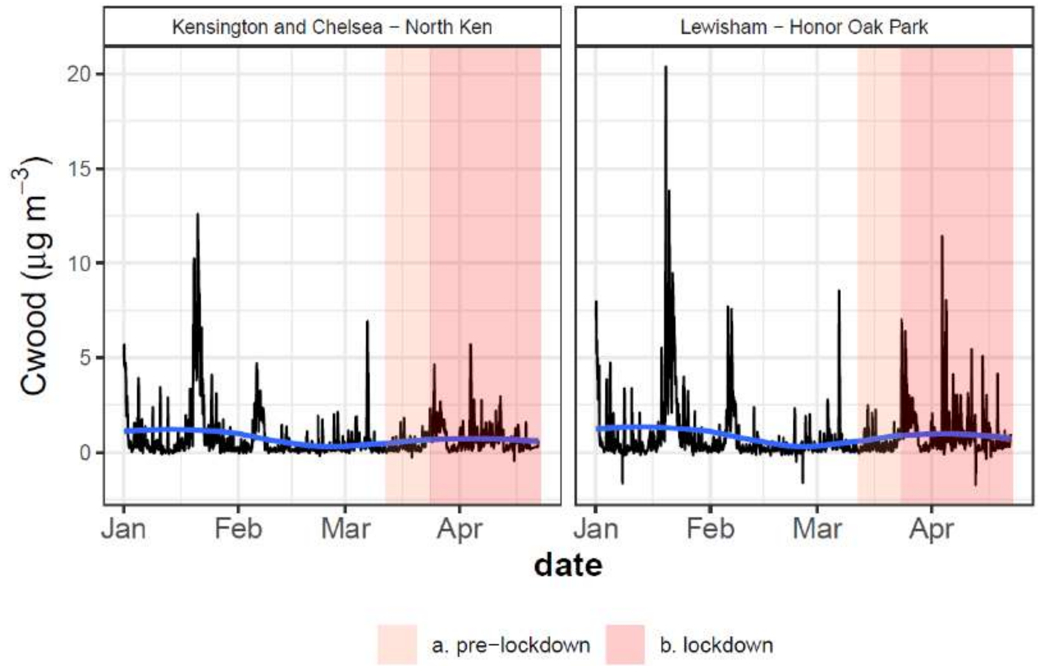
Figure 9. Density plots showing the ratio between weekday/weekend concentrations measured across the London roadside sites. Vertical lines denote the mean ratio for each period.

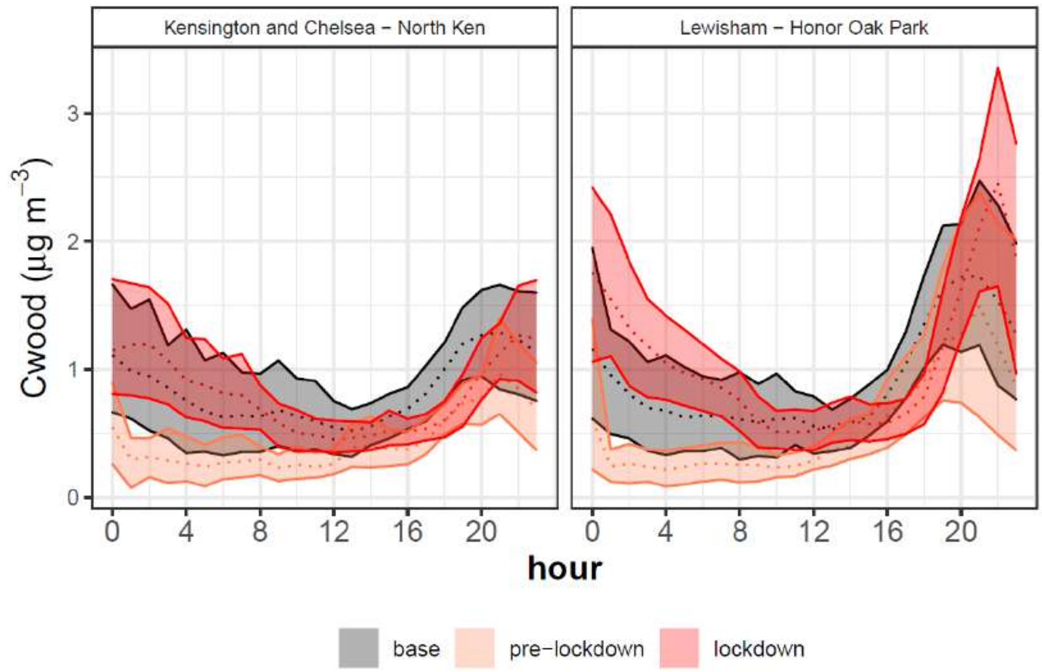


8. Wood burning contributions to PM concentrations

PM from wood burning (Cwood) has been calculated from aethalometer data using the Sandradewi et al. (2008) method, consistent with Font and Fuller et al (2017). Hourly time series of the Cwood concentration at the London background sites and the mean hourly diurnal variation for each period are shown. A series of peaks up to 5 and 10 µg m^{-3} were observed at the start of the lockdown period at North Kensington and Honor Oak Park, respectively. Greater concentrations were measured during the winter months; however, we expect a decrease in springtime. Diurnal plots show similar mean concentrations during lockdown to those measured in the 'base', but with a later evening peak, perhaps due to later use of home heating due to longer daylight hours.

Figure 10. Hourly time series in PM from wood burning (Cwood) and mean hourly variation per each period.





9. Implications for health effects

In normal circumstances the increases in $PM_{2.5}$ levels in the post lockdown period would suggest an increase in life years lost and increased morbidity. However, given the special circumstances of the

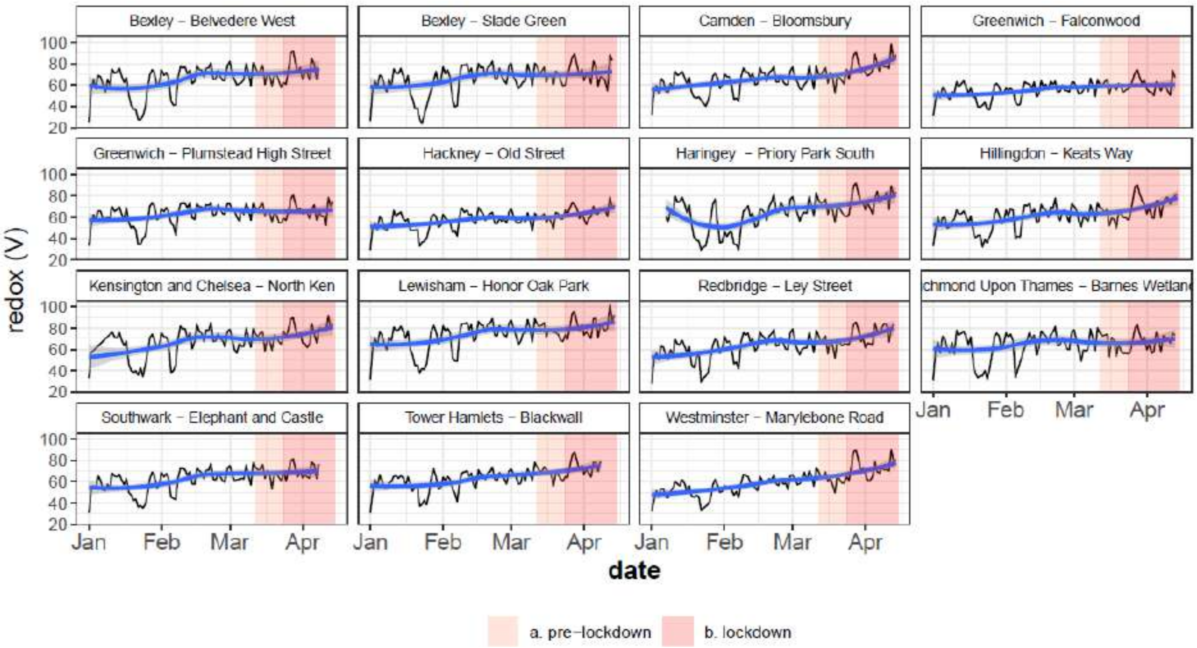
Covid-19 outbreak, the application of standard concentration-response functions and even baseline rates of mortality and disease in health impact assessment would be questionable. This is because health impact assessment relies on the assumption that the population and location characteristics applying to the epidemiological studies from which the concentration-response functions were derived also apply to the population and location to which they are being applied. For example, standard concentration-response functions apply to all-cause mortality but represent the typical relative proportions of respiratory and cardiovascular deaths. Respiratory deaths will be much higher than usual during the COVID-19 outbreak. Concentration-response functions for cause-specific mortality could be used but they would still come from studies of populations that were not experiencing an epidemic. For baseline rates, the exact question would need to be defined. If using the air pollution changes as a model for future air pollution reductions many years ahead, it could be argued that typical baseline rates (perhaps with adjustment for expected mortality trends irrespective of COVID-19) would be most appropriate. If modelling the expected actual changes in air-pollution associated health outcomes, then real baseline rates in this period might be used (although they may not yet be available). However, it would still be in the absence of knowledge about how air pollution affects those with COVID-19, something that is difficult to study to a high standard in a short timescale. With this providing a significant proportion of baseline mortality at the current time, this is a significant omission. The Environmental Research Group would be able to investigate these health impact assessment issues further if required but many significant uncertainties would remain.

We have seen that NO_2 concentrations have decreased and in normal circumstances this could also lead to a decrease in health effects, to some extent counteracting the increases in PM concentrations. The exact extent of this counteraction is complicated by the issues discussed above and by the fact that different pollutants have greater or lesser effects on different health outcomes. There is stronger evidence for effects of PM on cardiovascular mortality and changes in long-term exposure to PM often dominate cost-benefit analyses. This might still apply but the pollutants that have stronger links to respiratory rather than cardiovascular effects (NO_2 and O_3) might provide a higher proportion of estimated health effects than usual due to higher baseline rates for respiratory outcomes.

Ozone concentrations were higher in the post-lockdown period due to the episodes mentioned above. Emissions of ozone precursors, NOx and VOCs from the rest of northern Europe will have combined with emissions from the UK to produce the higher ozone concentrations. Short-term exposures to elevated levels of ozone are a cause of adverse health effects.

Even in the absence of the photochemical episodes, the reduction in NO_x emissions in London will have led to an increase in ozone concentrations purely as a result of the simple $NO/NO_2/O_3$ chemistry. These reactions produce no net ozone and the combined pollutant $O_x = O_3 + NO_2$ is conserved. Both ozone and NO_2 contribute to oxidative stress in the lung, ozone more than NO_2 on a unitary basis, reflecting their redox potentials. Both have been shown to be associated with short and long-term health effects in their own right, but the sum of O_x has also been related to mortality in London (Williams et al, 2012) and is therefore worthy of consideration here given the difference in the relative change in the concentrations of O3 and NO2 over the lockdown period. Figure 11 shows a plot of the time series of O_x at selected London sites. These plots show a clear increase in O_x over the lockdown period, but even without the episodes there is an increase in the gaseous oxidative potential of O_x in London. Reductions in O_x will require reductions in NO_2 but also reductions in the tropospheric background of ozone, necessitating reductions of pollutants, chiefly methane, at a global scale.

Figure 11. Time series of O_X (= O_3 + NO_2) weighted by redox potentials of O_3 and NO_2.



10. Changes in personal exposure

ERG's London Hybrid Exposure Model (LHEM) (Smith et al., 2016) has been used to estimate the impacts of the lockdown on personal exposure to $PM_{2.5}$ and NO_2 for several different population subgroups. The LHEM predicts the exposure of people indoors and outdoors, including whilst travelling (train, tube, bus, car, cycling and walking) and includes indoor sources such as cooking. The subgroups included children, professional drivers, hospital workers and tube users. Annual average exposure is estimated, assuming a typical working week and weekend.

For the purpose of this study, outdoor concentrations in the post-lockdown period have been modelled assuming a reduction in road transport (-53%) and aviation-related activity (-73%), with corresponding decreases in emissions relative to a base "before lockdown" case of 2019.

Before lockdown, the differences in exposure to $PM_{2.5}$ reflect the importance of the time spent indoors and the impact of transport-related exposure, with tube users and professional drivers having the highest exposure. This is more pronounced for exposure to NO_2, where the drivers' exposure is considerably higher than for the other subgroups.

After lockdown, owing to changes in both traffic sources of air pollution and people's work activity, average exposures were reduced by 5-24% ($PM_{2.5}$) and 18-27% (NO_2) for the different population subgroups (see Table 1 and Figures 12 and 13 and the Tables in Annex B). Despite these benefits, the drivers' average NO_2 exposure remained substantially higher than other groups.

The sensitivity of changes to indoor activity was tested by increasing time spent at home and, importantly, additional exposure in a kitchen environment where cooking is taking place. The addition of an hour of extra cooking time demonstrates this to be an important source, increasing average $PM_{2.5}$ exposure to above pre-lockdown levels except in the case of the tube user. NO_2 exposure was also increased, although it remained below pre-lockdown levels for all population subgroups.

A more comprehensive analysis for the specific lockdown period, combining changes to European, UK and London emissions as well as a wide range of population subgroups could be undertaken as part of a larger study. This could include different age groups, socioeconomic groups and ethnicities, as well as addressing the LHEM model's uncertainties. More detail on the model assumptions and uncertainties in this study is given in Annex B.

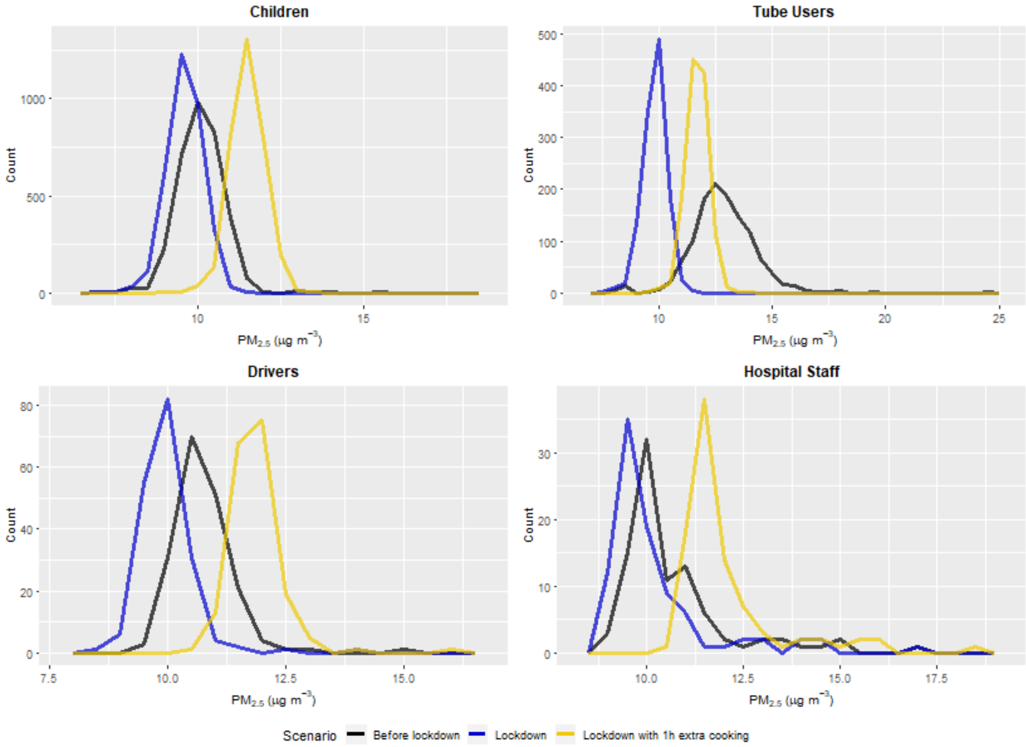


Figure 12. Histograms comparing modelled human exposure to $PM_{2.5}$ before and after the lockdown in different subgroups, with the “1h extra cooking” scenario also included.

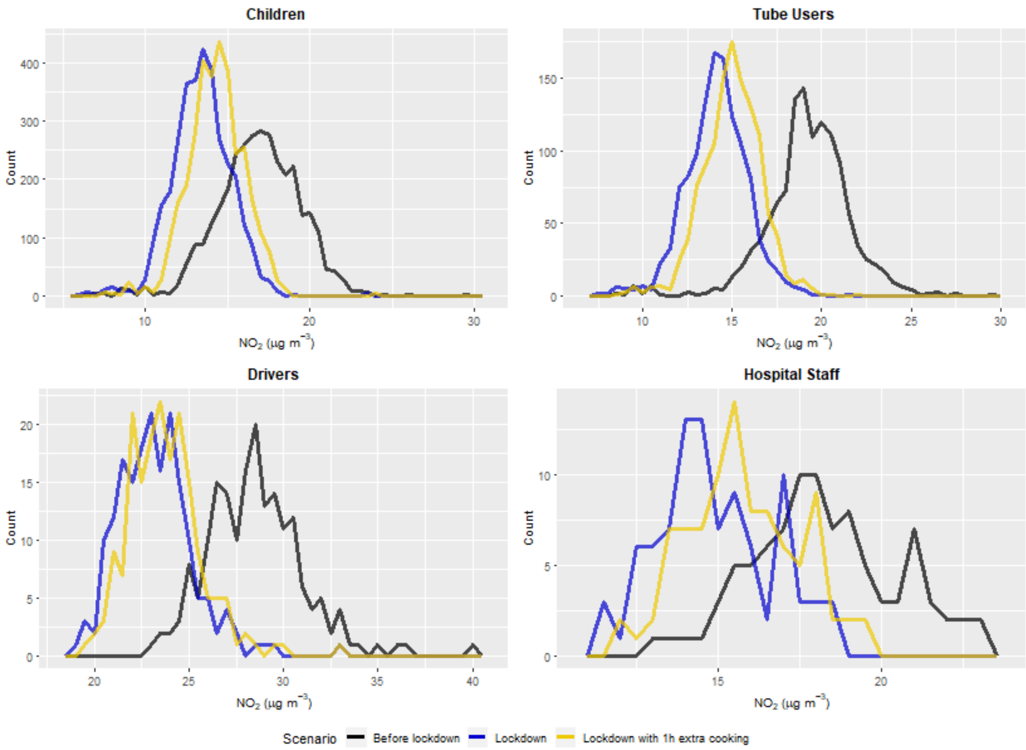


Figure 13. Histograms comparing modelled human exposure to NO_2 before and after the lockdown in different subgroups, with the “1h extra cooking” scenario also included.

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Font, A., Butterfield, D., Fuller, G. 2017. Air pollution from wood-burning in UK cities. Kings College London. Available at: https://uk-air.defra.gov.uk/assets/documents/reports/cat05/1801301017_KCL_WoodBurningReport_2017_FINAL.pdf

Williams, M L, Atkinson R W, Anderson H R and Kelly F J, Associations between daily mortality in London and combined oxidant capacity, ozone and nitrogen dioxide, Air Quality, Atmosphere and Health, DOI 10.1007/s11869-014-0249-8, 2014.

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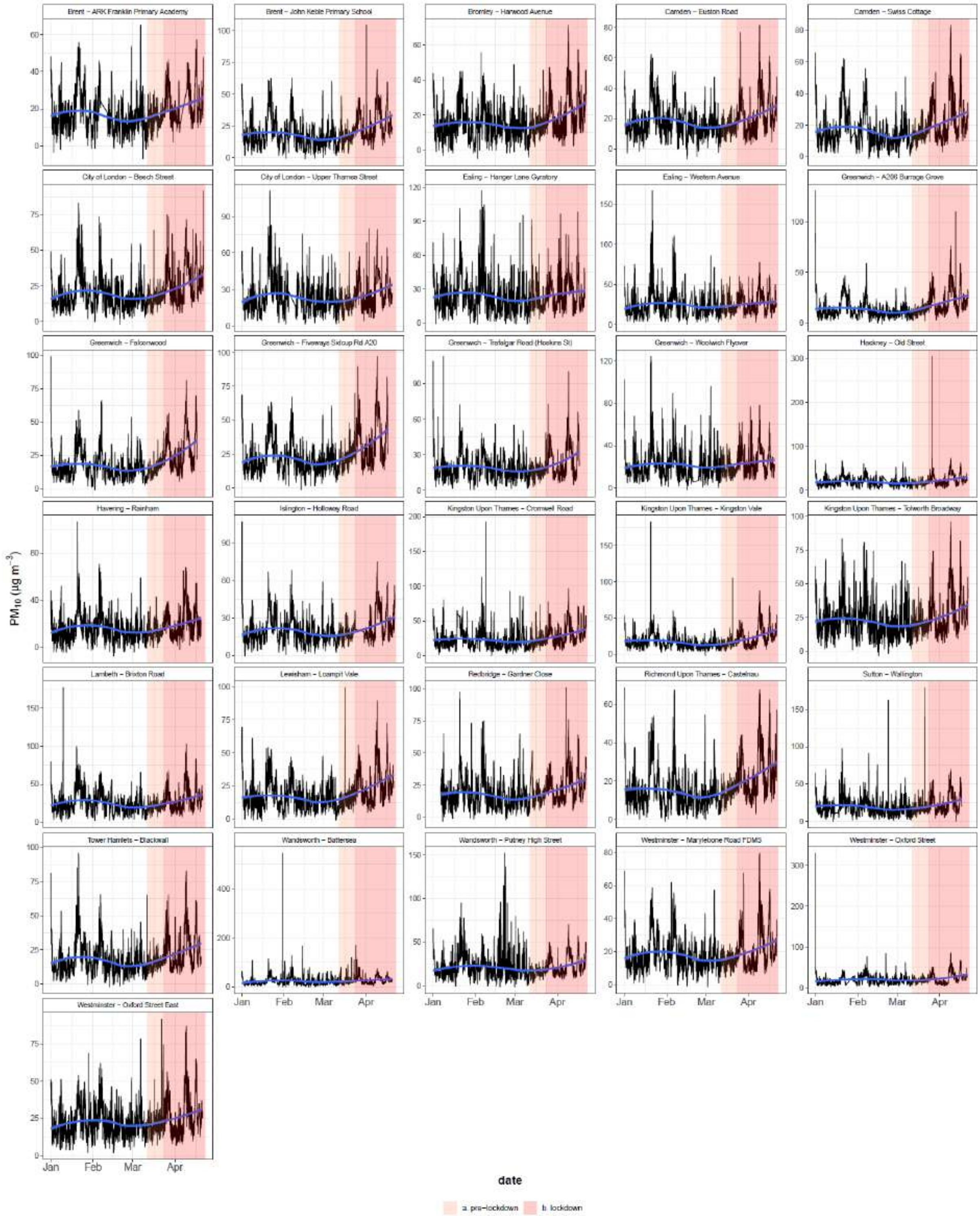
ANNEX A

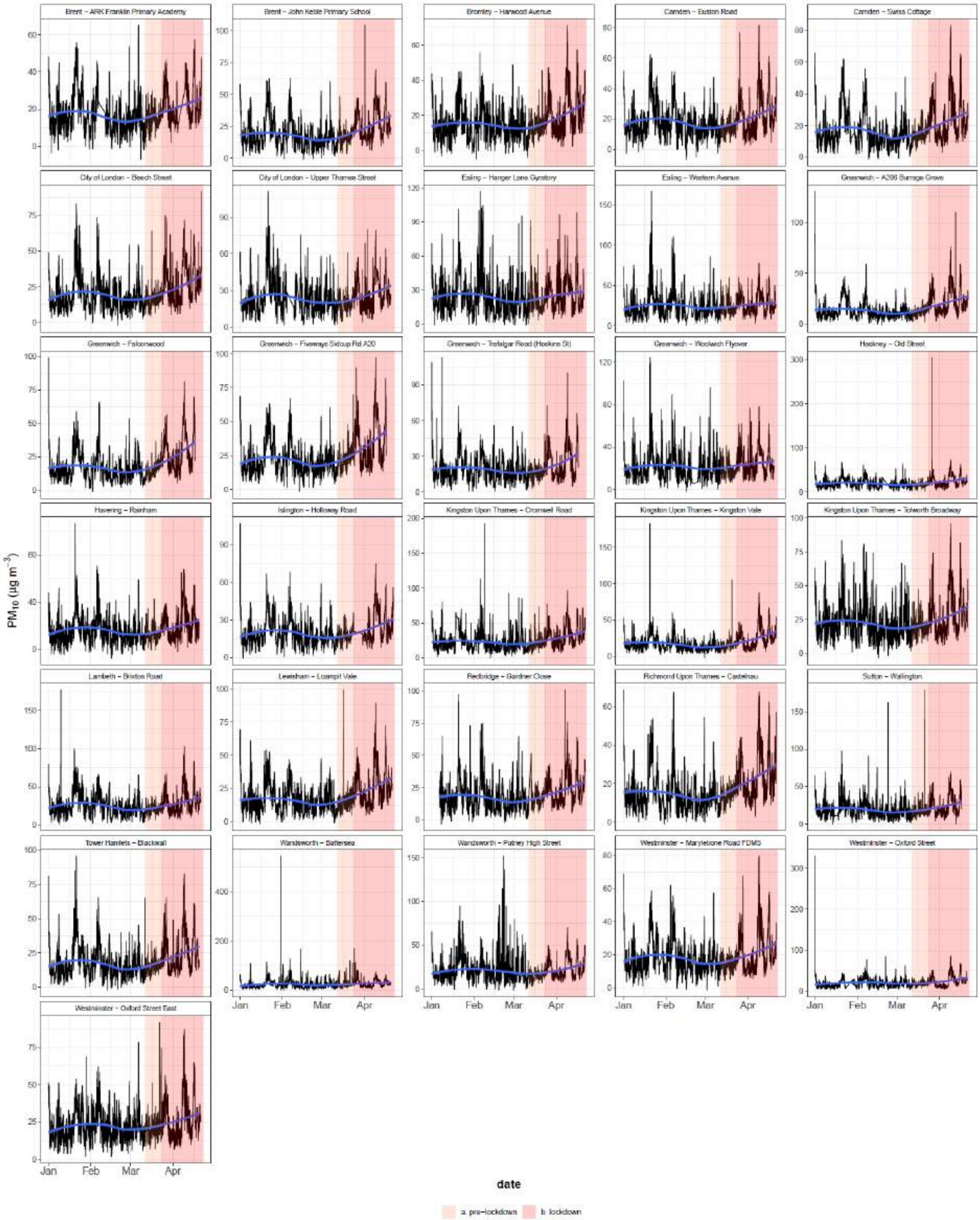
Hourly time series for **all roadside** locations across the **London Air Quality Network**.

Nitrogen oxides (NO_X = NO + NO_2)

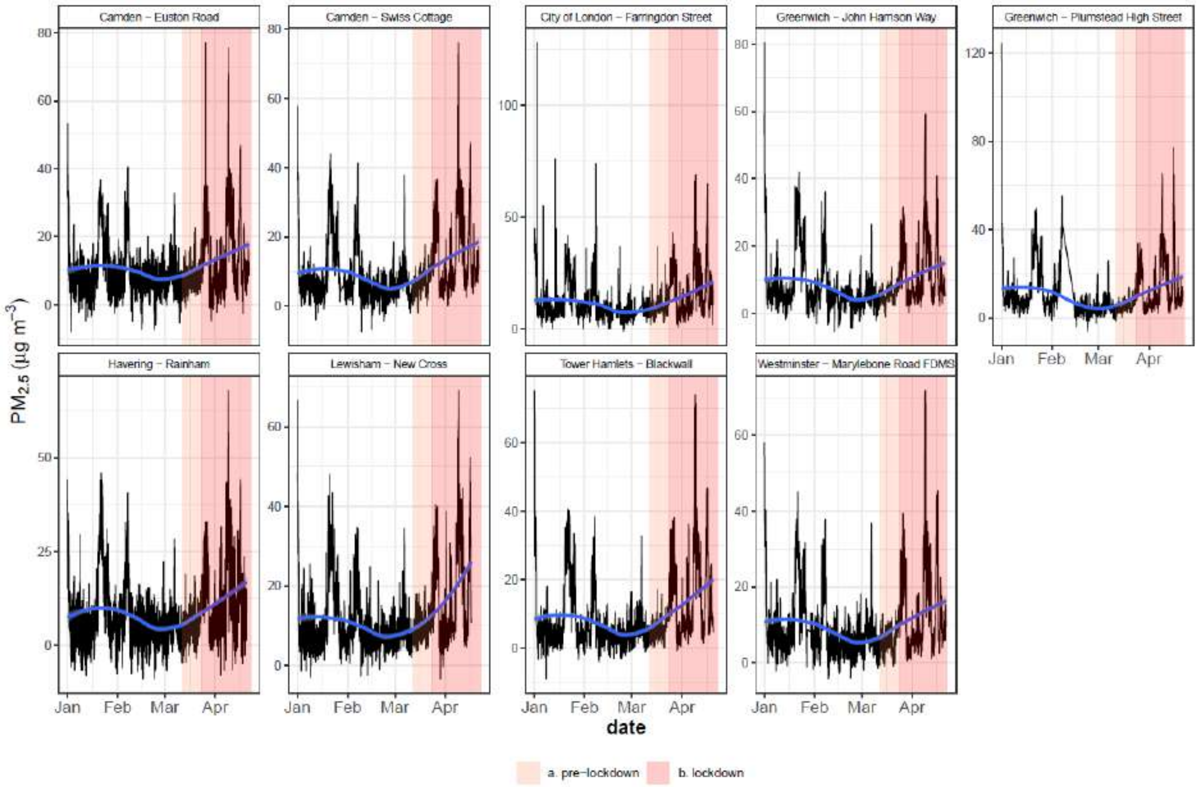
Nitrogen dioxide (NO_2)



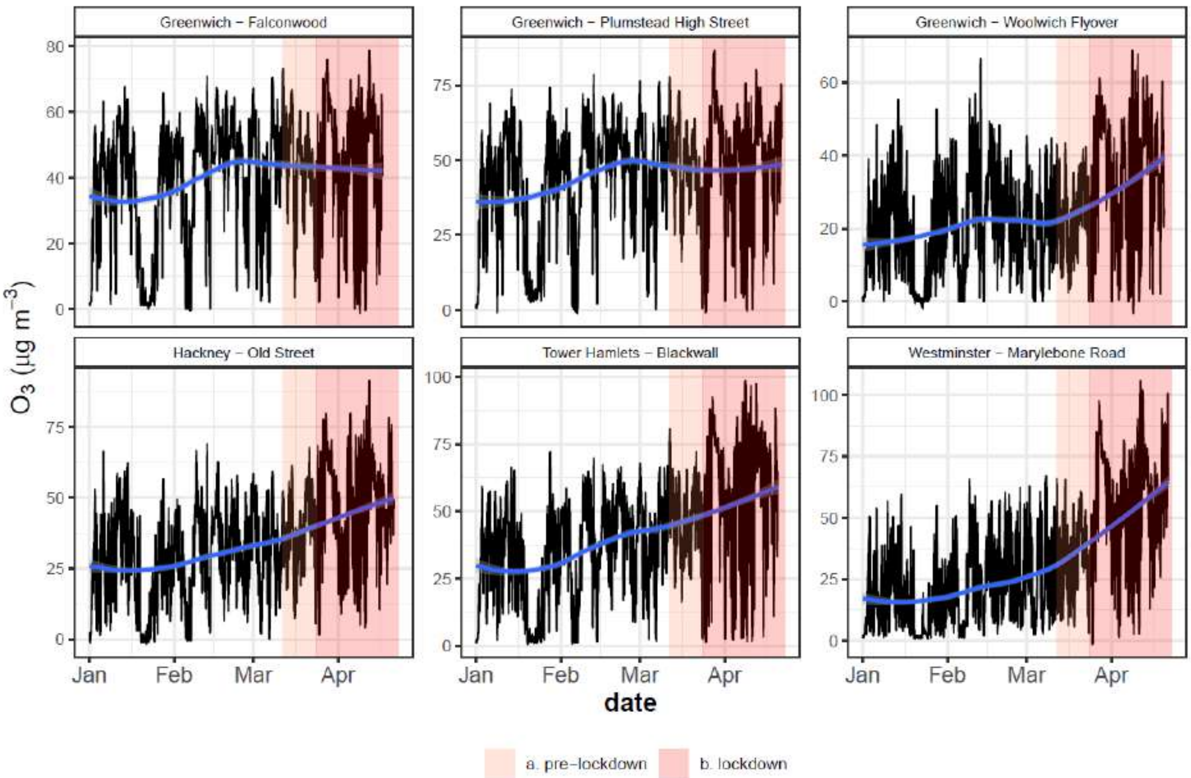
Particulate Matter 10 µm (PM_{10})



Particulate Matter 2.5 µm ($PM_{2.5}$)

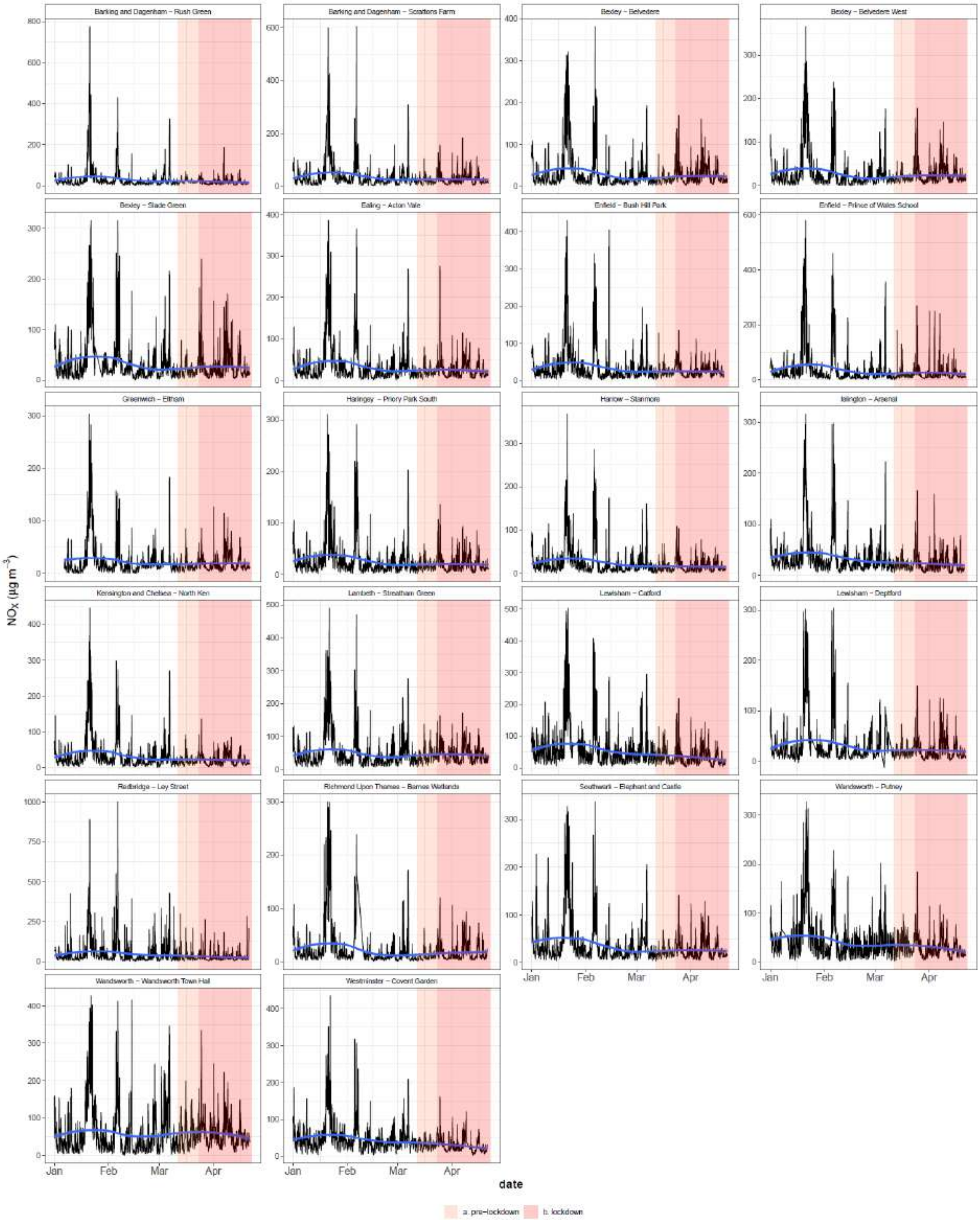


Ozone (O_3)

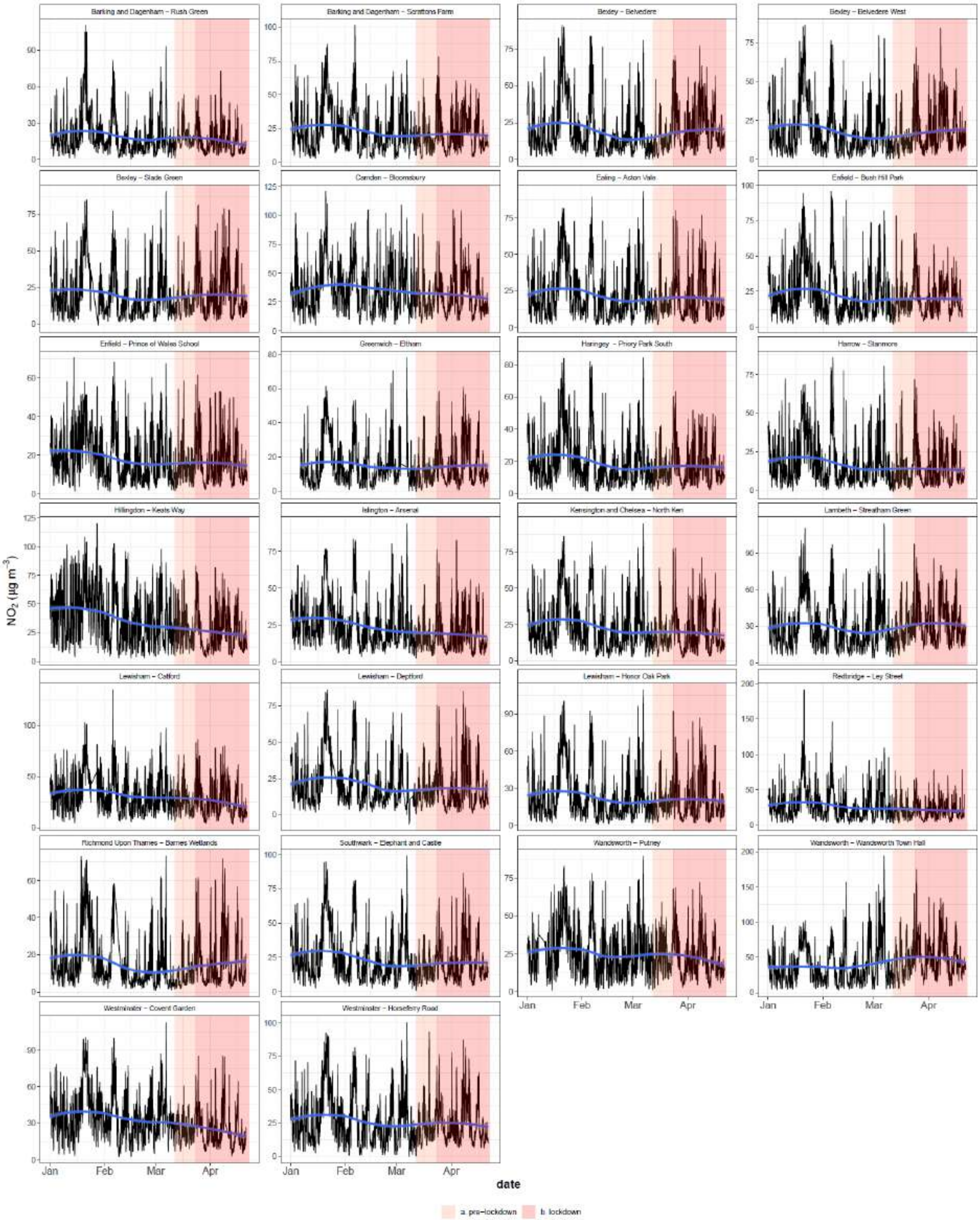


Hourly time series for **all background and suburban** locations across the **London Air Quality Network**.

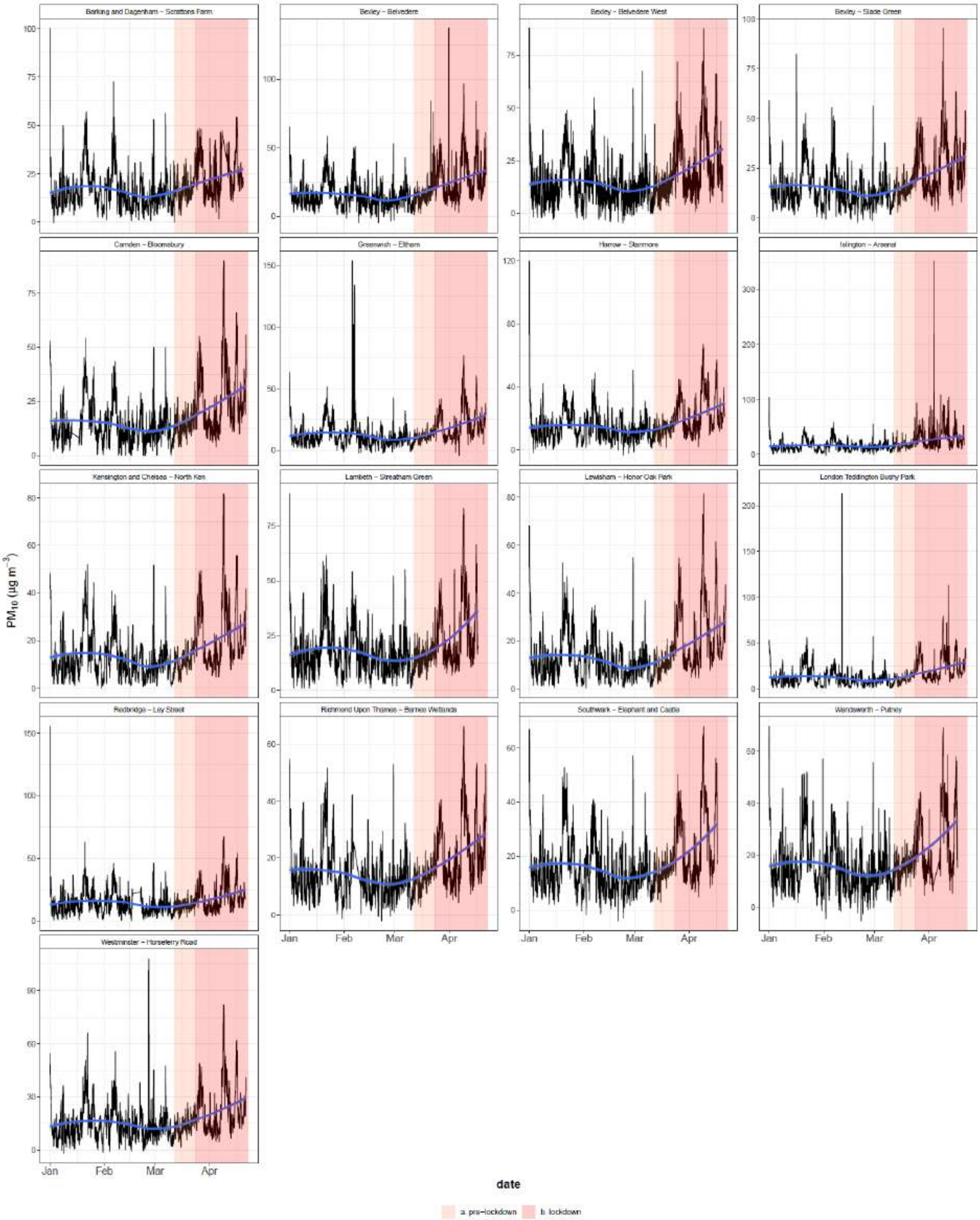
Nitrogen oxides ($NO_X = NO + NO_2$)



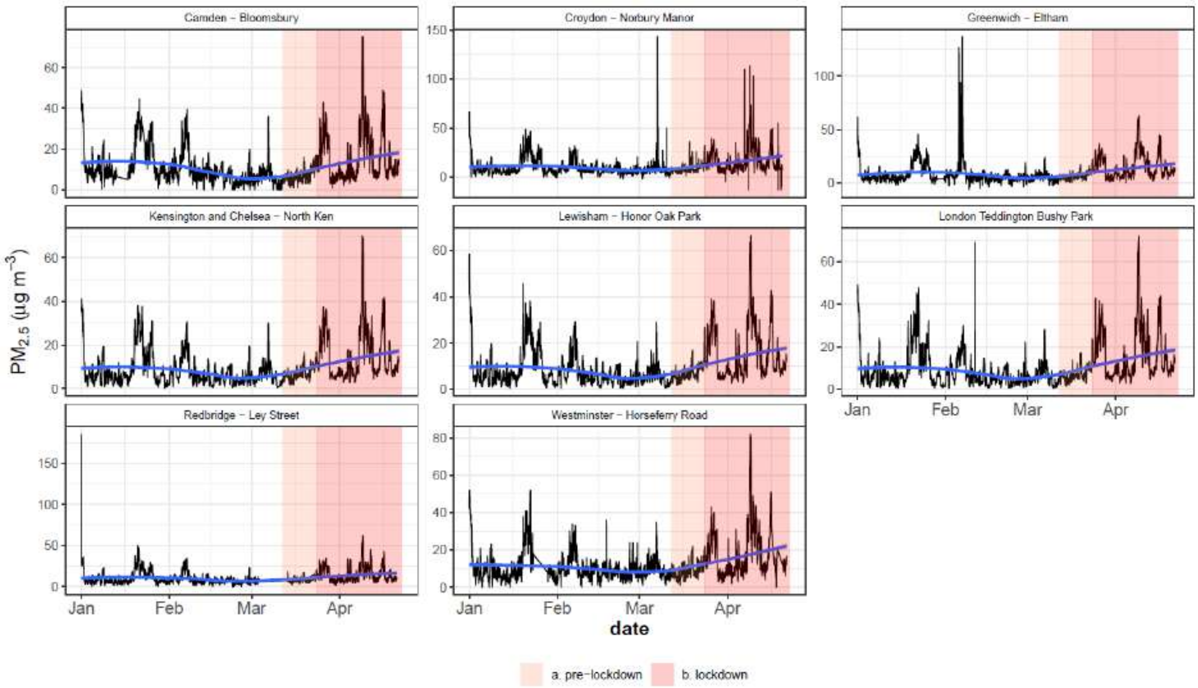
Nitrogen dioxide (NO_2)



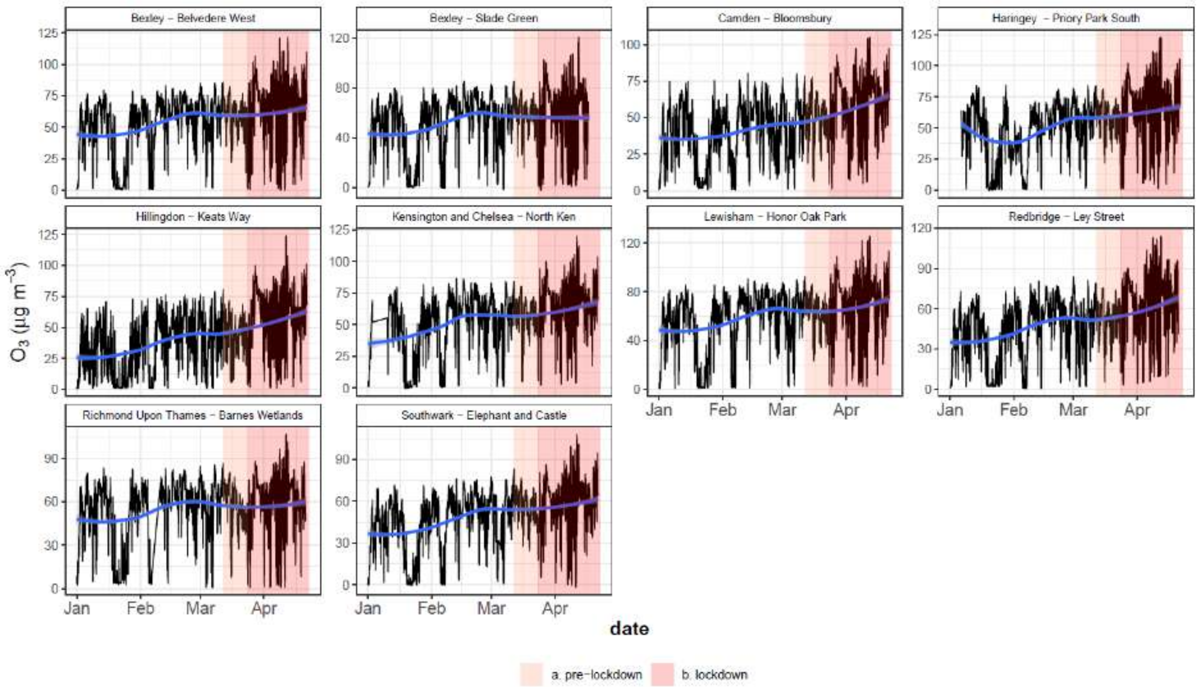
Particulate Matter 10 µm (PM_{10})



Particulate Matter 2.5 µm ($PM_{2.5}$)



Ozone (O_3)



ANNEX B

Key Messages (see Table 1):

- We have estimated the exposure to $PM_{2.5}$ and NO_2 of several population subgroups in London, indoors and outdoors, whilst travelling to work and working. The subgroups included children, professional drivers, hospital workers and tube users. It was assumed that hospital workers and professional drivers continued to work as normal, while tube users and children stayed at home. The exposure represents an annual average, assuming a typical working week and weekend.
- Before lockdown, the differences in exposure to $PM_{2.5}$ reflect the importance of the time spent indoors and the impact of transport-related exposure, with tube users and professional drivers having the highest exposure. This is more pronounced for exposure to NO_2, where the drivers' exposure is considerably higher than for the other subgroups.
- After lockdown, and due to changes in traffic sources of air pollution and people's work activity, average subgroup exposures were reduced by 5-24% ($PM_{2.5}$) and 18-27% (NO_2). Despite these benefits the driver remained the most exposed.
- The sensitivity of changes to indoor activity was tested by increasing time spent at home and, importantly, additional exposure in a kitchen environment where cooking is taking place. The addition of an hour of extra cooking time demonstrates this to be an important source, increasing $PM_{2.5}$ exposure to above pre lockdown levels except in the case of the tube user. In kitchen exposure also increases NO_2 although it remains below pre lockdown levels for all population subgroups.
- The addition of 2 hours cooking is further evidence of the kitchen as an exposure environment and is meant as a possible maximum exposure or reflective of modern kitchen diner design.
- A more comprehensive analysis for the specific lockdown period, combining changes to European, UK and London emissions as well as a wide range of population subgroups could be undertaken as part of a larger study. This could include different age groups, socioeconomic groups and ethnicities, as well as addressing the LHEM model's uncertainties.

Table 1. Mean (min-max) exposure to $PM_{2.5}$ and NO_2 in different scenarios.

Scenarios	Children (n = 3319)	Tube users (n = 1218)	Drivers (n = 183)	Hospital staff (n = 92)
	$PM_{2.5}$ ($\mu g\ m^{-3}$)			
Before lockdown	10.1 (7.1-18.1)	12.9 (7.8-24.6)	10.8 (9.3-14.9)	10.5 (9.1-14.9)
After lockdown	9.6 (9.3-9.9)	9.8 (7.6-12.4)	10.0 (8.6-14.2)	10.0 (8.8-14.6)
Add 1-h cooking	11.5 (9.1-13.3)	11. 7 (9.6-14.2)	11.8 (10.6-15.9)	11.9 (10.7-16.3)
Add 2-h cooking	13.4 (11.1-15.1)	13.4 (11.5-16.0)		
	NO_2 ($\mu g\ m^{-3}$)			
Before lockdown	16.9 (6.7-29.9)	19.2 (8.4-29.6)	28.5 (23.0-40.2)	18.2 (13.2-23.1)
After lockdown	13.4 (6.2-23.6)	14.1 (7.5-21.4)	23.2 (21.8-32.9)	14.9 (11.3-18.7)
Add 1-h cooking	15.8 (8.9-26.6)	16.4 (10.1-23.9)	25.2 (21.1-34.4)	17.2 (13.7-20.9)
Add 2-h cooking	18.0 (11.4-27.4)	18.7 (12.6-25.3)		

Background

We have used the London Hybrid Exposure Model (LHEM) (Smith et al., 2016), to assess the change in exposure of London's population as a consequence of the Covid-19 lockdown. The LHEM model predicts the exposure of people indoors and outdoors, including whilst travelling (train, tube, bus, car, cycle and walk) and includes indoor sources such as cooking. The model can predict the exposure of the entire London population and, in this analysis, we assumed that the 'before lockdown' exposures reflected typical activity during 2019. In contrast the 'after lockdown' exposures reflect a 53% reduction in road traffic, 73% reduction in aviation, changes to people's daily work activities, increased time spent at home and changes to indoor cooking activities. Specifically, we have added to the 'after lockdown' case an additional 1 hour of cooking activity per day to reflect increased levels of home cooking replacing alternatives such as restaurants and home delivery. In this case cooking activity means an hour longer spent in the kitchen environment. We have not added other changes to important indoor sources such as wood burning. We have selected specific population subgroups for this analysis: children and tube users, whose response to the lockdown is assumed to be to spend all of their time at home[1], and professional drivers and hospital staff, who are assumed to continue to work as normal[2].

Model evaluation and assumptions

[1] Owing to the lack of data on exercise patterns, it has been assumed that the children and tube user subgroups now spend all their time indoors at their home address. Tube users have been defined as anyone spending 20 minutes or more on the London underground on the day for which they completed the LTDS. Children have been defined as anyone under the age of 16

[2] For the purpose of this study, the driver category reflects the daily activity of bus, taxi and delivery drivers. These occupations were identified by answers given in the LTDS. The hospital staff category reflects the people who spent more than 7 hours at one of 43 London Hospitals on the day for which they completed the LTDS.

Estimating human exposure using the London Hybrid Exposure Model indoors, outdoors and in transit requires a detailed combination of outdoor and indoor air pollution concentrations, combined with a detailed knowledge of where, when and how people travel and where they live and work.

Home, work and travel information were taken from the London Travel Demand Survey (LTDS), an anonymised, comprehensive survey undertaken by Transport for London at ~8000 households each year and representative of the entire London population. The people selected for this analysis were all surveyed in the spring months of March-May.

Outdoor air pollution estimates ($PM_{2.5}$ and NO_2) were taken from King's London Air Quality Toolkit Model as average hourly concentrations every 20m across London. The model was evaluated against ~100 air pollution stations from kerbside to suburban background, giving r and normalised bias estimates of 0.7-0.8 and 7% and 0%, respectively.

Indoor air pollution estimates were calculated for kitchen and living rooms using indoor/outdoor measurements of 78 ($PM_{2.5}$) and 89 (NO_2) London households and incorporated into the LHEM mass balance model. Exposure concentrations to cooking emissions are estimated by averaged concentrations in kitchens during cooking hours (7-8 pm) from measurements of 16 London households.

Indoor travel concentrations were predicted using a mass balance approach, combining outdoor concentrations with vehicle air exchange rates, loss rates, area/volume estimates and occupancy, from the wider literature. Estimates of PM exposure on the tube were based around average measurements on the underground, described by Smith et al., 2020.

Model uncertainties

Whilst we have been careful to evaluate many of the LHEM model components, and to use robust indoor measurements where possible, there are still uncertainties surrounding our exposure estimates. Examples include uncertainty in transport exposure, where evaluation is limited to comparisons with the wider literature. Furthermore, due to the limited time available, the estimate of 1 hour of additional cooking is a working assumption to demonstrate the importance of indoor sources. Additional sources of $PM_{2.5}$/NO_2 in offices, hospitals and schools have also not been accounted for in the model.

Current LHEM model developments

- Tube line specific exposure measurements are currently being implemented in LHEM.
- Further assessment of in-vehicle exposure to black carbon is underway as part of the DeMIST project at KCL.
- A more comprehensive evaluation of indoor exposure is being developed, combining modelled indoor/outdoor ratios and the aforementioned household measurements. Further analysis of changes in domestic cooking activity would also be beneficial.
- Estimates of children's exposure in London and the effect on their cognitive development is being undertaken as part of the MRC CLUE project.

- Estimates of changes to exposure brought about by the ULEZ in London and Birmingham is also being undertaken as part of the APEX project.

References

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Appendix 1. Histograms of human exposure before and after lockdown

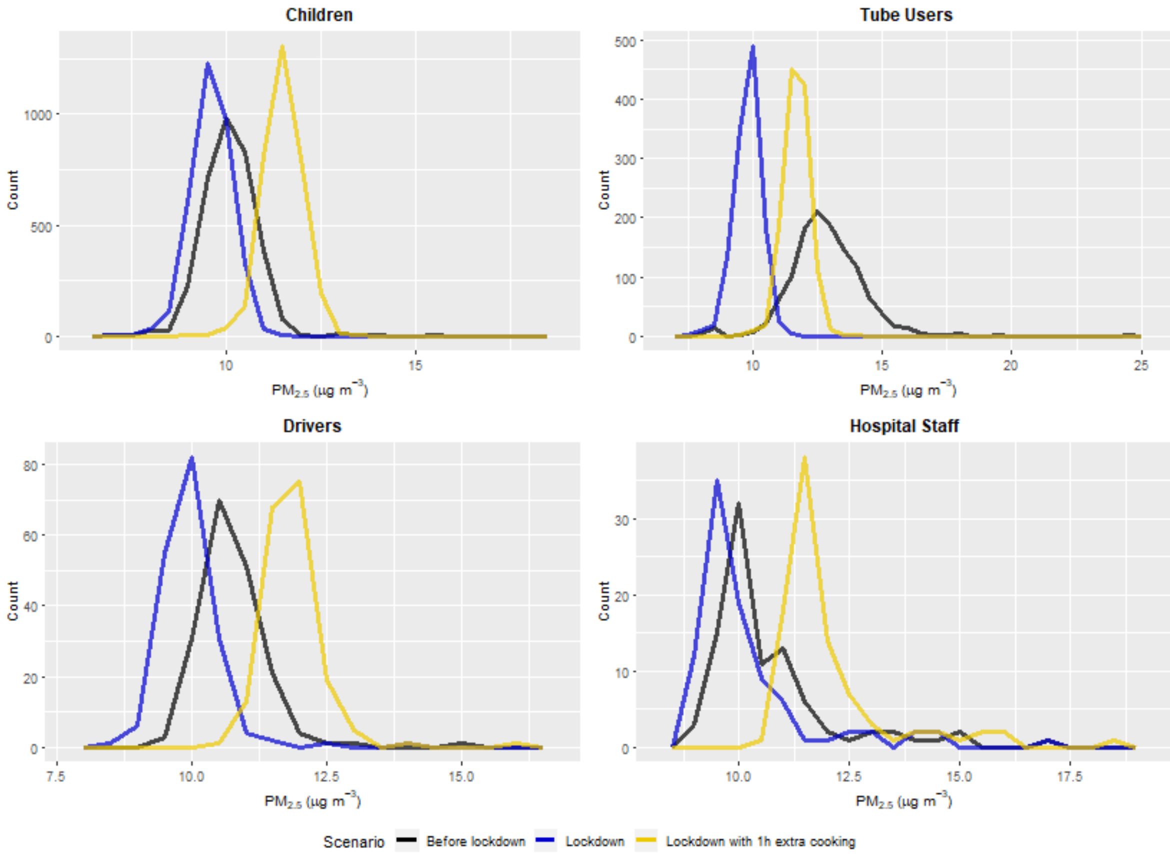

Fig.1. Histograms comparing modelled human exposure to $PM_{2.5}$ before and after the lockdown in different subgroups, with the "1h extra cooking" scenario also included.

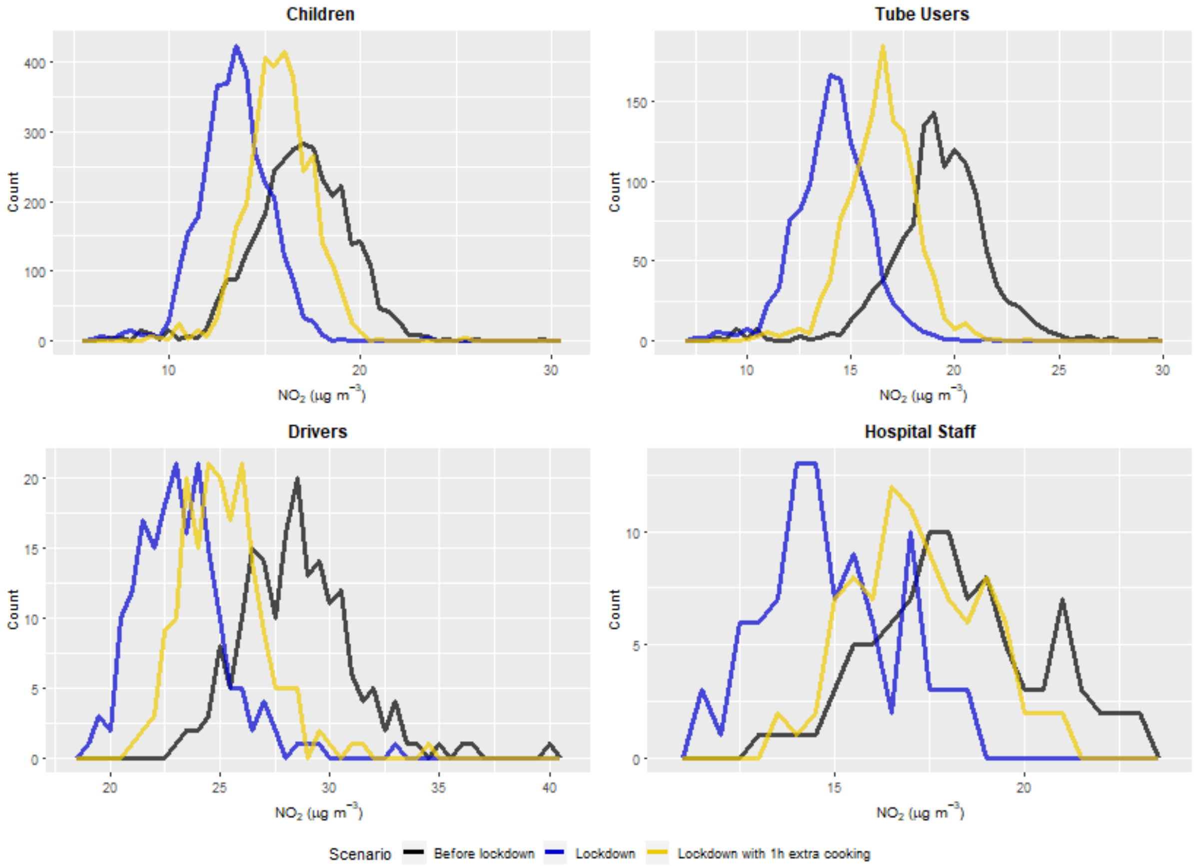

Fig.2. Histograms comparing modelled human exposure to NO_2 before and after the lockdown in different subgroups, with the “1h extra cooking” scenario also included.

COVID-19 effects on emissions and regional PM concentrations in the UK

A submission of evidence to Defra/AQEG.

Contributors: Eiko Nemitz, Marsailidh Twigg, Massimo Vieno, Carole Helfter, Christine Braban, Stefan Reis, Sarah Leeson, Matthew Jones, Ivan Simmons and Duncan Harvey: UK Centre for Ecology and Hydrology, Bush Estate, Penicuik, EH26 0QB; en@ceh.ac.uk

In collaboration with: Janet Barlow, Meteorology, School of Mathematical, Physical & Computational Sciences, University of Reading (BT Tower flux measurements) and Sanocka, A., Dernie, J. Ritchie, S., Conolly, C., Ricardo, Harwell, UK (Chilbolton MARGA data)

Note: all data are provisional and have not been fully ratified. Moreover, the model results have been calculated using a collection of forecasted meteorology rather than re-analysis.

Summary:

- Preliminary analysis of direct flux measurements of CO_2 made from the BT Tower suggests that Central London CO_2 emissions have closely followed the % reductions in road traffic and have declined by about 55% compared with previous years.
- Concentration measurements suggest that widespread, regional $PM_{2.5}$ pollution events have continued to occur since the start of lockdown and these have been dominated by ammonium nitrate. This demonstrates that (a) considerable emissions of ammonia and NOx must be continuing and (b) meteorology is a key driver of pollution events.
- Model scenarios of emission reductions suggest that even reductions in NOx emissions of nearly 50% across the UK and Europe (but excluding shipping) trigger only a small and non-linear response in ammonium nitrate concentrations.
- The model results so far confirm that a wide range of emission sources of primary PM and PM precursors, not just traffic, need to be addressed to effectively reduce $PM_{2.5}$ concentrations and they re-emphasise the importance of the need to reduce agricultural ammonia emissions to reduce spring-time pollution events.
- The COVID-19 lockdown period is a 'natural experiment' that provides the opportunity to test our understanding of emissions and their contribution to $PM_{2.5}$, which will be exploited further as more measurement data and as more realistic emission scenarios based on real-world changes in activity data become available.

1. *Changes in London's CO_2 and CH_4 emissions*

Pollutant fluxes have been measured on the BT Tower by eddy-covariance since 2011 in a collaboration between UKCEH and the Universities of Reading and York. These provide an integrated picture of the emission from a "flux footprint" that extends to typically 8 km upwind of the tower and, over time, maps out of the emissions of much of central London (Helfter et al., 2016). Unlike concentrations, micrometeorological flux measurements are independent of background concentrations and are largely unaffected by meteorological conditions and therefore provide a direct measure of the change in emissions. Figure 1 shows the evolution of the reduction of the weekly average daytime (defined as 12:00 to 18:00 hrs) emissions compared with the average emission in the same calendar week of previous years (closed symbols), in relation to the traffic counts measured in Central London (Westminster and City; open symbols).

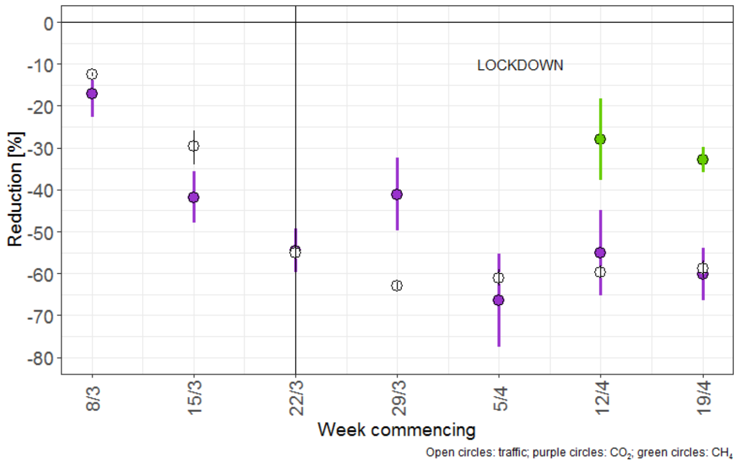

Figure 1. Weekly average emission reduction during the lockdown period compared with previous years for CO_2 (purple circles) and CH_4 (green circles), in relation to reductions in traffic flow for Central London (open circles). CH_4 fluxes were only restarted w/c 12-April.

The reductions in CO_2 daytime emission since 22-March averages 55% and closely follows the 59% reduction in traffic in the Central London (Westminster and City). Traffic would be expected to be the dominant CO_2 source, especially during daytime hours, but this comparison of emissions during the COVID-19 lockdown period with previous years also reflects changes in the terrestrial CO_2 sink (e.g. photosynthesis) which varies between years as a function of plant phenology and meteorology, and changes in natural gas combustion. Natural gas use will have decreased as commercial heating and the daytime population are likely to have declined within the footprint. According to CENSUS 2011 data, normally the daytime population in central London exceeds the night-time population by a factor of 10. Although the time-series is shorter, emissions of CH_4 have decreased by about 30% compared with previous years, and this originates largely from leakage from the gas supply network (which depends on supply pressure, but should otherwise be independent of use) and slippage during ignition (Helfter et al., 2016). Emerging data on gas usage and supply pressure offers the opportunity to use the COVID-19 to improve our understanding of the different sources to the CH_4 flux.

Unfortunately, the measurement period included periods where the CO_2 flux instrument was somewhat malfunctioning and a correction had to be applied. Now reinstated measurements with a second instrument will over time assess the robustness of the measurements to date.

2. Analysis of regional PM pollution episodes in the UK during the COVID-19 lockdown period

Whilst traffic emissions have decreased, the UK has experienced a number of $PM_{2.5}$ pollution episodes during the COVID-19 lockdown period so far. In fact, analysis of the $PM_{2.5}$ concentrations at Chilbolton, one of the two UK's EMEP Supersites, measured by FIDAS suggests that all 7 exceedances of the daily $PM_{2.5}$ limit of 25 µg m^{-3} observed in 2020 to date have occurred since lockdown commenced. Figure 2 shows the concentrations of inorganic aerosol chemical components within the $PM_{2.5}$ as measured at the two UK EMEP Supersites (Chilbolton and Auchencorth Moss) with the MARGA method (e.g. Twigg et al., 2015) and demonstrates that during the high-concentration episode aerosol chemical composition was largely dominated by nitrate (in blue) and associated ammonium (in orange). Comparison with a total $PM_{2.5}$ measurement by FIDAS (black line), highlights the amount of additional aerosol mass not resolved by the MARGA, which includes carbonaceous aerosols (organic aerosol compounds and black carbon) and some crustal material. The contribution of these additional components appears to differ greatly between episodes but it needs to be borne in mind that the response of the FIDAS, being based on an optical measurement, also depends on the nature of the aerosol. These measurements confirm the notion that regional pollution episodes are not dominated by traffic emissions and that emissions of key $PM_{2.5}$ aerosol precursors (such as ammonia) have remained largely unaffected by lockdown.

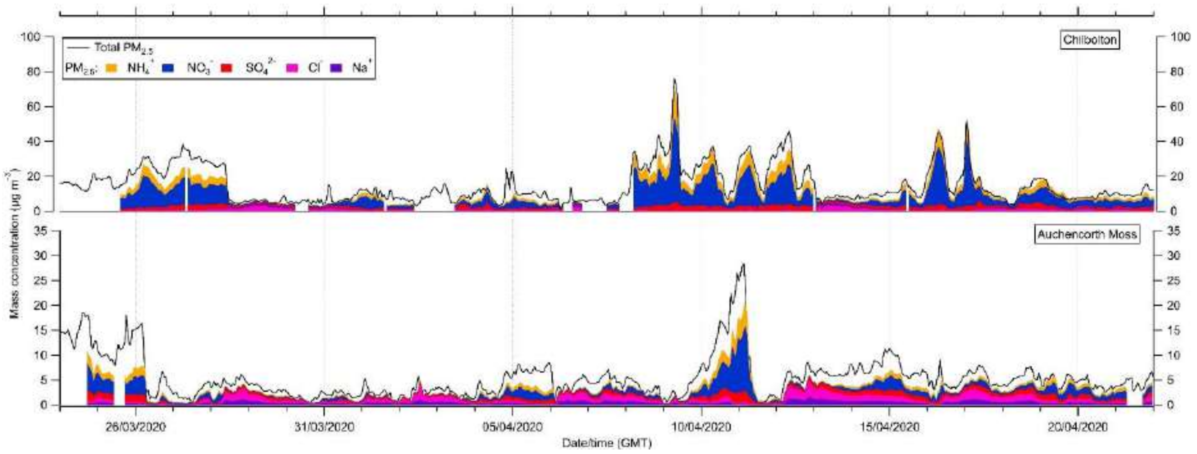

Figure 2. Non-ratified inorganic chemical composition of the $PM_{2.5}$ aerosol (measured by MARGA) at the Chilbolton and Auchencorth Moss EMEP Supersites over the COVID-19 lockdown period in relation to total $PM_{2.5}$ concentration (measured by FIDAS), covering a number of pollution episodes.

3. Model sensitivity investigations of regional $PM_{2.5}$ responses to COVID-19 induced emission reduction scenarios

The EMEP4UKrv4.34-WRFv4.1.1 atmospheric chemistry and transport modelling system (e.g. Vieno et al., 2016) was used to explore the response of individual aerosol chemical components under a number of plausible but simplified emission reduction scenarios. In addition, to the BASE run using standard 2016 NAEI emissions for the UK, 2015 FMI and ECLIPSEv6a for shipping, and the 2015 ECLIPSEv6a for the rest of the European model domain, three emission scenarios were considered as summarised in Figure 3. Also, natural fires were not included in these simulations.

COVID	- All UK & European emissions from SNAP Sector 2 (Non-industrial Combustion Plant): +20% - All UK & European emissions from SNAP7 (Road Transport): -70% - All UK emission from SNAP8 (Other mobile sources & machinery): -70%
COVID2	- As COVID - Plus additionally: UK and European emissions SNAP3 (Combustion in Manufacturing) and SNAP4 (Production Processes): -70%
COVID3	- Developed to be consistent with the MetOffice modelling work - Across all sectors for UK and Europe: NO_2: -70%; Primary PM: -50%; SO_2: -30%

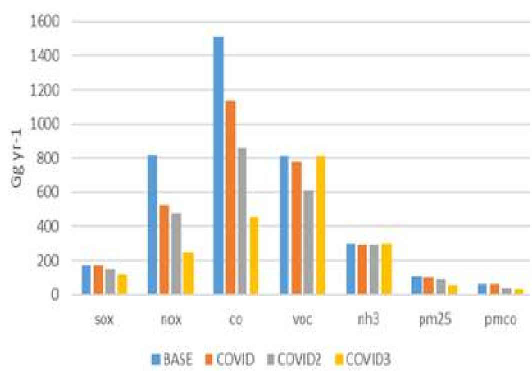

Figure 3. Definition of scenarios and resulting equivalent annual emissions for the UK (pmco = PM_{10} – $PM_{2.5}$).

Overall, the EMEP4UK model reproduces the timing and general magnitude of the $PM_{2.5}$ pollution episodes observed at Chilbolton generally well, which confirms that these are due to the meteorological conditions rather than the timing of particular emissions events (Figure 4). The changes between the different emission scenarios provide useful information on the sensitivity of PM components to precursor gas emissions during the lockdown period. With few exceptions, the modelled concentrations based on the COVID and COVID2 emission scenarios differ little from the BASE scenario, for total $PM_{2.5}$, and even for nitrate, despite NOx emissions being almost halved under the COVID2 scenario. This suggests that nitrate concentrations do not linearly change with country NOx emissions, confirming earlier sensitivity runs (Vieno et al., 2016; AQEG, 2015) with a real-world 'experiment'. Only for the COVID3 scenario, where NOx emissions are reduced by 70%, do we start to see a significant reduction in nitrate (and ammonium). Future work will need to confirm that this is due to the magnitude of the emission reduction rather than the spatial distribution of the emission reductions. In particular, Chilbolton is reasonably close to the English Channel and shipping NOx emissions were reduced in COVID3 but not in the other scenarios.

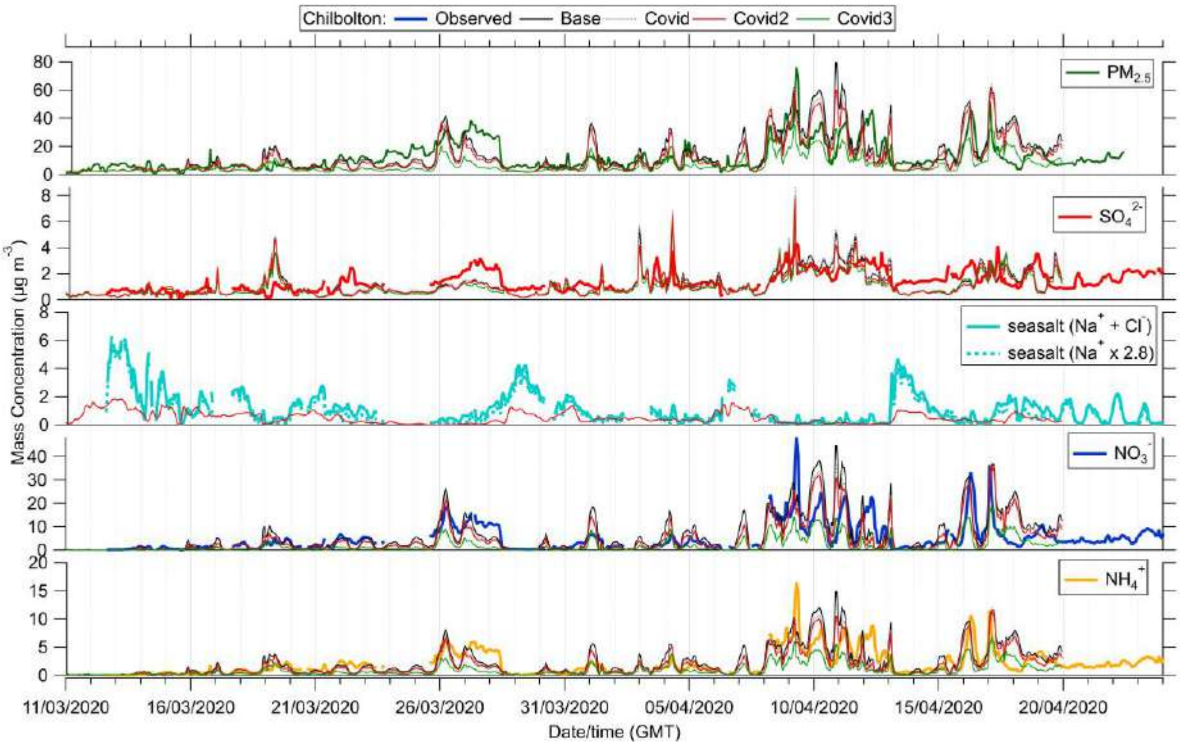

Figure 4. Concentrations of total $PM_{2.5}$ and individual $PM_{2.5}$ inorganic chemical components modelled for four different emission scenarios, compared with observations at Chilbolton (measured by the FIDAS and MARGA).

The detailed comparison between model results and measurements should not be over-interpreted. For example, even in the BASE scenario the results are based on generic temporal profiles and do not take into account real-time activity data such as the timing of agricultural activities in relation to weather conditions or the actual activity of coal-fired power plants which has been very low since 8 March (Drax Electric Insights). Nevertheless, for what it is worth, the model / measurement intercomparison suggests that the COVID3 scenario significantly underestimates the emissions required to reproduce the measured concentrations on most days.

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AMMONIA IN A TIME OF COVID-19

A submission of evidence to Defra/AQEG.

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In collaboration with Ricardo EE (Defra/EA UKEAP Contract) & the University of Manchester (NERC SPF OSCA project)

Note All 2020 data shown are provisional and have not been fully ratified

Summary

- Ammonia gas (NH_3) is a priority pollutant both as a precursor to particulate matter and for ecosystem impacts
- Three scenarios for UK emission reductions during COVID-19 in emission sectors, where activity is likely reduced ,have been assessed
- Total UK emissions of NH_3 are likely to have decreased slightly (~2%), which is within the uncertainty and meteorological variability of the UK atmosphere
- Urban background and urban on road and roadside emissions of NH_3 are likely to have decreased, by as much as 30% and 90% respectively compared with usual emissions before COVID-19
- Unratified data from three of the five UK automatic NH_3 analysers (Auchencorth Moss, Chilbolton Observatory, and Manchester OSCA Observatory) show typical springtime NH_3 concentrations across the UK.
- Data from the non-automatic National Ammonia Monitoring Network will enable analysis at UK level in the months ahead. This includes roadside data from London Cromwell Rd.
- Evidence gaps & future approaches are outlined. Future analysis of the Defra UKEAP rural networks proposed.
- The key measurement gap is urban roadside NH_3 (and PM ammonium) as there is only one long-term site in the UK measuring roadside NH_3 concentrations. It is suggested that a roadside network of samplers and/or analysers are urgently put in place (perhaps aligned with the UK Urban NO_2 Network; UUNN) to monitor NH_3 at roadsides during and post COVID-19 lock down where possible.

Background

Atmospheric ammonia (NH_3) gas is gaining increasing importance in the global pollution climate, with effects at local to international (transboundary) scales (see Tang et al. 2018 for a detailed overview UK NH_3 science). NH_3 is emitted in gaseous form and has a short atmospheric lifetime on the order of hours to a few days. It is primarily emitted at ground level in the rural environment; however, sources of NH_3 in an urban environment include, but are not limited to, emissions from humans and their activities: pets, vehicle exhausts, industrial processes including fertiliser manufacture, cigarette smoking, sewage, fertiliser, compost, soils, wild animals, landfill sites, combustion and household products. There are also concerns that the ammonia emission potential of the fleet when Selective Catalytic Reduction (SCR) retro-fits will increase transport based NH_3 emissions. There is already evidence of increasing use of three-way catalysts in cars (Borsari & de Assunção 2017) and SCR systems (Stelwagen & Ligterink 2015) reduces the emissions of nitrogen oxides (NO_x), but increases emissions of NH_3. (Reche et al. 2012). There are also tentative suggestions use of ammonia as a fuel for transport (e.g. McInley et al., 2020) could further affect future emissions.

Estimated emissions changes during COVID-19 lockdown.

Using the National Atmospheric Emissions Inventory (NAEI) 2017 emissions (see Data Sources below), three NH_3 emission reduction scenarios have been simply estimated. The approach taken was to assess which sectors might be expected to have reduced emissions due to decrease human activity during the COVID-19 shutdown. This was done at the UK, urban and urban-roadside scale (Table 1). Details of the sector changes are provided in the Appendix, but are primarily transport and industry related. The three scenarios looked at were 30, 50 and 90% reductions in NH_3 emissions in the sectors affected (low medium and high reductions).

From this exercise, UK-wide emission were found to be only slightly reduced even in the 90% reduction scenario for the affected source sectors, with an overall emission reduction of 2.6%. This is primarily due to the dominance of agricultural emissions in the UK NH_3 budget (estimated at 88% of national emissions, UK Clean Air Strategy 2019). When the set of emission sectors were narrowed to urban emissions, covering urban transport, domestic emissions etc. the low to high reduction scenarios resulted in a decrease of 10-30%. Focussing still further on urban roadside the NH_3 emission reductions were in the range 30-90% (see Appendix for sectors). It was assumed that smoking which might normally have been expected to occur by roads for some of the time was taken as being completely non-roadside due to the movement restrictions). This assessment is only a high-level estimate and a detailed emissions assessment would be appropriate.

Table 1 Estimated reduction of A: UK total NH_3 emissions, B: UK urban NH_3, and C: "on-road" NH_3 emissions under a low, medium and high ammonia emission reduction scenarios

UK 2017 NH_3 all emissions	294.3 ktonnes	Low reduction (30%)	Medium reduction (50%)	High reduction (90%)
Amount in sectors likely to reduce in ktonnes (% total)	8.57 (2.9%)	6.65	4.09	0.82
Amount in UK urban sectors likely to be reduced (ktonnes)		2.26	1.39	0.28
Relative reduction of total UK NH_3		0.65%	1.52%	2.63%
UK 2017 NH_3 "urban" emission	6.5 ktonnes	Low reduction	Medium reduction	High reduction
Amount in UK urban sectors likely to be reduced (ktonnes)		5.86	5.40	4.49
Relative reduction of UK urban NH_3		10.4%	17.4%	31.2%
UK 2017 NH_3 "urban roadside" emission	2.0 ktonnes	Low reduction	Medium reduction	High reduction
Amount in UK urban road transport sectors reduced (ktonnes)		1.40	1.00	0.20
Relative reduction of UK urban ***on-road*** NH_3		30.0%	50.0%	90.0%

Possible impact on UK NH_3 concentrations

Data from the two rural air quality supersites (Figure 1A) show that rural NH_3 concentrations are typical of Spring levels observed in previous years (as shown for Auchencorth Moss in Figure 1B), which is consistent with the estimate above that the COVID-lockdown will have minimal impact on NH_3 emissions nationally. The 72 sites in the UK National Ammonia Monitoring Network provide monthly average NH_3 concentrations. These will be able to provide the national picture once samplers are analysed (estimated data availability Sept 2020, depending on laboratory re-opening).

There are three urban background sites measuring NH_3 within the NERC SPF Clean Air OSCA project, Honor Road Park, Birmingham and Manchester. The NH_3 data from the Manchester site is also shown in Figure 1A with Auchencorth Moss and Chilbolton data for March and April 2020. Indicatively the concentration variation falls between the agriculturally influenced Chilbolton site and the background site at Auchencorth Moss (the three sites lie very approximately on a north-south transect). Once the data have been ratified from all five UK automatic NH_3 sites, coupled with wind direction and windspeed data, an assessment may give insight into the relative importance of local urban NH_3 and transported rural air masses. This may enable the relative importance of the urban NH_3 emission reductions on background urban concentrations during the COVID-19 shutdown to be assessed.

The UK currently has a long-term urban NH_3 monitoring site at London Cromwell Road. The site is part of the UK Eutrophying and Acidifying Atmospheric Pollutant networks, within the National Ammonia Monitoring Network. Figure 1D shows the data from the past twenty years (1999-2019). The annual mean concentration is 3.08±0.64 $\mu g.m^{-3}$ with peak concentrations occurring in spring and summer. The London Cromwell Road monitoring station is located in the southwest corner of the front gardens of the Natural History Museum at a traffic light controlled crossroads. The nearest road, the A4 Cromwell Road, is approximately 4 m from the monitoring station. The traffic flow on Cromwell Road is normally approximately 53,000 vehicles per day (information taken from UK-Air site April 2020). The surrounding area is generally open but there are trees occur within 2 metres distance of the monitoring station. Data will not be available for the COVID-19 shutdown period until September 2020 (depending on the COVID-19 regulations). Data from this site, being a highly traffic-emission dominated NH_3 site, will be a good test of the emission reduction estimate. If the estimation is accurate, then the NH_3 concentration may be as low as ~0.3 $\mu g.m^{-3}$. However, as above, it is noted, there are significant uncertainties associated with this estimate. Historical data from previous research campaigns are shown in Figure 1C. Concentrations at Cromwell Road fall between those at Marylebone Road and those at the urban background site at North Kensington. Data from the BT Tower show the much lower concentrations observed at altitude compared to ground-level measurements.

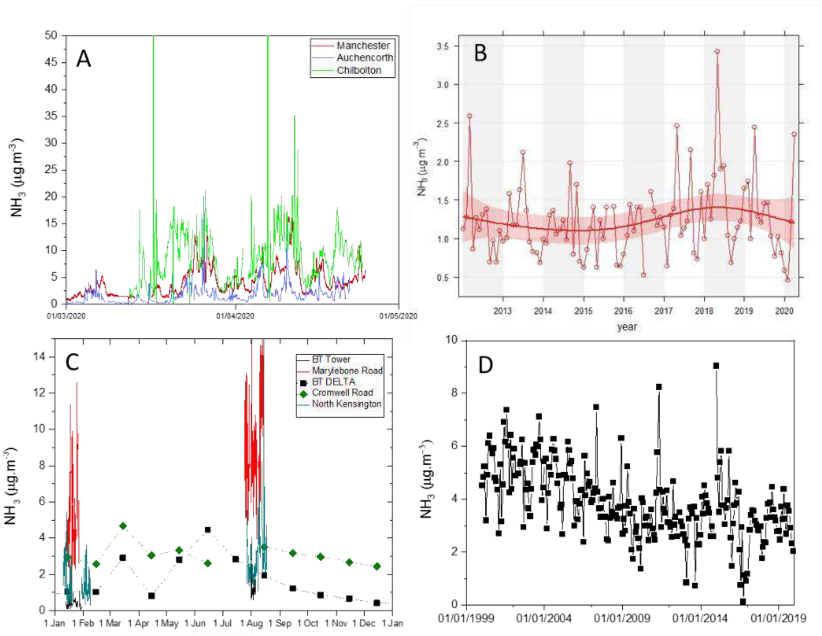

Figure 1 A: NH_3 concentrations over March and April 2020 for Chilbolton, Auchencorth Moss and Manchester OSCA Observatory (note all data provisional and unratified), B: Trend in ammonia concentration at Auchencorth Moss 2012-2020; C: 2012 NER Clearflo (BT Tower, Marylebone Road and N Kensington) and UKEAP NAMN London Cromwell Rd NH_3 data; D Defra/EA National Ammonia Monitoring Network site London Cromwell Road 1999-2019 monthly passive (ALPHA© sampler) data

During COVID-19, ammonia concentrations are likely to be reduced at the roadside, but in general, only small reductions in NH_3 concentrations would be observed at urban background sites given the relatively small magnitude of the transport sector NH_3 emissions. This is indicatively evidenced by the Manchester data (Figure 1A).

Implications of urban NH_3 concentration reductions

The impact of reducing urban on-road ammonia emissions (at the same time as reducing all other vehicular emissions) is largely unknown. NH_3 gas partitions into aerosol and water droplets in the form of ammonium, as well as being transported away through the atmosphere and dry deposition on surfaces. Ammonium is a major component of UK $PM_{2.5}$. A major reduction in roadside NH_3 is likely to have the largest impact locally due to the small magnitude of on-road NH_3 emissions compared to the total ammonia emission budget. However, although urban NH_3 emissions are small, they are spatially collocated with high NO_x emissions and are therefore, on a per mass basis, more efficient in acting as $PM_{2.5}$ precursors than emissions in the rural environment. The introduction of more SCR systems to remove NO_x emissions is predicted to increase the on-road NH_3 emissions. Therefore understanding this local atmospheric chemistry and the impacts of the changing chemical climate is a priority.

There is likely a direct interaction with PM within metres of the roadside as well as significant dispersion. Ammonium in particles have also been linked with processing of organic chemicals and therefore any significant reduction in ammonium may change the chemical properties of the PM (Montoya-Aguilera et al. 2018, Strangl et al., 2017; Huang et al. 2018). However it is noted that PM chemical processing of ammonium with respect to organic compounds is a less well studied area of atmospheric chemistry.

Key action for evidence

Given the absence of NH_3 and NH_4 measurement at roadside locations in the UK, unless a short term investment in installing monitoring locations, little evidence will be available to assess the changes of urban ammonia with the reduction of human activity and transport (and hence better quantify the on-road emissions when traffic returns to the roads).

There is an opportunity, if acted upon immediately to gain a baseline of urban NH_3 in the UK, prior to the hypothesised increase in emissions from transport.

> ***A***: *The National Ammonia Monitoring Network and the UUNN networks could potentially co-locate to gather NO_2 and NH_3 in roadside locations for spatial NH_3 concentration data*
>
> ***B***: *roadside automatic instruments deployed for high temporal resolution data.*
>
> ***C:*** *Simultaneous gaseous NH_3 and aerosol NH_4^+ measurements to understand ammonium nitrate formation*

Key questions to address and investigate once data are available

- Further assessment of the differences in emission of UK urban and rural NH_3 and its partitioning into PM
- Assessment of whether London Cromwell Road site could be upgraded to an automatic site given the data record from 1999 to present and its unique NH_3 measurement dataset
- ***Analyse of the Acid Gas and Aerosol Network gas phase (SO_2, HNO_3) and PM inorganic composition data*** to assess the UK national distributions and changes in atmospheric composition. (27 sites across UK).
- Use of the UKEAP supersite and rural network data to ***verifying modelled speciation of reduced and oxidised nitrogen*** and assess changes to nitrogen deposition from NH_3, PM and precipitation (UKEAP data).
- **Use of the 2020 evidence and current models to drive a step change in understanding the formation of ammonium nitrate** in urban areas and the role of transport compared to other emission sectors.
- It is uncertain whether the reduction in urban ammonia will have positive impacts for nature and human health. There is a modelling and literature knowledge gap which might usefully be undertaken to understand benefits (and hence inform future air quality scenario assessment within the Clean Air Strategy).
- Analysis of urban NH_3 reduction co-benefits for waste management, nature and human health

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Appendix

A Summary of NH_3 emission sectors which may be expected to reduced during the COVID-19 lockdown with a high, medium and low reduction scenario

NFR/CRF Group	Source	Activity	Activity	2017 emission (ktonne)	COVID reduction possible?	Why	Low reduction (30%) (ktonne)	Low reduction (50%) (ktonne)	High reduction (90%) (ktonne)
1A1a	Miscellaneous industrial/commercial combustion	MSW	MSW	0.01023504	Y	Non-essential Businesses closed	0.00716453	0.00511752	0.0010235
1A2f	Cement - non-decarbonising	Clinker production	Clinker production	0.38038336	Y	Building work stopped	0.26626835	0.19019168	0.03803834
1A2gvii	Industrial off-road mobile machinery	DERV	DERV	0.00261058	Y	Non-essential Businesses closed	0.00182741	0.00130529	0.00026106
1A2gvii	Industrial off-road mobile machinery	Gas oil	Gas oil	0.01221137	Y	Non-essential Businesses closed	0.00854796	0.00610569	0.00122114
1A2gvii	Industrial off-road mobile machinery	Petrol	Petrol	0.00083819	Y	Non-essential Businesses closed	0.00058674	0.0004191	8.3819E-05
1A2gviii	Other industrial combustion	Biomass	Biomass	2.24963059	Y	Non-essential Businesses closed	1.57474142	1.1248153	0.22496306
1A3bi	Road transport - cars - rural driving	Petrol	Petrol	1.48370697	Y	Reduced travel	1.03859488	0.74185348	0.1483707
1A3bi	Road transport - cars - rural driving	DERV	DERV	0.33069308	Y	Reduced travel	0.23148516	0.16534654	0.03306931
1A3bi	Road transport - cars - urban driving	Petrol	Petrol	0.2651019	Y	Reduced travel	0.18557133	0.13255095	0.02651019
1A3bi	Road transport - cars - urban driving	DERV	DERV	0.15904565	Y	Reduced travel	0.11133196	0.07952283	0.01590457
1A3bi	Road transport - cars - motorway driving	Petrol	Petrol	1.18518976	Y	Reduced travel	0.82963283	0.59259488	0.11851898
1A3bi	Road transport - cars - motorway driving	DERV	DERV	0.16382688	Y	Reduced travel	0.11467882	0.08191344	0.01638269
1A3bi	Road transport - cars - cold start	Petrol	Petrol	0.31381645	Y	Reduced travel	0.21967151	0.15690822	0.03138164
1A3bi	Road transport - cars - cold start	DERV	DERV	0.07665138	Y	Reduced travel	0.05365597	0.03832569	0.00766514
1A3bii	Road transport - LGVs - rural driving	Petrol	Petrol	0.02556061	Y	Reduced travel	0.01789243	0.01278031	0.00255606
1A3bii	Road transport - LGVs - rural driving	DERV	DERV	0.10847675	Y	Reduced travel	0.07593372	0.05423837	0.01084767
1A3bii	Road transport - LGVs - urban driving	Petrol	Petrol	0.00461532	Y	Reduced travel	0.00323072	0.00230766	0.00046153
1A3bii	Road transport - LGVs - urban driving	DERV	DERV	0.05170443	Y	Reduced travel	0.0361931	0.02585221	0.00517044
1A3bii	Road transport - LGVs - motorway driving	Petrol	Petrol	0.02207364	Y	Reduced travel	0.01545155	0.01103682	0.00220736
1A3bii	Road transport - LGVs - motorway driving	DERV	DERV	0.04708146	Y	Reduced travel	0.03295702	0.02354073	0.00470815
1A3bii	Road transport - LGVs - cold start	Petrol	Petrol	0.00359981	Y	Reduced travel	0.00251987	0.0017999	0.00035998
1A3bii	Road transport - LGVs - cold start	DERV	DERV	0.02522601	Y	Reduced travel	0.01765821	0.01261301	0.0025226
1A3biii	Road transport - buses and coaches - rural driving	DERV	DERV	0.00384734	Y	Reduced travel	0.00269314	0.00192367	0.00038473
1A3biii	Road transport - HGV articulated - rural driving	DERV	DERV	0.05176142	Y	Reduced travel	0.03623299	0.02588071	0.00517614
1A3biii	Road transport - HGV rigid - rural driving	DERV	DERV	0.05433467	Y	Reduced travel	0.03803427	0.02716733	0.00543347
1A3biii	Road transport - buses and coaches - urban driving	DERV	DERV	0.0070264	Y	Reduced travel	0.00491848	0.0035132	0.00070264
1A3biii	Road transport - HGV articulated - urban driving	DERV	DERV	0.00850976	Y	Reduced travel	0.00595683	0.00425488	0.00085098
1A3biii	Road transport - HGV rigid - urban driving	DERV	DERV	0.02238787	Y	Reduced travel	0.01567151	0.01119393	0.00223879
1A3biii	Road transport - buses and coaches - motorway driving	DERV	DERV	0.00105715	Y	Reduced travel	0.00074	0.00052857	0.00010571
1A3biii	Road transport - HGV articulated - motorway driving	DERV	DERV	0.07994018	Y	Reduced travel	0.05595812	0.03997009	0.00799402
1A3biii	Road transport - HGV rigid - motorway driving	DERV	DERV	0.03425915	Y	Reduced travel	0.02398141	0.01712958	0.00342592
1A3biv	Road transport - motorcycle (>50cc 2st) - rural driving	Petrol	Petrol	0	Y	Reduced travel	0	0	0
1A3biv	Road transport - motorcycle (>50cc 4st) - rural driving	Petrol	Petrol	0.00376958	Y	Reduced travel	0.00263871	0.00188479	0.00037696
1A3biv	Road transport - mopeds (<50cc 2st) - urban driving	Petrol	Petrol	0.0001393	Y	Reduced travel	9.7508E-05	6.9649E-05	1.393E-05
1A3biv	Road transport - motorcycle (>50cc 2st) - urban driving	Petrol	Petrol	0.0001071	Y	Reduced travel	7.4969E-05	5.3549E-05	1.071E-05
1A3biv	Road transport - motorcycle (>50cc 4st) - urban driving	Petrol	Petrol	0.00422388	Y	Reduced travel	0.00295672	0.00211194	0.00042239
1A3biv	Road transport - motorcycle (>50cc 4st) - motorway driving	Petrol	Petrol	0.00072605	Y	Reduced travel	0.00050823	0.00036302	7.2605E-05
1A3c	Railways - intercity	Gas oil	Gas oil	0.00205897	Y	Reduced travel	0.00144128	0.00102949	0.0002059
1A3c	Railways - regional	Gas oil	Gas oil	0.00233646	Y	Reduced travel	0.00163552	0.00116823	0.00023365
1A3c	Railways - freight	Gas oil	Gas oil	0.00168318	Y	Reduced travel	0.00117823	0.00084159	0.00016832

1A3c	Rail - coal	Coal	Coal	0.0021	Y	Reduced travel	0.00147	0.00105	0.00021
1A3dii	Shipping - coastal	Fuel oil	Fuel oil	0.00166216	Y	Reduced travel	0.00116351	0.00083108	0.00016622
1A3dii	Shipping - coastal	Gas oil	Gas oil	0.01207051	Y	Reduced travel	0.00844936	0.00603525	0.00120705
1A3dii	Sailing boats with auxiliary engines	DERV	DERV	1.69E-05	Y	Reduced travel	1.1833E-05	8.4524E-06	1.6905E-06
1A3dii	Sailing boats with auxiliary engines	Gas oil	Gas oil	0	Y	Reduced travel	0	0	0
1A3dii	Sailing boats with auxiliary engines	Petrol	Petrol	0	Y	Reduced travel	0	0	0
1A3dii	Motorboats / workboats (e.g. canal boats, dredgers, service boats, tourist boats, river boats)	DERV	DERV	0.00078679	Y	Reduced travel	0.00055075	0.00039339	7.8679E-05
1A3dii	Motorboats / workboats (e.g. canal boats, dredgers, service boats, tourist boats, river boats)	Gas oil	Gas oil	0.00031994	Y	Reduced travel	0.00022396	0.00015997	3.1994E-05
1A3dii	Motorboats / workboats (e.g. canal boats, dredgers, service boats, tourist boats, river boats)	Petrol	Petrol	0.00034049	Y	Reduced travel	0.00023835	0.00017025	3.4049E-05
1A3dii	Personal watercraft e.g. jet ski	DERV	DERV	0	Y	Reduced travel	0	0	0
1A3dii	Personal watercraft e.g. jet ski	Gas oil	Gas oil	0	Y	Reduced travel	0	0	0
1A3dii	Personal watercraft e.g. jet ski	Petrol	Petrol	0.00013644	Y	Reduced travel	9.551E-05	6.8221E-05	1.3644E-05
1A3dii	Inland goods-carrying vessels	DERV	DERV	0	Y	Reduced travel	0	0	0
1A3dii	Inland goods-carrying vessels	Gas oil	Gas oil	1.51E-05	Y	Reduced travel	1.0586E-05	7.5613E-06	1.5123E-06
1A3dii	Inland goods-carrying vessels	Petrol	Petrol	0	Y	Reduced travel	0	0	0
1A3dii	Shipping between UK and Gibraltar	Fuel oil	Fuel oil	0.00015182	Y	Reduced travel	0.00010628	7.5912E-05	1.5182E-05
1A3dii	Shipping between UK and OTs (excl. Gib and Bermuda)	Fuel oil	Fuel oil	4.67E-05	Y	Reduced travel	3.2676E-05	2.334E-05	4.6679E-06
1A3dii	Shipping between UK and Bermuda	Fuel oil	Fuel oil	4.53E-06	Y	Reduced travel	3.1703E-06	2.2645E-06	4.5289E-07
1A3dii	Shipping between UK and CDs	Fuel oil	Fuel oil	2.54E-06	Y	Reduced travel	1.7785E-06	1.2703E-06	2.5407E-07
1A3dii	Shipping between UK and CDs	Gas oil	Gas oil	6.48E-05	Y	Reduced travel	4.5386E-05	3.2418E-05	6.4837E-06
1A3eii	Aircraft - support vehicles	Gas oil	Gas oil	0.00141825	Y	Reduced travel	0.00099277	0.00070912	0.00014182
2H1	Paper production	Process emission	Process emission	0.005	Y	Reduced travel	0.0035	0.0025	0.0005
z_11C	Adult breath and sweat	Population	Population	0.95152789	Y	Reduced outdoor activity	0.66606952	0.47576394	0.09515279
	TOTALs			8.57345008			6.64967824	4.09231119	0.81846224
	Fraction of UK annual emission								

B Summary of NH_3 urban emission sectors which may be expected to reduced during the COVID-19 lockdown with a high, medium and low reduction scenario

NFR/CRF Group	Source	Activity	Units	2017 emission (ktonne)	COVID reduction possible?	Why	Low reduction (30%) (ktonne)	Low reduction (50%) (ktonne)	High reduction (90%) (ktonne)
1A2f	Cement - non-decarbonising	Clinker production	kilotonne	0.38038336	Y	Building work stopped	0.26627	0.19019	0.03804
1A3bi	Road transport - cars - urban driving	Petrol	kilotonne	0.265101903	Y	Reduced travel	0.18557	0.13255	0.02651
1A3bi	Road transport - cars - urban driving	DERV	kilotonne	0.159045654	Y	Reduced travel	0.11133	0.07952	0.01590
1A3bi	Road transport - cars - cold start	Petrol	kilotonne	0.313816449	Y	Reduced travel	0.21967	0.15691	0.03138
1A3bi	Road transport - cars - cold start	DERV	kilotonne	0.07665138	Y	Reduced travel	0.05366	0.03833	0.00767
1A3bii	Road transport - LGVs - urban driving	DERV	kilotonne	0.051704427	Y	Reduced travel	0.03619	0.02585	0.00517
1A3bii	Road transport - LGVs - cold start	Petrol	kilotonne	0.00359981	Y	Reduced travel	0.00252	0.00180	0.00036
1A3bii	Road transport - LGVs - cold start	DERV	kilotonne	0.025226013	Y	Reduced travel	0.01766	0.01261	0.00252
1A3biii	Road transport - buses and coaches - urban driving	DERV	kilotonne	0.007026405	Y	Reduced travel	0.00492	0.00351	0.00070
1A3biii	Road transport - HGV articulated - urban driving	DERV	kilotonne	0.008509759	Y	Reduced travel	0.00596	0.00425	0.00085
1A3biii	Road transport - HGV rigid - urban driving	DERV	kilotonne	0.022387869	Y	Reduced travel	0.01567	0.01119	0.00224
1A3biv	Road transport - mopeds (<50cc 2st) - urban driving	Petrol	kilotonne	0.000139297	Y	Reduced travel	0.00010	0.00007	0.00001
1A3biv	Road transport - motorcycle (>50cc 2st) - urban driving	Petrol	kilotonne	0.000107098	Y	Reduced travel	0.00007	0.00005	0.00001
1A3biv	Road transport - motorcycle (>50cc 4st) - urban driving	Petrol	kilotonne	0.004223885	Y	Reduced travel	0.00296	0.00211	0.00042
1A4bi	Domestic combustion	Anthracite	kilotonne	0.005265088	N	NO REDUCTION	0.00527	0.00527	0.00527
1A4bi	Domestic combustion	Coal	kilotonne	0.007833286	N	NO REDUCTION	0.00783	0.00783	0.00783
1A4bi	Domestic combustion	Coke	kilotonne	0	N	NO REDUCTION	0.00000	0.00000	0.00000
1A4bi	Domestic combustion	Wood	kilotonne	2.256142175	N	NO REDUCTION	2.25614	2.25614	2.25614
1A4bii	House and garden machinery	DERV	kilotonne	8.63E-05	N	NO REDUCTION	0.00009	0.00009	0.00009
1A4bii	House and garden machinery	Petrol	kilotonne	0.000440591	N	NO REDUCTION	0.00044	0.00044	0.00044
2D3a	Non-aerosol products - household products	Non-fuel domestic	kilotonne	1.208218406	N	NO REDUCTION	1.20822	1.20822	1.20822
2G	Cigarette smoking	Cigarettes	kilotonne	0.114809318	N	NO REDUCTION	0.11481	0.11481	0.11481
6A	House and garden machinery	Domestic fertilizer	kilotonne	0.277767857	N	NO REDUCTION	0.27777	0.27777	0.27777
6A	Infant emissions from nappies	Population 0to4yrs	kilotonne	0.038910743	N	NO REDUCTION	0.03891	0.03891	0.03891
6A	Parks, Gardens & Golf courses	Non-fuel combustion	kilotonne	0.358559535	N	NO REDUCTION	0.35856	0.35856	0.35856
z_11C	Adult breath and sweat	Population	kilotonne	0.951527889	Y	Reduced time outside	0.66607	0.47576	0.09515

C Summary of NH_3 urban on-road emission sectors which may be expected to reduced during the COVID-19 lockdown with a high, medium and low reduction scenario

NFR/CRF Group	Source	Activity	Units	2017 emission (ktonne)	COVID reduction possible?	Why	Low reduction (30%) (ktonne)	Low reduction (50%) (ktonne)	High reduction (90%) (ktonne)
1A3bi	Road transport - cars - urban driving	Petrol	kilotonne	0.265101903	Y	Reduced travel	0.18557	0.13255	0.02651
1A3bi	Road transport - cars - urban driving	DERV	kilotonne	0.159045654	Y	Reduced travel	0.11133	0.07952	0.01590
1A3bi	Road transport - cars - cold start	Petrol	kilotonne	0.313816449	Y	Reduced travel	0.21967	0.15691	0.03138
1A3bi	Road transport - cars - cold start	DERV	kilotonne	0.07665138	Y	Reduced travel	0.05366	0.03833	0.00767
1A3bii	Road transport - LGVs - urban driving	DERV	kilotonne	0.051704427	Y	Reduced travel	0.03619	0.02585	0.00517
1A3bii	Road transport - LGVs - cold start	Petrol	kilotonne	0.00359981	Y	Reduced travel	0.00252	0.00180	0.00036
1A3bii	Road transport - LGVs - cold start	DERV	kilotonne	0.025226013	Y	Reduced travel	0.01766	0.01261	0.00252
1A3biii	Road transport - buses and coaches - urban driving	DERV	kilotonne	0.007026405	Y	Reduced travel	0.00492	0.00351	0.00070
1A3biii	Road transport - HGV articulated - urban driving	DERV	kilotonne	0.008509759	Y	Reduced travel	0.00596	0.00425	0.00085
1A3biii	Road transport - HGV rigid - urban driving	DERV	kilotonne	0.022387869	Y	Reduced travel	0.01567	0.01119	0.00224
1A3biv	Road transport - mopeds (<50cc 2st) - urban driving	Petrol	kilotonne	0.000139297	Y	Reduced travel	0.00010	0.00007	0.00001
1A3biv	Road transport - motorcycle (>50cc 2st) - urban driving	Petrol	kilotonne	0.000107098	Y	Reduced travel	0.00007	0.00005	0.00001
1A3biv	Road transport - motorcycle (>50cc 4st) - urban driving	Petrol	kilotonne	0.004223885	Y	Reduced travel	0.00296	0.00211	0.00042
2G	Cigarette smoking	Cigarettes	kilotonne	0.114809318	N	NO REDUCTION (assume displaced)	0.11481	0.11481	0.11481
z_11C	Adult breath and sweat	Population	kilotonne	0.951527889	Y	Reduced time outside	0.66607	0.47576	0.09515