AIR QUALITY EXPERT GROUP

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

This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of Agriculture, Environment and Rural Affairs in Northern Ireland, on the estimation of changes in air pollution emissions, concentrations and exposure during the COVID-19 outbreak in the UK. The information contained within this report represents a review of the understanding and evidence available at the time of writing, and is based on information received via a public call for evidence.

United Kingdom air quality information received from the automatic monitoring sites and forecasts may be accessed via the following media:

Freephone Air Pollution Information Service 0800556677

Internet http://uk-air.defra.gov.uk

PB 14624

Terms of Reference

The Air Quality Expert Group (AQEG) is an expert committee of the Department for Environment, Food and Rural Affairs (Defra) and considers current knowledge on air pollution and provides advice on such things as the levels, sources and characteristics of air pollutants in the UK. AQEG reports to Defra's Chief Scientific Adviser, Defra Ministers, Scottish Ministers, the Welsh Government and the Department of Agriculture, Environment and Rural Affairs in Northern Ireland (the Government and devolved administrations). Members of the Group are drawn from those with a proven track record in the fields of air pollution research and practice.

AQEG's functions are to:

- Provide advice to, and work collaboratively with, officials and key office holders in Defra and the devolved administrations, other delivery partners and public bodies, and EU and international technical expert groups;
- Report to Defra's Chief Scientific Adviser (CSA): Chairs of expert committees will meet annually with the CSA, and will provide an annual summary of the work of the Committee to the Science Advisory Council (SAC) for Defra's Annual Report. In exception, matters can be escalated to Ministers;
- Support the CSA as appropriate during emergencies;
- Contribute to developing the air quality evidence base by analysing, interpreting and synthesising evidence;
- Provide judgements on the quality and relevance of the evidence base;
- Suggest priority areas for future work, and advise on Defra's implementation of the air quality evidence plan (or equivalent);
- Give advice on current and future levels, trends, sources and characteristics of air pollutants in the UK;
- Provide independent advice and operate in line with the Government's Principles for Scientific Advice and the Code of Practice for Scientific Advisory Committees (CoPSAC).

Expert Committee Members are independent appointments made through open competition, in line with the Office of the Commissioner for Public Appointments (OCPA) guidelines on best practice for making public appointments. Members are expected to act in accord with the principles of public life.

Further information on AQEG can be found on the Group's website at: https://www.gov.uk/government/policy-advisory-groups/air-quality-expert-group

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Introduction and Scope of the report

The Air Quality Expert Group (AQEG) in conjunction with Defra issued a call for evidence on 7th April 2020, identifying seven areas of current scientific uncertainty related to the potential interactions between COVID-19 and UK air pollution.

See: https://uk-air.defra.gov.uk/news?view=259

The initial aim of the evidence review was to support a rapid expert assessment of available data sources and analyses that had been recently completed by the academic community and the air quality consulting and management sector. The call was structured around seven questions posed by Defra:

1. 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?
2. Can you provide estimates for how emissions and ambient concentrations of NO_x, NO_2, PM, O_3, VOC, NH_3 etc. may have changed since the COVID-19 outbreak? Where possible please provide data sets to support your response.
3. What changes do you anticipate in indoor air quality as a result of the COVID-19 pandemic?
4. How might public exposure to air pollution have changed as a consequence of recent restrictions on movement?
5. How might altered emissions of air pollutants over the next three months affect UK summertime air quality?
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?
7. Are there any insights that can be gained from aerosol science on possible viral transmission mechanisms?

By close of the call on 30th April over 50 responses had been received from a range of organisations including research groups at universities and institutes, commercial organisations, industry bodies and Local Authorities. Annex 1 shows the contributing organisations. This has provided a body of information that is particularly useful for assessing emerging issues associated with changes in emissions, concentrations and exposure to air pollution since the UK lockdown was imposed during the COVID-19 pandemic.

This report has been prepared by AQEG with input from a number of *ad hoc* members and the Defra secretariat. Question 6 was passed to the secretariat and members of the Committee on the Medical Effects of Air Pollutants (COMEAP) and their response is included in this report along with details of the contributors.

It is important to stress that this report predominantly evaluates air pollution data available up to 30th April 2020. It does not draw conclusions on air quality or emissions changes that may have arisen in the UK as a result of the government easing of lockdown restrictions in May 2020.

Unusually for AQEG reports, this analysis is based predominantly on scientific and technical material that is not yet peer-reviewed, and indeed it often draws on observational data that has yet to receive final quality assurance ratification from the data providers. An expert judgement has therefore been made with regard to the weight given to different evidence sources and the associated uncertainties when drawing conclusions. In this rapidly evolving situation, it would be anticipated that a large body of peer-reviewed scientific literature will become available later in 2020, and beyond.

Initial conclusions

- There have been significant changes in the emissions of air pollutants from several sectors, but, with the exception of the transport sector which showed a marked decrease, availability of activity and emissions data for the lockdown period is still limited.
- UK air quality has been negatively influenced by a significant change in meteorology between the weeks preceding and following the lockdown in addition to changes (both positive and negative) arising from actions in response to COVID-19.
- The most pronounced changes in UK air quality during lockdown have been in the urban environment, notably for nitrogen oxides (NO_x). Once weather effects are accounted for, mean reductions in urban NO_x averaged over the lockdown period considered have been typically 30-40%, with mean NO_2 reductions of 20-30%. In general, NO_x and NO_2 reductions have been greater at roadside than at urban background sites. These reductions would typically correspond to decreases in concentrations of 10-20 $\mu g\ m^{-3}$ if expressed relative to annual averages.
- Meteorological conditions have led to higher $PM_{2.5}$ during lockdown than the average experienced in equivalent calendar periods from previous years. Analysis combining observations and models indicates however that $PM_{2.5}$ concentrations were of the order 2 - 5 $\mu g\ m^{-3}$ lower in Southern England than would have been expected under a business-as-usual emissions scenario. The changes to UK $PM_{2.5}$ in terms of contributing sources and transboundary influences have yet to be determined.
- Changes to population exposure to air pollution are variable and more uncertain than estimates of changes in ambient concentrations. Some urban locations have seen significant falls in NO_2, and wider working from home has reduced travel exposure more generally in cities. In London, initial estimates of reduction

in $PM_{2.5}$ exposure compared to business-as-usual are in the range 5-24% depending on factors such as commuting mode.

- Little is known about the impact of lockdown on indoor air quality, since homes are not routinely monitored in the UK. Whilst exposure to pollution in the workplace and during commuting will have likely reduced for many people, increased time spent on activities in the home such as cooking and cleaning may have increased emissions and concentrations of pollutants such as $PM_{2.5}$ and Volatile Organic Compounds (VOCs).
- Increased ozone has been observed at some urban monitoring stations, a result of lower local NO. Models suggest the responses in UK ozone for this summer compared to business-as-usual are variable, with no single direction of change, although there may be some modest increases in urban areas and in central and south-eastern parts of the UK.
- Long-term exposure to air pollution is associated with increased morbidity and mortality from chronic diseases, some of which have also been identified as increasing the risk of severe COVID-19 symptoms. Given this, it would not be surprising if there was a link between exposure to air pollution (past or present) and the occurrence or severity of COVID-19 infection. Whilst several unpublished studies have examined this effect, and have reported associations with past exposure to both $PM_{2.5}$ and NO_2, there is currently no consensus on the pollutant responsible or the magnitude of any effect. Such studies require very careful control for confounding influences, and further work is needed before there can be confidence in their findings.
- Very small amounts of RNA from SARS-CoV-2 have been observed in outdoor particulate matter but it is not yet known whether breathing air outdoors provides a significant route for transmission of live virus or infection. Whilst aerosol containing the virus can build up indoors in poorly ventilated rooms, dilution is rapid in an open outdoor environment, which is likely to reduce the dose of virus inhaled compared to indoors. The lifetime of active virus in the outdoor atmosphere has yet to be determined.

Evidence gaps

The call for evidence highlighted that extensive research work has already been completed in the UK quantifying reductions in air pollution during lockdown, particularly for NO_2 and its links to changes in transport and mobility patterns. It would be anticipated that a significant body of peer-reviewed literature will emerge on this topic. The evidence review has revealed a number of areas where there are however key gaps, and where generating new knowledge would be particularly valuable to inform policy and regulation.

- The impacts of lockdown on $PM_{2.5}$ may shed valuable observational constraint to support future air quality target setting in the UK. The full value of blended datasets drawing on, for example, PM chemical composition, PM precursors, network PM data and regional and urban models have yet to be realised.
- Initial modelling of summertime ozone with reduced NO_x emissions from transport is reflective of possible UK urban atmosphere around 2030. The possible changes this induces in ozone are however finely balanced in many places, and ozone modelling is notoriously non-linear. Greater confidence would be gained from coordinated multi-model assessment, validated against observations. There is a significant opportunity here for basic research to directly inform national emissions reduction plans.
- Links between atmospheric concentration response and underlying emissions activity appear robust for urban NO_x and road transport, but for many other pollutants and processes emissions and activity data are not currently available. Emissions data for 2020 at UK level from sectors such as energy, industry, commercial, and domestic will likely emerge as part of inventory reporting activities, and the research value of these activity datasets should be maximised alongside observational information and models. There will likely remain greater uncertainties on how activities and emissions have been affected at local level and over shorter time periods during the year.
- Virtually no research-grade observations are available to quantify changes to exposure to air pollution in UK homes, despite the major behavioural changes occurring during the COVID-19 lock-down. This absence of evidence on in-home exposure then adds very large uncertainties to estimates of overall population impacts. There are however, some opportunities to further explore non-traditional data sources, for example from sensors built into internet-enabled indoor air purifiers operating during this period. Whilst low cost sensor data may give some indication of past indoor air quality, establishing some basic UK observing capability appears urgent, given the possibility of further lockdown periods should virus infection rates increase.
- Changes to the emissions profile during COVID-19 will likely have fed through into not only changed overall ambient concentrations and exposure to particulate

matter, but also to changes in the size distribution and chemical composition of those particles. The impacts of these changes are not known, but the datasets already collected, including from online instruments and offline samples, provides the necessary starting point to explore this further.

- There is currently very limited evidence to support the concept of transmission of viable live SARS-Cov-2 virus *via* ambient (outdoor) particulate matter, something that requires further evaluation through a range of laboratory experiments alongside ambient sampling and analysis.

1. Expected changes in air pollution emissions

Emissions during COVID-19 lockdown to 30th April 2020

Changes in activity need to be interpreted in the context of the timeline of the introduction of lockdown in the UK: the UK government advised against all non-essential travel and contacts on 16th March 2020, closed schools and restaurants on 20th March and announced full lockdown on 23rd March. This report covers the period to 30th April. Subsequently, restrictions were partially eased in England on 13th May 2020 when government advice switched to "stay alert". Many businesses had already started to switch to homeworking in the two weeks prior to lockdown and many industries ceased or reduced operation.

To date there has been data showing the reduction in transport activities, particularly road traffic, rail services and aviation, as well as a reduction in overall energy use. In addition, there is a general consensus that the lockdown has reduced activities and therefore emissions from construction, commercial heating, combustion and processes in industry and power generation, but the underlying statistical data are still lacking. Whilst this section focuses on UK emissions it should be noted there have also been large reductions in air pollution emissions across Europe as a consequence of COVID-19. International reductions in emissions can have significant impacts on the UK particularly for longer-lived pollutants such as $PM_{2.5}$ and ozone that are subject to transboundary transport.

Transport Sector

The reduction in mobility decreased vehicle traffic by about 70% by mid-April according to Department for Transport (DfT) data (Figure 1), with various Local Authorities seeing overall reductions of 50% (London), 62% (Central London), 60 to 70% (Leeds), 60% (Newcastle), 55-60% (East & West Sussex) and 60 to 75% (Manchester) (GLA, 2020; Ropkins *et al.*, *University of Leeds*, 2020; Fawcett and Chan, *ARUP*, 2020; Jenkins *et al., Phlorium Ltd*, 2020; Alfarra *et al., University of Manchester*, 2020). Traffic volumes have since gradually increased again. Weekend reductions were greater than weekday reductions on some rural and urban roads (and in London), but weekday reductions were greater on other urban roads. The diurnal traffic pattern has changed closer to Sunday conditions and there have been changes in relative vehicle fleet composition, with likely a much smaller reduction in HGV traffic on motorways and outside town centres. Until 18th May, the congestion charge, LEZ and ULEZ charges in London were suspended to help key workers travel allowing the use of older, higher emitting vehicles in the LEZ/ULEZ charging zone.

As seen in data from the Waze For Cities Programme, the length of congested road segments in Greater London dropped by more than 75% and the free-flowing traffic and higher vehicle speeds will likely have decreased urban emission factors i.e. decreased the emissions per vehicle km for a given vehicle, a conclusion supported by the Breathelondon submission (Carruthers and Jones *et al. CERC / University of Cambridge,* 2020) which used an urban model inversion to estimate reductions of order 80% in road traffic emissions of NO_x in London (see Figure 2).

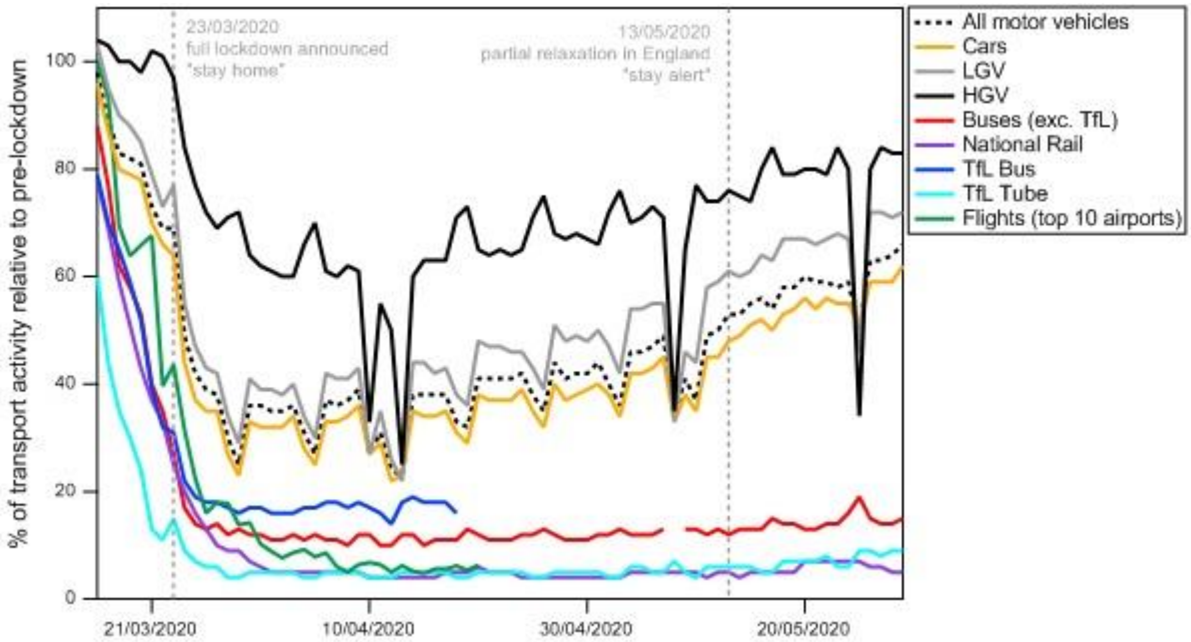

Figure 1. Relative reductions in traffic according to the data shown at the government's COVID-19 briefing on 31st May 2020 (Prime Minister's Office, 2020). The flight data were taken from the absolute values from flightradar24 data as reported by the BBC (2020) and rescaled assuming 100% activity was occurring on day 1.

Overall, the reduction in transport will have led to a marked decrease in road transport emissions of NO_x, exhaust PM and non-exhaust PM. Road transport normally accounts for 49% of the NO_x emissions in London according to the London Atmospheric Emissions Inventory (LAEI, 2016). Where LEZ/ULEZs were introduced in the past 18 months it becomes challenging to distinguish the impact of COVID-19 related restrictions from those due to ongoing changes in traffic management on traffic numbers and fleet average emission factors. However, a reduction in NO_x emissions in London has been supported by evidence for a reduction in roadside hydrocarbon concentrations at Marylebone Road, analysis of NO_x at roadside sites and CO_2 flux measurements in London, all being broadly consistent with a 50-60% reduction in traffic.

The reduction in traffic will also have led to a reduction in other pollutants emitted from vehicle exhausts, including volatile organic compounds (VOCs) and ammonia (NH_3). Although traffic is a relatively small source of NH_3 compared with agriculture, the emissions are co-located with high NO_x emissions in an urban environment and are therefore, on a per mass basis, more efficient in acting as $PM_{2.5}$ precursors than emissions in the rural environment. A submission from the UK Centre for Ecology and

Hydrology (Braban *et al.*, *UKCEH*, 2020) estimated that urban on-road and roadside emissions of NH_3 might have decreased by as much as 90% since the start of the lockdown. The introduction of more Selective Catalytic Reduction systems to remove NO_x emissions from diesel vehicle, as well as any increase in the use of petrol vehicles, is predicted to increase on-road NH_3 emissions.

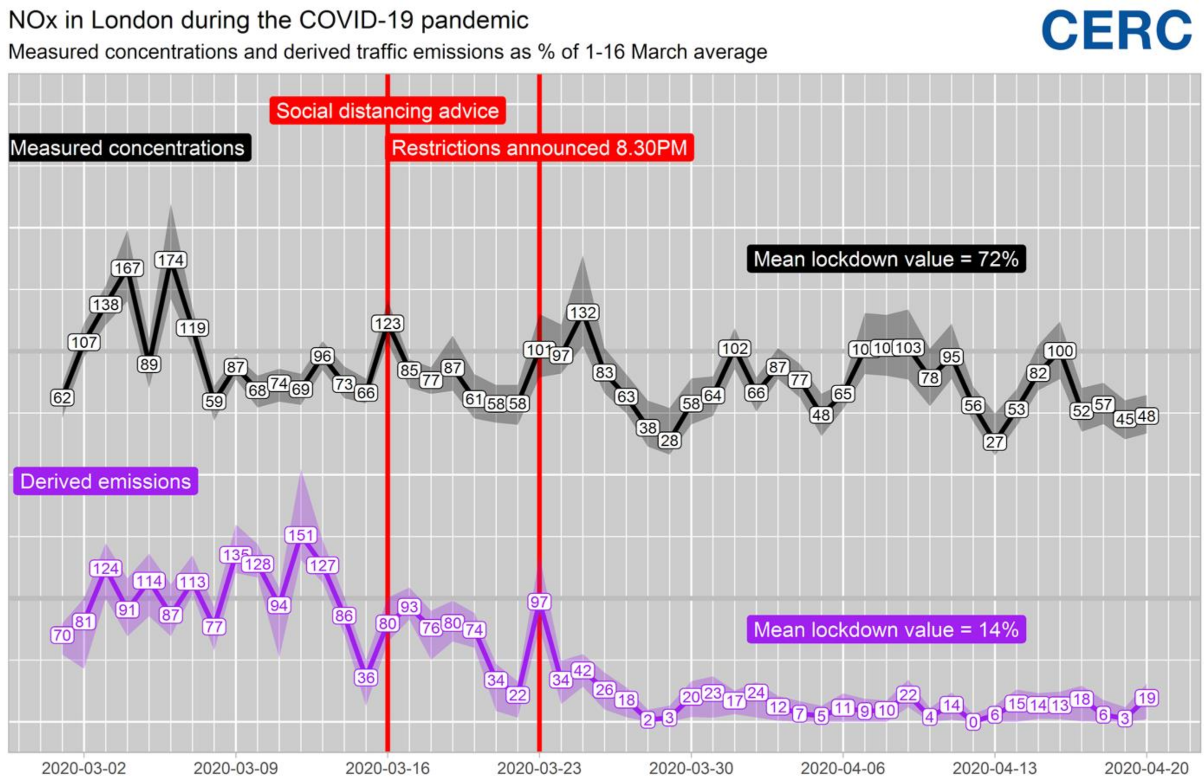

Figure 2. London measured concentrations (black) and derived road traffic emissions (purple), both as a percentage of the average over the 1-16 March pre-lockdown period. Measured concentrations are from all available sites in the LAQN, AQE and BL networks (184 sites in total); traffic emissions have been derived 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).

Reproduced from: Carruthers, D. and Jones, R.L. on behalf of Breathe London partners. *CERC and University of Cambridge*, 2020. “Breathe London submission to the Defra / AQEG COVID-19 call.”

The decrease in road traffic was mirrored by even larger declines in other transport activity (Figure 1). The drop in flight departures was accompanied by reduced airport traffic on and around the airports. Figure 1 hides the fact that a smaller relative drop at the UK’s busiest airport (Heathrow) is masking larger reductions elsewhere. Passenger rail services in terms of number of journeys had dropped by over 90% by the end of April according to DfT and by almost 40% on average in terms of mileage according to the Office of Rail and Road over the same period. According to National Rail, there has been a smaller reduction in freight services although these normally account for only 25% of the rail sector’s diesel fuel consumption. Clearly, reductions of all modes of transport started well before 23rd March.

The situation for shipping appears more complex with large decreases in activities for some vessel types (particularly offshore, passenger vessels and fishing) and little change in others, depending on cargo type. Whilst there are no confirmed data currently available, the view of the shipping industry is that there has been an overall reduction in movements globally. However, there is some evidence of ships being laid up in UK ports leaving auxiliary engines running which could increase emissions locally.

Emissions from commercial and residential properties

In particular in cities, emissions would be expected to be reduced from commercial heating and commercial cooking. There are some recognised differences between the NAEI and LAEI in the estimated size of these emissions for London. Commercial heating is estimated to account normally for 44% of the NO_x emissions and cooking for 55% of the primary $PM_{2.5}$ emissions in central London according to the LAEI (2016). Observational evidence of a reduction in $PM_{2.5}$ from cooking based on changes in ambient concentrations is not currently available; the COVID-19 lockdown period may provide a useful test for this component of emissions inventories. Nationally, according to various industry indicators for the sector, construction activity was down by around 25% in April and this sector is estimated to normally account for 35% of the PM_{10} emission in London as a whole (LAEI, 2016). Whilst not a pollutant of direct health concern, daytime CO_2 emissions have fallen by 55% in central London (Nemitz *et al., UKCEH*, 2020). This can act as a measurement-based marker of the overall reduction in fossil fuel combustion from all sources, which in turn dominate the primary emissions of NO_x and contributes to primary PM. By contrast, at the country scale, *e.g.* including industry and energy generation not located in urban centres, daily CO_2 emissions are thought to be down by 20% from typically 1.0 to 0.8 Mt CO_2 per day (Liu *et al.,* 2020).

With the move to home working, reductions of emissions in commercial areas (shops and offices) will have been partly compensated by increases from homes, but still resulting in a net decrease in most emissions, but also a change in the spatial pattern. There has not been a detectable surge in residential gas use, which is strongly influenced by ambient temperatures not just by home occupancy. Overall gas distribution is down by 11% (International Energy Agency (IEA), 2020). It is likely that future analysis of gas consumption data corrected for meteorological conditions will give further insight on the impact of restrictions on domestic gas consumption and therefore domestic heating emissions.

There is evidence that solid fuel combustion in domestic fires and stoves went up initially after lockdown and that the emissions peak shifted to later in the day. However, this initial increase is likely to have subsided with rising ambient temperatures and would also be limited by fuel stocks. Nuisance reports linked to bonfires and burning of garden waste have risen since the lockdown began despite discouragement and even banning by some local authorities (e.g. in Edinburgh, (CIEH, 2020)).

It should be noted that the reporting level by the public might also have changed: it might have gone up as people spend more time at home to notice the burning of garden waste, or down, as people have become more tolerant. It is also possible that increased public perception of a link between air quality and COVID-19 has changed reporting of local pollution events. An increase in emissions from domestic combustion is supported with evidence associated with the detection of certain markers of particulate matter from wood burning at monitoring sites in Manchester and London.

Power generation and industry

Overall electricity generation has decreased by 10-20% driven by less demand from industry and the commercial sectors (e.g. IEA, 2020), but partly offset by the residential sector. Emission reductions from the power generating sector will be larger, however, because renewables have claimed a greater share as demand has fallen. According to Drax Electric Insights the UK has been able to go without coal power since 11th April, with the COVID-19 reductions adding to a long-term trend.

Industrial emissions (including combustion in industry) are clearly down, but robust industrial output statistics are not yet available for the first quarter of 2020. It is likely that there is a larger relative reduction from smaller industries. Demand for mineral production (sand & gravel) has fallen by about 50% according to one local authority source (CIEH, 2020).

There are a number of other fairly specific sources for which emissions are likely to have changed since the lockdown began, but for which there is currently no firm supporting evidence to support this. Some sources may have shown an increase in emissions since lockdown started, while others have decreased. 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 which are affected by emissions from a very diverse range of different sources.

These potentially changed sources and the pollutants emitted are summarised in Table 1. The symbol ↑ indicates an increase in emissions may be expected, ↓ a decrease may be expected. To put the source into overall context, the table also shows the percent contribution of the source to total UK emissions in 2018, according to the National Atmospheric Emissions Inventory (NAEI, see https://naei.beis.gov.uk). The reasoning behind these anticipated changes is shown and is largely based on expert opinion, but in some cases it is supported by anecdotal evidence from industry on trends previously observed. One local authority also reported that emissions of VOCs from industrial production of solvents were declining locally by around 30%.

Emissions (e.g. of mercury) from medical waste incinerators and crematoria will have increased due to increased activity, and locally traffic around hospitals including the new

Nightingale hospitals. This will continue to be closely related to the progression of the pandemic in the UK.

Table 1: Emissions from UK sources that may have changed since the start of the COVID-19 lockdown period based on expert opinion but for which there is currently no formal emissions reporting or activity data.

Reproduced from: Murrells, T. *et al.*, *Ricardo Energy and Environment,* 2020. "Response from Ricardo to Key Questions"

Emission source	Pollutants	Directional change	% UK totals in 2018	Reasoning
Burning of garden waste	NOx $PM_{2.5}$ Benzo(a)pyrene	↑	0.02% 1.3% 0.12%	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 fewer 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
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

Agriculture and natural emissions

As with industrial statistics, agricultural output statistics are not yet available for 2020, but it is likely that agriculture has changed little over the lockdown period, leaving agricultural emissions (NH_3, primary PM, CH_4, N_2O) largely unchanged. Because NH_3 emissions are influenced by meteorology, the unusually dry and sunny April may have increased emissions compared with the long-term mean for this time of year; similarly, the dry conditions would have favoured agricultural PM emission from tilling activities. Equally not associated with COVID-19 but with meteorology, the dry April has seen a number of wildfires across the country that need to be considered when interpreting measurement data.

Overall picture of emissions including greenhouse gases

The EU H2020 CONSTRAIN project (Forster and Rosen, *University of Leeds*, 2020) has estimated changes in daily UK emissions for a range of pollutants for 6 selected sectors based on currently available proxies for activities (smart meters, mobility data from Google and Apple, flight data) (Figure 3). It is important to note that the total fractional emissions (black lines) only cover those sectors included. For example, for NH_3 the 45% reduction in emission from these non-agricultural sources is consistent with estimates of on-road and roadside emission reductions of 30% and 90% estimated by another study (Braban *et al*., *UKCEH*, 2020).. However, the agricultural sector accounts for 85% of the emission under normal conditions, so assuming that agricultural NH_3 emissions have continued unchanged (agricultural statistical data pending), total emissions would have decreased by only 2% (Braban *et al., UKCEH,* 2020).

Anticipated emission changes over the next three months and beyond

Some concerns have been raised about an increase in emissions from outdoor burning as the UK enters the warmer summer season (CIEH, 2020). This includes possible burning of commercial waste on building sites as well as fly-tipped waste. Several local authorities have expressed the need for national government to send out a stronger message to the public to store or compost garden wastes, support the re-opening of waste sites and garden waste collection, ban the use of charcoal BBQs and fire-pits in urban areas over the upcoming summer season. A moratorium on prescribed muirburn was also suggested.

The evolution of air pollution emissions in the near future will depend on how the government decides to ease restrictions and society’s response. With the government advice currently (in May 2020) to avoid public transport and car sharing incompatible with social distancing, there is some risk that traffic volumes might increase above pre-lockdown levels despite continuation of home working by many. If the advice against foreign travel stays in place, rural areas could see unusual levels of traffic associated with holidaying in the UK.

In the longer run, emissions will also depend on the economy and the “new normal” society might adapt to. This applies to the level of aviation, homeworking (and related commuter traffic), use of delivery services and a potential long-term increase in walking and cycling. Traffic levels have already started to increase again (as of end May 2020) with the easing of lockdown restrictions. Traffic may return to pre-lockdown levels to a different extent in different areas. Relative changes in daily regional mobility data from the Google Community Mobility Reports provides a breakdown by category (grocery stores, parks, residential, workplaces, transit stations, and retail) in changes to daily movement at a regional level. As the UK comes out of lockdown measures, these considerations are particularly important as some regions of the country will be more adept to supporting larger portions of their workforce staying at home, therefore maintaining lower level of emissions, than others. Areas with a high proportion of manufacturing work, where the workforce is less able to work from home, may experience a greater rate of increase in home-to-work traffic than those areas with less manufacturing and more office-based employment.

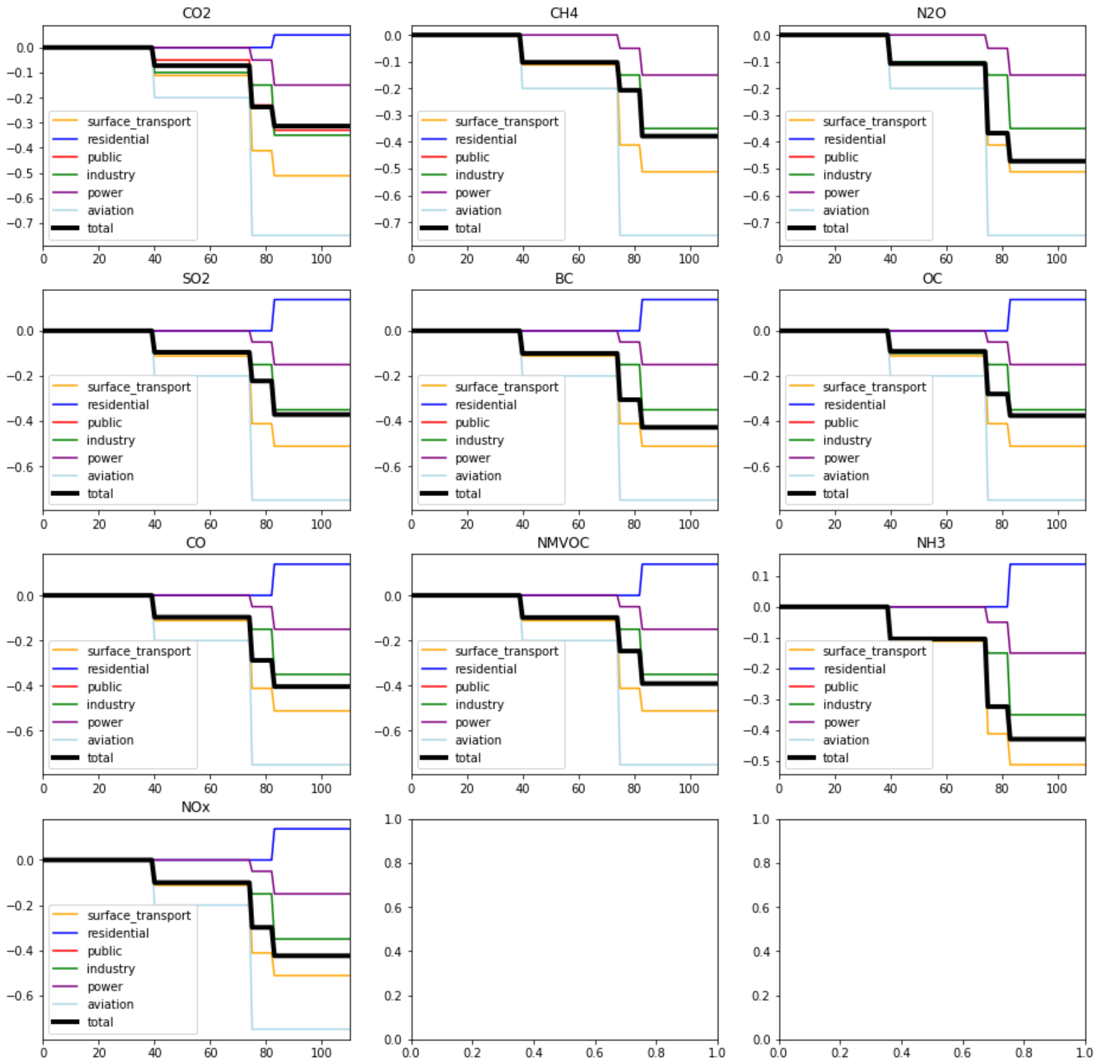

Figure 3. Fraction of UK pollutant emissions from selected sectors vs days from start of the pandemic. Note: the total does not reflect the country total as not all sectors are included.

Reproduced from: Forster, P. and Rosen, D.Z. *University of Leeds*, 2020. “Submission to DEFRA on COVID-19 by the EU H2020 CONSTRAIN project.”

New car sales have fallen substantially in recent months and if they remain low this will impact on the future rate of vehicle fleet turnover, slowing down the penetration of lower emitting vehicles in the fleet, resulting in larger emissions in future years than originally projected. One NAEI-based study cited by Murrells *et al. Ricardo*, (2020) estimated a potential 4% increase in urban NO_x emissions for 2021 compared with the current (pre-lockdown) forecast resulting from the slower fleet turnover, assuming the traffic flow projections remain unaffected

2. How have ambient concentrations of NOx, NO_2, PM, O_3, VOC, NH_3 etc. changed since the COVID outbreak?

Meteorological Context

When considering the analysis of ambient pollutant concentrations and their potential change due to COVID-19 interventions, it is important to understand the general climatology. The weather during 2020 up to May has been exceptional in many ways across the UK. February 2020 was characterised as having much higher wind speeds than long-term average values and high rainfall levels. In the 6 weeks following the lockdown date, most of the UK experienced a high proportion of easterly air flows, very low rainfall (reducing wet deposition) and significantly higher than average sunshine hours (potentially increasing photochemistry). Figure 4 shows the wind rose for London Heathrow split by the before and after periods.

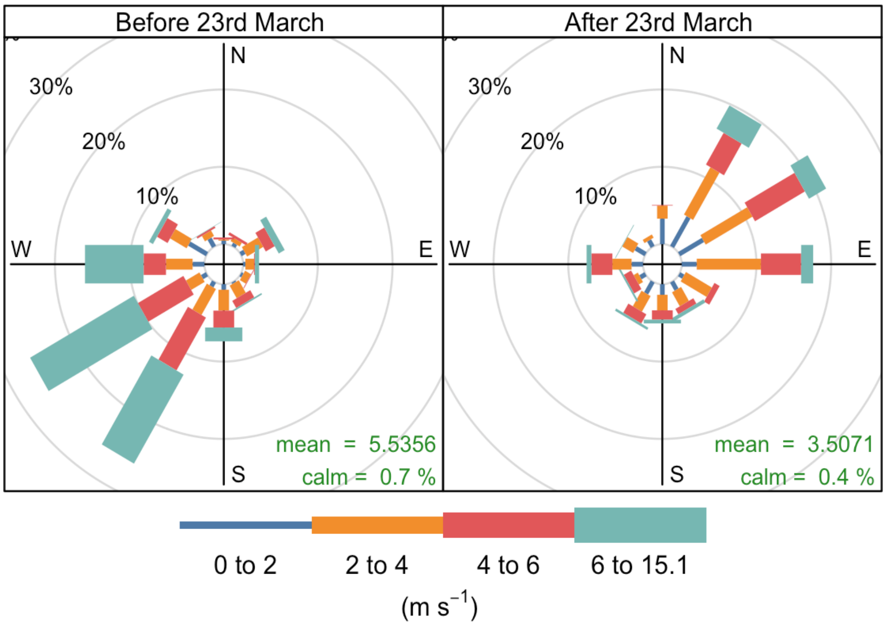

Figure 4. Wind roses from London Heathrow from 1 January 2020 to 18 May 2020 split by before and after the lockdown date.

These meteorological factors are of direct importance when considering changes in pollutant concentrations due to COVID-19 for all sites and all pollutants. Furthermore, the lockdown period covering late March, April and most of May also coincides with a time of year where O_3 concentrations tend to be elevated in the UK and there is an increased frequency of regional $PM_{2.5}$ pollution episodes from mainland Europe. In addition, the combination of the application of fertilizers and manure spreading which increases NH_3 emissions and concentrations at this time of year with still relatively cool temperatures is also of importance for $PM_{2.5}$. In this respect, the timing of the lockdown and the high frequency of winds from Europe could well prove useful in the longer term for understanding how continental reductions in emissions affect the UK air pollution picture.

The responses to the AQEG call for evidence cover a wide range of measurement approaches including the analysis of data from continuous analysers from the AURN, local authority data, low cost sensors and measurements made by satellites. Additionally, air quality models have been used either directly or in tandem with measurements to understand potential changes.

Approaches to analysis

Several types of analysis can be broadly identified:

- Simple before / after comparisons making direct use of measurement data. These types of approach have the benefit of simplicity but can be strongly influenced by meteorological and seasonal changes. Many of the responses from local authorities fall into this category. A variation on this approach involves the direct comparison of measurements from paired roadside and background instruments, which provides some control for confounding effects.
- Comparison of a post-lockdown period with a summary of previous periods e.g. the average (or variation) of the 5 previous years. Many organisations used this or a similar approach. Most of these studies show that the period after lockdown had markedly lower concentrations of pollutants such as NO_x and NO_2. These approaches have the benefit of averaging-out previous meteorological variability through the consideration of several years of data. On the other hand, for pollutants such as NO_x and NO_2 there has been a clear downward trend at most urban / roadside sites over the past few years, which means the most recent measurements would tend to show lower concentrations than previous periods irrespective of any additional changes due to COVID-19 actions specifically.
- The use of 'meteorological normalisation' (sometimes referred to as *deweathering*') to account for the variation in concentrations due to changes in meteorology. These methods use statistical models to explain concentrations in terms of commonly measured (or modelled) variables such as wind speed, wind direction and ambient temperature. Submissions included the use of multi-linear regression, Boosted Regression Trees and Random Forests. These methods have been used in two ways: to provide an estimate of pollutant concentrations if COVID-19 actions had not taken place, and to run many simulations to average the meteorological variation over the period of interest. The former approach attempts to provide a counterfactual i.e. estimated concentrations of pollutants had COVID-19 not occurred.
- Many studies also considered the evidence of 'change-points' in the concentrations of different species. These studies provide an extra level of

inference in that the timing of the change(s) calculated in pollutant time series can be compared with the lockdown date or changes in activity data such as traffic flows.

For some pollutants and some locations such as roadside NO_x, the scale of the reduction in concentrations (over 50%) is large enough to reasonably be detected by most of the methods described above, including satellite measurements described below. The difficulty arises when attempting to robustly quantify these changes because biases can easily be introduced depending on the approach used e.g. due to varying meteorological conditions and imperfect models.

Figure 5 shows how meteorology can mask the underlying trends in concentrations. The first column shows the relative variation in measured daily mean NO_2 concentrations in 2020 across 225 UK monitoring sites. The second column then shows the equivalent concentrations following statistical analysis to remove the variability caused by differences in the weather. While there is an obvious trend for lower measured NO_2 concentrations at many sites following the lockdown, the scale of this reduction only becomes clear after removing the confounding effects of meteorology.

Pollutants and data sources considered

Most of the responses considered changes in NO_x, NO_2 and O_3 concentrations from the AURN and other local networks. However, there are also examples of data from sensor networks being used and two examples of DOAS (Differential Optical Absorption Spectroscopy). Less attention was paid to PM_{10} and $PM_{2.5}$. There were also a few examples of the analysis of speciated data on PM composition and VOC measurements. Overall, the analysis of data reflects both its ubiquity and accessibility.

It seems likely that the lockdown, and its subsequent easing, will provide significant opportunities for future analyses to better understand atmospheric chemistry in the UK and, in particular, the role of road transport. These future analyses will be reliant on current measurements, and the point was made that some pollutants (notably NH_3) may not be optimally represented in roadside monitoring networks at present.

Satellite measurements of air quality

Several of the responses consider the use of satellite measurements, focusing mostly on column densities of NO_2 using the Sentinel-5p satellite. These studies have the advantage of providing useful spatial information on the distribution of NO_2 that cannot be provided by point sampling methods. However, there are many challenges in working with satellite data, some of which are common to all approaches used to analyse the effect of COVID-19. For example, the analysis of satellite data is still prone to meteorological variation, which is an issue for most approaches. There have been some specific limitations in retrieving column NO_2 data from satellites for the pre-lockdown periods of February and early March 2020 when there was extensive cloud cover over the UK. Whilst satellite observations showed anecdotal examples of lower NO_2 on selected days pre-lockdown there was a lack of quantitative information on the before-after changes from the submissions received. With more data and more time to analyse satellite data it is likely that improved quantitative information will become available, potentially also for NH_3.

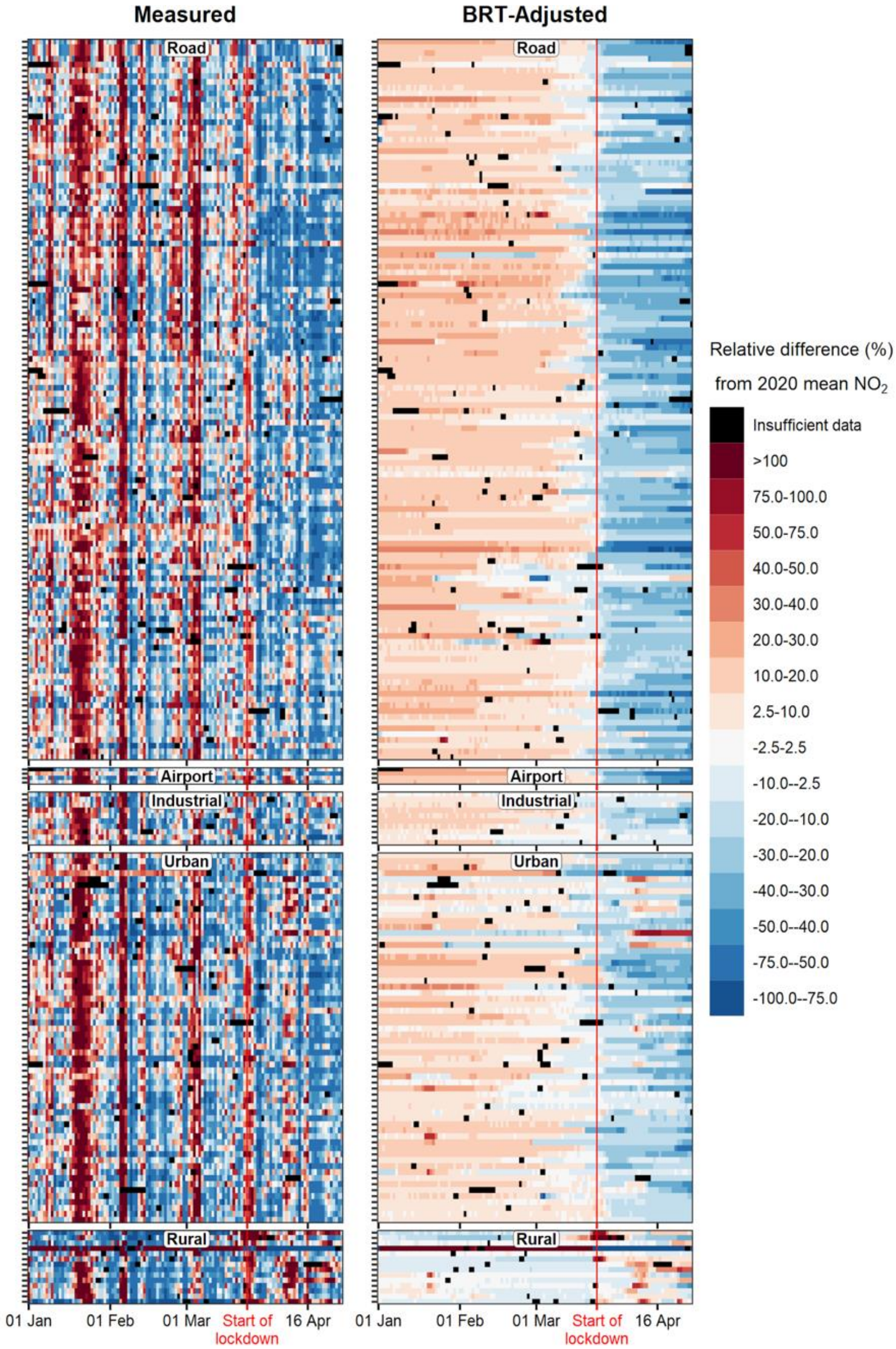

Figure 5. Relative change in daily-mean NO_2 between 1st Jan and 29th April 2020 at 225 monitoring sites on the AURN and other UK Networks. Each row of pixels represents a single site, with individual sites denoted by tick marks on the *y* axis. First column (left) shows raw measured data. Second column (right) shows equivalent values with the effects of meteorological variability removed using Boosted Regression Trees through the *deweather* package in openair.

Reproduced from Marner, B., Laxen D., Gellatly, R., Liska, T. *Air Quality Consultants*, 2020 "Response to AQEG Request for Rapid Evidence on COVID-19 & UK Air Quality."

Use of models to estimate changes in concentrations

Several submissions used air quality models to provide insight into changes in ambient concentrations of different pollutants. These models range from local (street level) to regional scales. Two main approaches were used. The first approach used models to run simulations of estimated emission reduction scenarios to explore the response in concentration of different pollutants. The challenge here is that at this time the actual change in emissions is still somewhat uncertain and must be estimated.

The second more sophisticated methods were to conduct inverse modelling to infer an emission reductions that could be explained based on observed concentration values, and also the use of machine learning approaches to develop a predicted counterfactual (similar to the meteorological normalisation methods). The initial findings from local-scale model inversion are promising because they can be related in a direct way to specific changes in emissions required to explain the observations.

A more complex GEOS-GF global forecasting model that was 'corrected' using machine learning approaches to explain sub-grid scale measurements (e.g. for roadside monitoring locations and data), suggested that there was approximately a 30% reduction in NO_2 across a wide range of sites, similar to many of the analyses of air quality network data.

Changes to NO_x and NO_2 concentrations

Many submissions provided quantitative information on how concentrations of pollutants changed. This information focused mostly on the changes in the concentrations of NO_x, NO_2 and O_3. A direct comparison between the studies is frustrated by the different periods and sites considered, and the methods used to quantify the change. A summary of the key studies providing quantitative information is given in Table 2. Despite issues related to the analysis approaches, Table 2 reveals some good consistency in relative concentration changes across monitoring sites.

Figure 6 summarises the relative changes in calculated NO_2 during the lockdown from the four studies which applied some form of deweathering to multiple sites to calculate comparable statistics. Each of these studies takes a different methodological approach and also focuses on different monitoring periods. It is nevertheless clear that each of the studies shows quite similar results: both when comparing individual monitoring sites and when viewed more broadly across the UK as a whole.

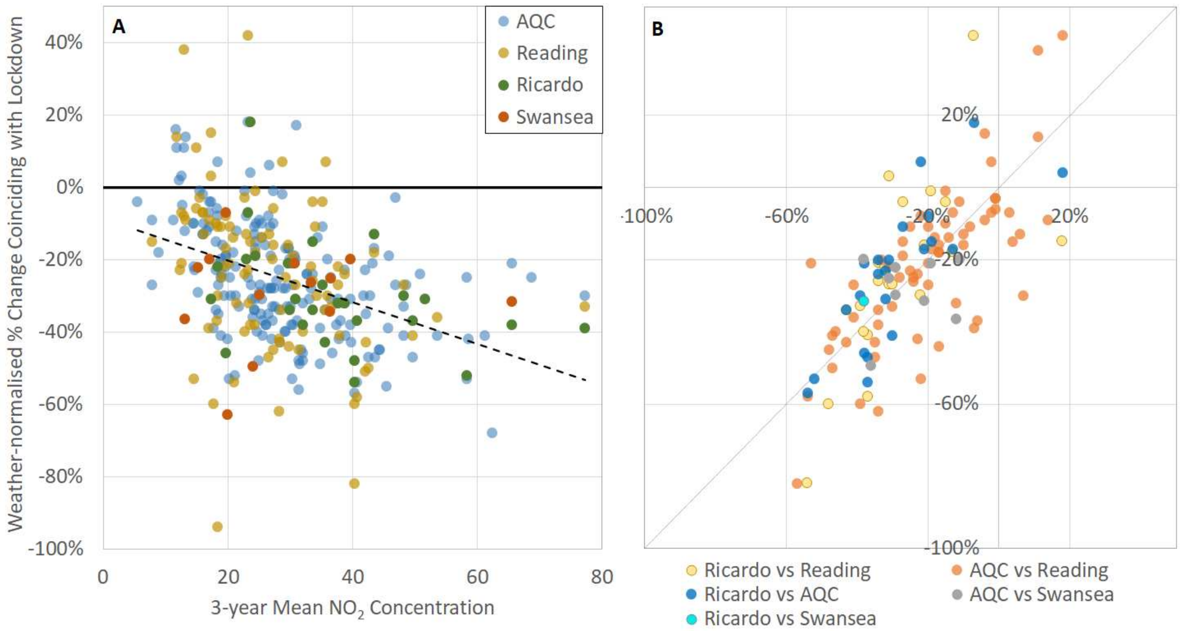

Figure 6. Changes in mean NO_2 during lockdown at 224 non-rural UK monitoring sites calculated using four alternative approaches to weather normalisation: A) Relative change in mean NO_2 during lockdown vs long-term (Jan 2017- Dec 2019) mean NO_2; and B) Comparison of approaches where equivalent sites are used (*x* axis shows the change calculated by the first listed study, y axis shows the second listed study). AQC, Ricardo and Reading all present % changes and these have been used directly. The Swansea study presents absolute changes which have been expressed relative to the 3-year mean.

Data reproduced from: Marner, B. *et al. Air Quality Consultants*, 2020; Murrells, T. et al., *Ricardo Energy and Environment,* 2020; Lewis, P.D., Seller, V., Price, T., and Eskandari, H., *Swansea University*, 2020; and Dacre, H., and Mortimer, H., *University of Reading / STFC*, 2020

In general, there have been greater reductions in NO_x than in NO_2 and other air pollutants. Analyses which have applied some form of deweathering, and which have included substantial numbers of sites, show mean reductions in urban NO_x, averaged over the period of lockdown considered, of typically 30-40% and mean NO_2 reductions of 20-30%. These reductions might typically correspond to decreases in NO_2 concentrations of 10-20 μg m^{-3} if expressed relative to annual means.

Analyses of changes in NO_x and NO_2 based directly on measured data tend to suggest greater relative changes than those undertaking some form of normalisation, presumably reflecting confounding issues of weather and/or seasonal and longer-term trends on air pollutant concentrations.

There is a trend for reductions in NO_x and NO_2 to be greatest where NO_x and NO_2 concentration were high, thus there are greater reductions (both proportional and absolute) at roadside sites compared with urban background sites. This clearly reflects the direct impact of local traffic reduction at sites where traffic is the dominant contribution to NO_x and NO_2. Figure 6 highlights this general trend, but also shows considerable between-site variability at similar ambient concentrations.

Analyses of diurnal cycles in NO_2 concentrations show reductions are dominated by decreases during the daytime and the loss of typical morning and afternoon rush-hour maxima in concentrations, consistent with the change to a weekend driving pattern reported above. The major influence of changes in traffic on NO_x and NO_2 concentrations is supported by the decline in roadside increments of NO_x and NO_2 in London during the pre-lockdown period (12-23 March 2020) which continued to reduce after the full lockdown. Weekday/weekend ratios of NO_x and NO_2 suggest traffic activity reduced at weekends more than during the week.

There is, however, considerable variation in the magnitude of reduction across sites, including some sites where NO_x and NO_2 is estimated to have increased. The reasons for this are not clear but may be caused by site-specific local traffic or other source changes. Increases in NO_x and NO_2 at some rural sites may reflect the regional scale pollution conditions encountered post-lockdown. It is also important to note that NO_x and NO_2 concentrations are much lower at rural sites so absolute concentration changes are often small.

Changes to ozone concentrations

Measurements show significant increases in O_3 at roadside sites (mean increases averaged over the period of lockdown considered of the order of 15-20%), and smaller increases at urban background sites (mean increases of ~5%) although again with significant between-site variability. The increase in urban O_3 is a consequence of the reductions in primary NO emissions decreasing the extent of chemical loss of O_3 through reaction with NO. In urban areas there has likely been a redistribution in the constituents of O_x (= NO_2 + O_3) from NO_2 to O_3.

In urban NO_x regimes, reductions in NO emissions not only redistribute O_x into O_3, but also lead to the reduction of loss of RO_x *via* OH + NO_2. Reducing NO_x concentrations in high NO_x urban conditions can also lead to increased concentrations of peroxy radicals (HO_2 and RO_2) with the potential to increase photochemical production of O_3. The relative contributions from these two effects is not yet quantified, nor the impact of possible changes that may have occurred to VOC emissions on O_3. These uncertainties could be addressed with further chemical modelling.

Applying a deweathering approach to ozone observations in the UK provides some indication of the scale of change of ozone compared to business as usual, accepting that there are transboundary contributions to ozone that are not always accounted for by local meteorological normalisation. Figure 7 shows the raw ozone data and deweathered data for urban, rural, roadside and industrial sites across the UK, with largest relative increases in ozone seen roadside for reasons described earlier. There is some indication however of higher rural ozone than under a business as usual scenario.

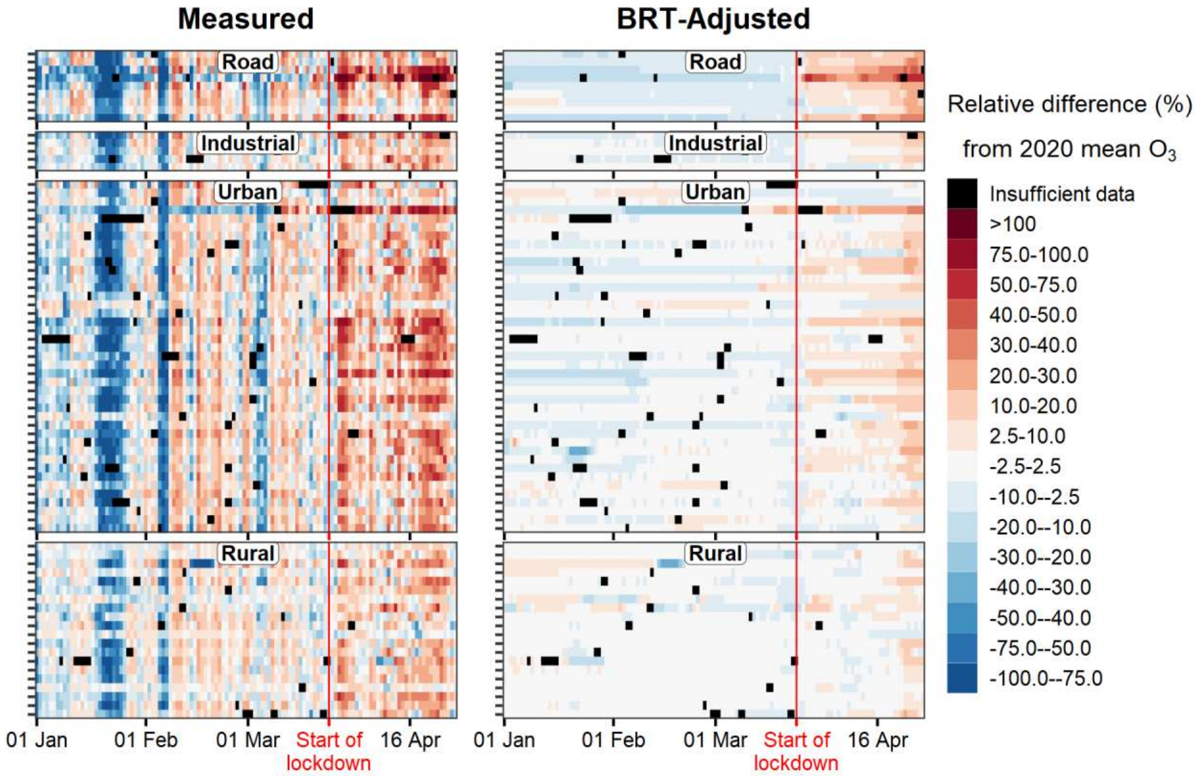

Figure 7. Relative change in daily-mean ozone between 1st January and 29th April 2020 at 75 monitoring sites in the AURN and other UK Networks. First column shows raw measured data. Second column shows equivalent values with the effects of meteorological variability removed using Boosted Regression Trees through the deweather package in openair. Each row of pixels represents a single site, with individual sites denoted by tick marks on the *y* axis.

Reproduced from: Marner, B. *et al.*, *Air Quality Consultants*, 2020 "Response to AQEG Request for Rapid Evidence on COVID-19 & UK Air Quality."

Changes to $PM_{2.5}$ concentrations

Quantifying the changes in $PM_{2.5}$ concentrations during the lockdown period is a more complex problem than for NO_2, because it is affected by both local sources and the transport of pollution from wider regions, and indeed from well beyond UK borders. Southern and Eastern parts of the UK can be particularly susceptible to higher $PM_{2.5}$ during anticyclonic conditions, drawing air already elevated in $PM_{2.5}$ from mainland Europe, and this has been a dominant type of weather pattern over much of April and May. Statistical methods for removing the effects of meteorology are not as effective as they are for NO_2, since variability of $PM_{2.5}$ is influenced by not only local emissions and meteorology, but also longer-range air mass trajectory and origins.

The changes in NO_2 emissions have been sufficiently large that over the early lockdown period (17th Mar to 29th Apr 2020) concentrations are clearly lower than when compared to the pre-lockdown period from 1st Jan to 16th March 2020. This is shown as a percentage difference on Figure 8 for nine UK cities. Figure 8 (top) shows that $PM_{2.5}$ during the lockdown period was in most places higher than was seen before lockdown in 2020. This was a consequence of cleaner than normal periods with high winds and rain in February 2020 (pre-lockdown) and then greater anticyclonic transboundary transport of $PM_{2.5}$ from mainland Europe during the lockdown period. Figure 8 (bottom) shows the lockdown period in 2020 and compares this against the same calendar period for the average for 2015-2019. When the lockdown period in 2020 is compared to previous years it is clear that NO_2 is lower everywhere, but the picture for $PM_{2.5}$ is mixed compared to previous years.

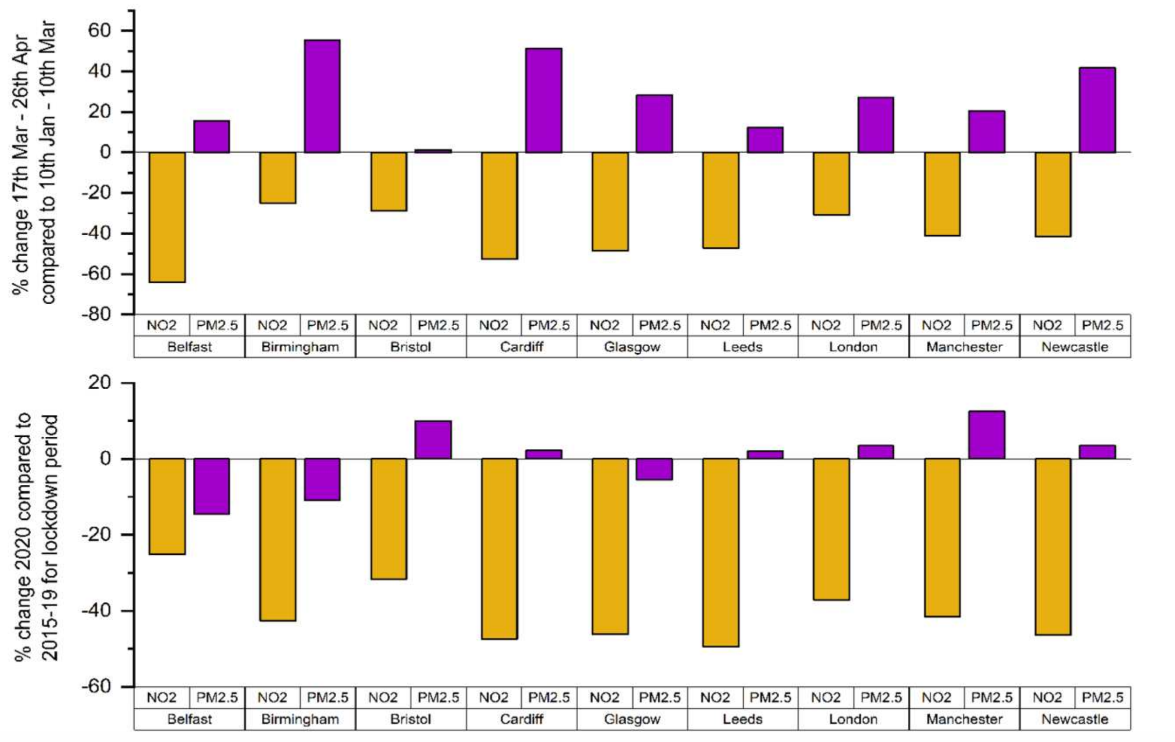

Figure 8: Top: Percentage change in NO_2 (orange) and $PM_{2.5}$ (purple) for nine UK cities for pre- and post-lockdown period (1st Jan -16th March and 17th Mar – 29th Apr 2020). **Bottom**: the lockdown period in 2020 and compared to the same calendar period averaged over 2015-2019.

Reproduced from: Lee, J.D., Drysdale, W., Wilde, S., *National Centre for Atmospheric Science / University of York*, 2020. “Air Quality in the UK during the COVID-19 pandemic – evidence from national monitoring stations.”

One evidence submission used observational data from the AURN over the period 2018-2020 and compared this with the GEOS Composition Forecasting (GEOS-CF) system, using the GEOS-Chem gas and aerosol chemical scheme. Models such as this provide a means to account also for the pollution history of an air mass as it is transported. The model was run for the period 2018-2020 compared against surface observations and any bias removed using machine learning methods. Once the bias was removed from the whole dataset any remaining bias seen for given periods was then attributed to the reduction in emissions of $PM_{2.5}$ (and precursors) right across the model domain. The model showed a bias that over-predicted $PM_{2.5}$ during the lockdown period. The Keller and Evans submission does not attempt to quantify how much emissions have reduced, or where those reductions occurred, but does provide an 'air mass corrected' estimate of how COVID-19 period concentrations have differed compared to what would have been expected under business-as-usual.

Figure 9 shows the residuals from the model minus measurement, once machine learning trained, for 2018/2019 for a representative sample of AURN monitoring locations. It also shows the residual of model - measurement specifically for the early lockdown period. Some significant residuals emerge in this period, indicative of reductions in $PM_{2.5}$ concentrations relative to those that would be expected under business-as-usual, of the order 2 - 5 $\mu g\ m^{-3}$. Some fraction of this reduction will undoubtedly have been caused by wider reductions in continental European emissions, also affected by national lockdowns.

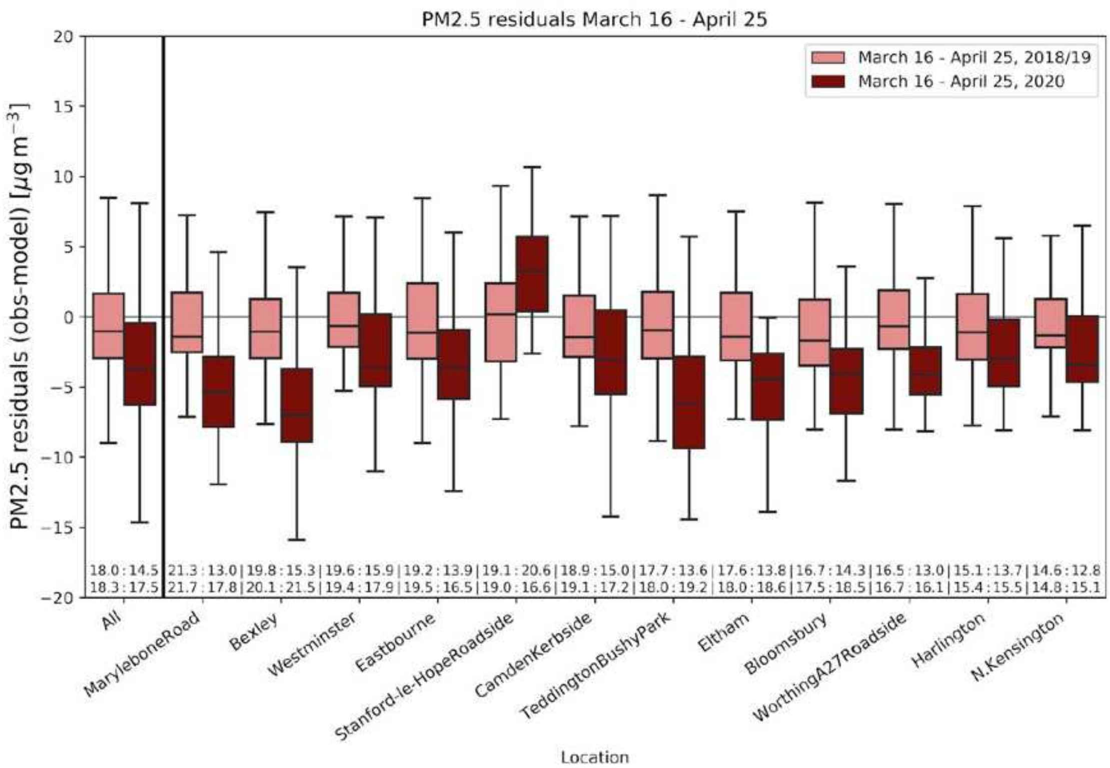

Figure 9. Distribution of AURN observations minus model residuals of $PM_{2.5}$ from 16th March to 25th April for a number of sites in the Southern England. Pink boxplots show the difference between observed and predicted $PM_{2.5}$, or the period 16th March to 25th April for 2018 and 2019, once the model had been trained and bias corrected. The dark red boxplots show the difference between observed and model-predicted $PM_{2.5}$ for the COVID-19 restriction period (16th March to 25th April 2020) once the model had been trained and bias correct. 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 left-most 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 $PM_{2.5}$ pollution concentrations, high to low.

Reproduced from Keller C, and Evans, M.J., *University of York and NASA Goddard Space Flight Center*, 2020. “Understanding the impact of COVID-19 restrictions on air quality pollutants with a mixed atmospheric chemistry transport model / machine learning approach.”

Particle composition changes

For the period of lockdown included in these analyses, absolute concentrations of $PM_{2.5}$ have increased across the UK. However, as previously described, these increases have been influenced by trans-boundary pollution transport, favouring conditions with higher $PM_{2.5}$ from mainland Europe. A question to be addressed in the future through modelling of the kind shown in Figure 9 is the extent to which the $PM_{2.5}$ episodes experienced would have been worse without the lockdown and which sources contribute most to the changes experienced. The extent to which the continuation of (agricultural) NH_3 emissions in a lower NO_x emission climate impacts on the NH_4NO_3 component of $PM_{2.5}$, is an issue that will require further investigation. The sources responsible for a significant contribution from organic aerosol to $PM_{2.5}$ concentrations are also not known. From the aerosol composition measurements provided in the evidence, it is not currently possible to quantify a clear change that could be associated with COVID-19 actions.

A detectable change in the proportion of black carbon (BC) arising from wood burning is reported for Manchester and changes to patterns of wood burning emissions are also noted in London. This is consistent with a submission from the Chartered Institute of Environmental Health (CIEH, 2020) whose members noted rising complaints to some local authorities about domestic burning and bonfires. This source of potential pollution change warrants further investigation.

Looking ahead

The many analyses submitted have provided a clear first indication of the scale of changes in pollutant concentrations, especially for NO_x, NO_2 and O_3. It is not surprising however, that there are inherent limitations in this early analysis that could be resolved in the longer term. These limitations include the use of data that are not fully ratified, the variation in the sites considered, the periods over which comparisons are made and the techniques used to analyse the data. All these factors can be addressed in follow-up work to refine the analyses. More challenging is the quantification of the effects of long-range transport and how it has affected the concentrations of species such as PM_{10}, $PM_{2.5}$ (including the speciated composition) and non-urban O_3. It is difficult to observe a clear change in the concentration of these species from the analysis of atmospheric composition data alone. Conversely, even though air quality modelling suggests likely reductions in $PM_{2.5}$ e.g. due to ammonium nitrate, it is difficult to verify the predicted changes are reasonable.

Some of the analyses considered changes in exceedances of air quality limits either based on the year so far and how exceedances compare with the same period in previous years, or forward projections to estimate the outcome on the whole of 2020. The latter approaches could provide a useful indication of how to use measurements made during a short-term intervention to estimate effects on compliance with air quality standards. Another promising approach was to link the estimated changes in the concentration of NO_2 with traffic data to establish a link between traffic flow and fleet composition changes to corresponding change in the concentration of different pollutants. Such approaches may be helpful in quantifying the level of local change needed for a particular desired outcome in terms of local air quality, particularly for NO_2. Similarly, further, focused analysis of the data could provide estimates of changes in emissions for some pollutants, rather than concentrations, which has been the focus of this section.

Data sources used in Table 2.

Alfarra, R., Allan, J., Alzahrani, K., Alzahrani, S., Coe, H., Jay, C., Marsden, N., McFiggans, G., Reyes-Villegas, E., Ricketts, H., Taylor, J., Topping, D., Wu, T. *University of Manchester*, 2020. “Contribution to the AQEG/Defra Call for Evidence.”

Fonseca et al., Breathe London, 2020. “New Breathe London data: Covid-19 confinement measures reduce London air pollution.”

Carruthers, D. and Jones, RL., *CERC and University of Cambridge*, 2020. Breathe London submission to the Defra / AQEG Covid-19 call.”

Dacre, H., and Mortimer, H., *University of Reading / STFC*, 2020. “How have NO_2, NO_x and O_3 changed since the COVID outbreak?”

Environmental Research Group, *King’s College London*, 2020. “The effect of covid-19 lockdown measures on air quality in London in 2020”.

Finch, D.P. and Palmer, P.I., *University of Edinburgh*, 2020. “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.”

Lee, J.D., Drysdale, W., Wilde, S., *National Centre for Atmospheric Science / University of York*, 2020. “Air Quality in the UK during the COVID-19 pandemic – evidence from national monitoring stations.”

Lewis, P.D., Seller, V., Price, T. and Eskandari, H., *Swansea University*, 2020. “Estimation of ambient NO_2 and $PM_{2.5}$ concentration change in Wales during COVID-19 outbreak.”

Marner, B., *Air Quality Consultants*, 2020. “Response to AQEG Request for Rapid Evidence on COVID-19 & UK Air Quality.”

Murrells, T. *et al.*, *Ricardo Energy and Environment,* 2020. “Response from Ricardo to Key Questions”

Table 2: Summary of changes in measured NO_x, NO_2 and O_3 post-lockdown, divided into analyses that applied some form of meteorological normalisation and analyses that used measured data directly. This is a selection from the evidence submitted.

Ref	Methodology and scope	Comparison time periods	Pollutant changes	Comments
With meteorological normalisation				
Marner, 2020	Boosted-regression tree using *deweather* package to predict conditions under average meteorology. All UK AURN sites + other online networks	BRT-adjusted mean for 24th Mar to 9th Apr *versus* 1st Jan to 14th Mar 2020	**NO_x** Road mean (n = 122): −30% UB mean (n = 57): −16% Rural mean (n = 14): +23% **NO_2** Road mean (n = 123): −31% UB mean (n = 56): −14% Rural mean (n = 13): +15% **O_3** Road mean (n = 10): +17% UB mean (n = 40): +4% Rural mean (n = 20): -1%	N.B. Absolute NO_x/NO_2 is low at rural sites. For roadside sites, % reduction in NO_x/NO_2 increases with absolute NO_x/NO_2, and % increase in O_3 decreases with absolute O_3
Dacre, 2020	Multiple linear regression with met variables to predict counterfactual. 110 AURN sites including some rural (51 for O_3)	Measured *versus* model-predicted for 16th Mar to 25th Apr 2020.	**NO_x** Mean all sites (n = 110): −36% (sd 14%) **NO_2** Mean all sites (n = 110): −23% (sd 13%) **O_3** Mean all sites (n = 51): +20% (sd 9%)	Largest % decreases in NO_x/NO_2 at road sites, smallest – including some increases – at rural sites. Largest increases in O_3 at road sites.
Ricardo 2020	*deweather* package to predict counterfactual. 29 AURN sites, mixture of road and UB	Measured *versus* model-predicted for 16th Mar to 18th Apr 2020	**NO_x** Road mean (n = 17): −48% UB mean (n = 12): −31% **NO_2** Road mean (n = 17): −37% UB mean (n = 12): −25%	
Lewis 2020	Random Forest to predict conditions under average meteorology. All non-rural AURN sites in Wales.	RF-adjusted median for 24th Mar to 29th Apr 2020 *versus* 1st Jan to 23rd Mar 2020.	**NO_2** All non-rural sites in Wales (n = 13) mean: −8.5 (sd 5.1) $\mu g\ m^{-3}$	Reductions as a % not available

On direct measurements				
Lee 2020	AURN sites across the UK.	16th Mar to 27th Apr 2020 *versus* same period averaged for 2015-2019	**NO_2** Road mean (n = 97): −45% UB mean (n = ~50): −38%	Reductions may be over-estimated as comparison is against mean of previous 5 years and NO_2 concentrations likely higher in the past.
Alfarra 2020	AURN sites across the UK.	April 2020 *versus* same period averaged for 2015-2019	**NO_x** Road (n ~100): "most sites in range −20% to −80%" UB (n ~50): "most sites "range −10% to −60%" **O_3** "little systematic change"	No summary statistics provided. Considerable site-to-site variability emphasised, including some sites showing NO_x increases
ERG 2020	London AURN/LAQN sites.	24th Mar to 22nd Apr *versus* 1st Jan to 12th Mar 2020	**NO_x** Road mean (n = 53): −44% (range +5% to –75%) **NO_2** Road mean (n = 53): −21.5% (range +32% to –55%) UB mean (n = 27): −14.5% (range +13% to –38%)	Reductions in NO_2 were smaller at non-central London roadside sites and at urban background sites.
Fonseca 2020	London Breathe project AQMesh sensors.	17th Mar to 13th Apr *versus* 1st Mar to 16th Mar 2020	**NO_2** Range (n = 71): −9 to −17% Central London only: −20 to −24%	Greatest reductions during day.
Carruthers 2020	London Breathe project. AQMesh sensors.	17th Mar to 20th Apr *versus* 1st Mar to 16th Mar 2020	**NO_x** Mean (n = ~100): −23% ULEZ only: −29% **NO_2** Mean (n = ~100): −15% ULEZ only: −20%	Greatest reductions during day. Greater reductions in ULEZ
Finch 2020	AURN sites across the UK.	23rd Mar to 25th Apr 2020 *versus* same period averaged for 2015-2019	**NO_2** Road & UB mean (n = 125): −13 μg m^{-3} Rural mean (n = 22): −2 μg m^{-3} **O_3** Road & UB mean (n = ~40): +12 μg m^{-3} Rural mean (n = ~20): +1 μg m^{-3}	Reductions in NO_2 may be over-estimated as comparison is against mean of previous 5 years and NO_2 concentrations likely higher in the past.

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

As outdoor air quality has improved, more attention is being paid to the indoor environment as a source of exposure to air pollutants. An increase in exposure could hypothetically be significant for various vulnerable groups such as the elderly and the sick, with many in these groups shielding due to COVID-19. Further, many of those groups could be more susceptible to poor air quality (e.g. COPD sufferers). Compared to outdoor and workplace air quality, there is no systematic monitoring of domestic air quality or emissions, so the effects of COVID-19 and associated lockdown are largely theoretical, however experimental work is currently taking place in Oxford to offer insight on this question.

We would expect the following to be factors in indoor air quality:

1. **Individuals spending more time indoors in residential settings.** Unlike workplace settings, domestic air quality is not as carefully managed and engineered and regulated, so in many cases it can be inferior (Shulman, *BlockDox Ltd*, 2020). Indoor air quality is dependent on many factors including the design, management and operation of the buildings themselves and in particular the level of ventilation (passive or mechanical), but also the sources of pollution that may be present both inside and outside a building. Ventilation rates in UK homes are often influenced by weather conditions and recent warmer weather may have increased ventilation and air exchange.
2. **More activities being performed at home that are detrimental to indoor air quality.** In spending more time at home, it is likely that more activities will take place that are known to impact indoor air quality, particular examples being smoking, cooking, cleaning, DIY activities, and solid fuel burning (CEIH, 2020; Heydon and Chakraborty, *University of Nottingham*, 2020). There are also possible sources of VOCs from hobby and craft activities that use glues and solvents. Cooking in particular is known to be a highly significant source of indoor PM (Environmental Research Group, *King's College London*, 2020) and there is anecdotal evidence that more domestic wood burning taking place at the start of the lockdown period (Murrells, T. et al., *Ricardo Energy and Environment,* 2020). There is also the possibility of increased exposure to emissions from faulty appliances (Gulliver, J., Hansell, A. and Jephcote, C., *University of Leicester*, 2020) and general occupancy will also increase the amount of household dust being shed and re-suspended.
3. **A change in indoor air chemistry resulting from the observed increase in outdoor ozone concentrations.** There are numerous indoor sources of volatile organic compounds (VOCs) (Lee, *et al. National Centre for*

Atmospheric Science / University of York, 2020; CIEH, 2020.) In addition to directly affecting indoor air quality, some react with ozone penetrating from outdoors and produce potentially harmful by-products. It follows that the observed increases in ambient ozone following lockdown in response to COVID-19, will also increase the concentrations of these reaction products indoors. Modelling of the indoor environment has predicted possible increases in indoor formaldehyde concentrations of around 30% and an enhancement of secondary PM resulting from cleaning activities, owing to the enhancement of indoor chemistry by higher outdoor ozone concentrations (Carslaw, N. *University of York*, 2020).

4. **Reduced activity in workplace settings.** Monitoring observations submitted by Shulman, N. (*BlockDox Ltd*, 2020) show reduced CO_2 and VOC concentrations associated with reduced building occupancy, indicating an improvement in air quality. It follows that the members of a reduced workforce would experience better air quality than business-as-usual. However, this may be offset at least in part for some workplaces that have remained open (e.g. supermarkets) where VOCs associated with increased cleaning activities have produced additional formaldehyde and particles indoors.

While an increase in exposure to indoor air pollutants is expected, overall exposure may be offset by a reduction in exposure associated with commuting and a reduction in outdoor NO_2. The Kings College London (see next section) exposure model that accounts for both indoor and outdoor pollution predicts a net increase in $PM_{2.5}$ exposure for most individuals (with the exception of tube users) as a result of COVID-19, if it is assumed that an additional domestic cooking activity is performed each day. Without the extra cooking activity, it predicts a slight decrease[1]. The model also predicts a net decrease in NO_2 exposure both with and without additional cooking activities.

It should also be noted that relatively little is known about the chemical composition of indoor particles and how this affects their toxicity, compared to outdoors. Indoor particles will be a mixture of outdoor particles, primary indoor particles (e.g. cooking fumes, household dust, and smoke from candles, stoves and cigarettes) and secondary ultrafine particles formed through cleaning and VOC degradation.

Measures to mitigate indoor air pollution exposure could include encouraging people to spend more time outdoors (including gardens and balconies), regular vacuuming of carpets, not smoking indoors, and ensuring that homes are adequately ventilated (e.g. leaving windows open), particularly whilst cooking and cleaning (Okam, *Cheshire East Council*, 2020). Regarding cooking, kitchen filtration or extraction devices will help to mitigate exposure, as long as they expel the emissions to the outdoors rather than recirculate them within the kitchen. Proper maintenance of

[1] Note that the outdoor $PM_{2.5}$ concentrations in the exposure model were averages over a year of each hour of the day for 2019, not the actual outdoor $PM_{2.5}$ concentrations during the 2020 lockdown. These could be modelled subsequently.

cooker extraction devices is essential (cleaned/filters replaced) if they are to be effective.

More detailed measures to reduce exposure to indoor air pollution are provided in the recent RCPCH report in a manner accessible to the general public (https://www.rcpch.ac.uk/resources/inside-story-health-effects-indoor-air-quality-children-young-people).

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

The pandemic has altered exposures to air pollution both outdoors and indoors. Both of these exposure routes have been affected by emission changes occurring in the outdoor atmosphere during the lockdown, behaviour changes affecting exposures to those outdoor pollutants and the way COVID-19 altered behaviours may affect indoor pollutants and exposures.

Particulate matter ($PM_{2.5}$) exposure

Concentrations of total $PM_{2.5}$ mass have increased in southern England at least during the first part of the lockdown period compared with the previous months of 2020 due to the persistence of anticyclonic weather, leading to a higher incidence of easterly winds bringing $PM_{2.5}$ and its precursors from the rest of Europe into the UK (see Question 2 earlier). Average concentrations during the lockdown period (post 23 March to 25 April 2020) in London have increased by ~74% compared with the previous months of 2020. Increases in $PM_{2.5}$ have been observed across the UK to varying degrees with the largest increases in the south-east of the UK.

There has been very little difference between changes seen in total $PM_{2.5}$ at roadsides and urban background locations so that people living near busy roads, or in other buildings near busy roads like hospitals, will have experienced little difference in exposures compared with people living elsewhere. Traffic components of $PM_{2.5}$ (such as black/elemental carbon) emissions, and non-exhaust emissions from vehicles however will have reduced in line with reductions in traffic activity. This reduction in activity has amounted to about 50% on average across London and up to about 70% centrally compared with the same period in 2019 and these reductions were greater at roadside sites.

Since outdoor concentrations of total $PM_{2.5}$ have increased in April it is likely that in the early stages of the lockdown outdoor exposures will have increased too as people are likely to have spent more time outdoors exercising during that phase of the lockdown, and where possible, spending time in gardens at home. This change may have been offset to some degree however as many people would no longer have commuted to larger and more polluted cities. Exposures to total $PM_{2.5}$ would have been spatially reasonably uniform so the absence of commuting may not have changed exposures significantly, but exposures to the traffic components of $PM_{2.5}$ (and PM_{10}) are likely to have reduced.

The Chartered Institute of Environmental Health (CIEH, 2020) reported responses from 14 local authorities that showed a large increase in complaints about bonfires during the lockdown; some authorities reported 2 to 4-fold increases compared with the same period last year. Being mindful of vulnerable people isolating at home, several local authorities asked residents to refrain from lighting bonfires and solid fuel burning in indoor fireplaces and stoves (https://www.theguardian.com/environment/2020/apr/06/pollutionwatch-lockdown-boosting-air-quality-we-can-do-more-coronavirus).

Measurements from the London NERC air quality research supersite did not show an obvious increase in daytime wood-burning PM, as might be expected from bonfires. Evening wood burning was detected, with a diurnal pattern that showed wood burning was beginning later in the evening compared with winter months. Measurements from the Manchester NERC supersite showed an increase in the proportion of black carbon from wood-burning, compared with the pre-lockdown (Alfarra, *et al. University of Manchester*, 2020). Similar to the London data, the wood-burning measured in Manchester was prevalent in the evening, though a wild fire (at Rakes Moss) had an important effect on local air pollution in south Manchester. Further analysis of data from Defra's black smoke network may provide further insight into bonfires and home wood-burning during the lockdown.

The good weather over much of the early lockdown period and also the second half of May will have probably increased the exchange between outdoor and indoor air so that ingress of the elevated outdoor $PM_{2.5}$ will likely have increased. Behavioural changes during the lockdown will have altered the pattern of 'normal' indoor $PM_{2.5}$ concentrations, largely due to what are likely to have been an increased frequency of cooking in the home. An attempt has been made to quantify this (Environmental Research Group, *Kings College London*, 2020).

The London Hybrid Exposure Model was run to estimate total personal exposure (i.e. to indoor and outdoor pollution) for a 2019 "before lockdown" case, a "lockdown" case with a reduction in traffic activity (-53%) and aviation activity by (-73%), along with behavioural changes assuming children and tube users stayed at home (no change for professional drivers and hospital staff), and a "lockdown with 1 hour extra cooking" case. The results are shown below as frequency distributions of average concentrations over the respective time periods shown in Figure 7 and the mean, min and max exposure in Table 3 for each of the four population sub-groups. No account was taken of the actual outdoor concentrations during March onwards and the outdoor $PM_{2.5}$ (and NO_2 – see below) component was taken to be the annual average from 2019 incorporating ULEZ measures, so that the reductions in exposure were due to the combination of traffic emission reductions and the changes in travel behaviour. The actual outdoor $PM_{2.5}$ concentrations would need to be added to these exposures. The results are shown below as frequency distributions of average concentrations over the respective time periods shown in Figure 10.

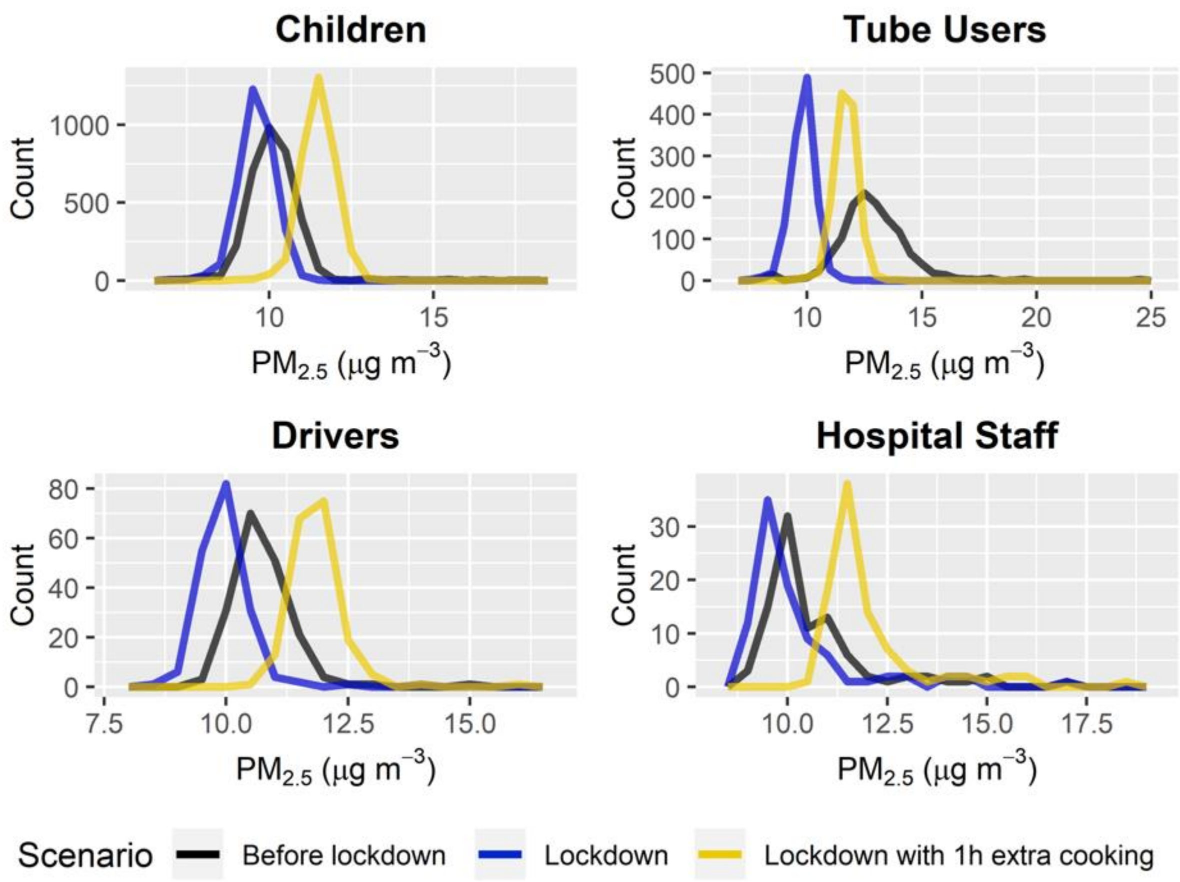

Figure 10. Histograms comparing modelled human exposure to $PM_{2.5}$ before and after the lockdown in different subgroups, with the "1h extra cooking" scenario also included.
It should be stressed here that there is still a lack of data on the patterns of indoor behaviours, and of cooking in particular, affecting indoor emissions. Equally uncertain is the toxicity of the different sources of indoor PM.

Reproduced from Williams, M. on behalf of Environmental Research Group, *King's College London*, 2020. "The effect of COVID-19 lockdown measures on air quality in London in 2020".

It should be stressed here that there is still a lack of data on how the patterns of indoor behaviours, and of cooking in particular, affecting indoor emissions. Equally uncertain is the toxicity of the different sources of indoor PM.

Chemical analysis of the sub-components of $PM_{2.5}$ show that the traffic components have reduced but the concentrations of secondary aerosol composed mainly of ammonium nitrate and secondary organic aerosol were still high (although possibly not as high as they might have been in the absence of lockdown in northern Europe, or business as usual urban NO_x). Whether changes in particle composition, and indeed particle size (for example the fraction of ultrafine particles to $PM_{2.5}$) have changed overall toxicity remains an open question.

Nitrogen Dioxide (NO_2) exposure

NO_2 concentrations reduced significantly at busy roadside sites in London 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 locations. The reduction in average NO_2 at North Kensington was 22% and the mean reduction across all urban background sites in London was 14%. Changes in NO_2 were variable at other roadside sites and urban background sites across the UK, these are shown graphically for locations across the UK in Figure 5.

Outdoor exposures to NO_2 will have reduced virtually everywhere in the UK and the outdoor contribution to indoor concentrations will similarly have reduced. Changes in travel patterns will also have reduced exposures to NO_2 insofar as exposures to traffic emissions outdoors are replaced by attenuated indoor concentrations, or where staying at home involves staying in areas less polluted locations than the normal working environment.

Indoors, the most important source of NO_2 across the UK population is likely to be gas cooking. Indeed, the current WHO guideline and EU Limit Value was originally based on respiratory effects in homes with gas cookers. Taking into account the additional use of gas cookers during lockdown, modelled total exposures of London's population are shown in Figure 11 below based on the same assumptions as discussed for $PM_{2.5}$ above. A summary table of the modelled $PM_{2.5}$ and NO_2 concentrations is given in Table 3.

Table 3. Mean (min-max) estimated exposure to $PM_{2.5}$ and NO_2 in different scenarios. Data from Environmental Research Group, *King's College London*, 2020. "The effect of covid-19 lockdown measures on air quality in London in 2020".

Scenarios	Children (n = 3319)	Tube users (n = 1218)	Drivers (n = 183)	Hospital staff (n = 92)
	$PM_{2.5}$ (μg 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 (μg 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)		

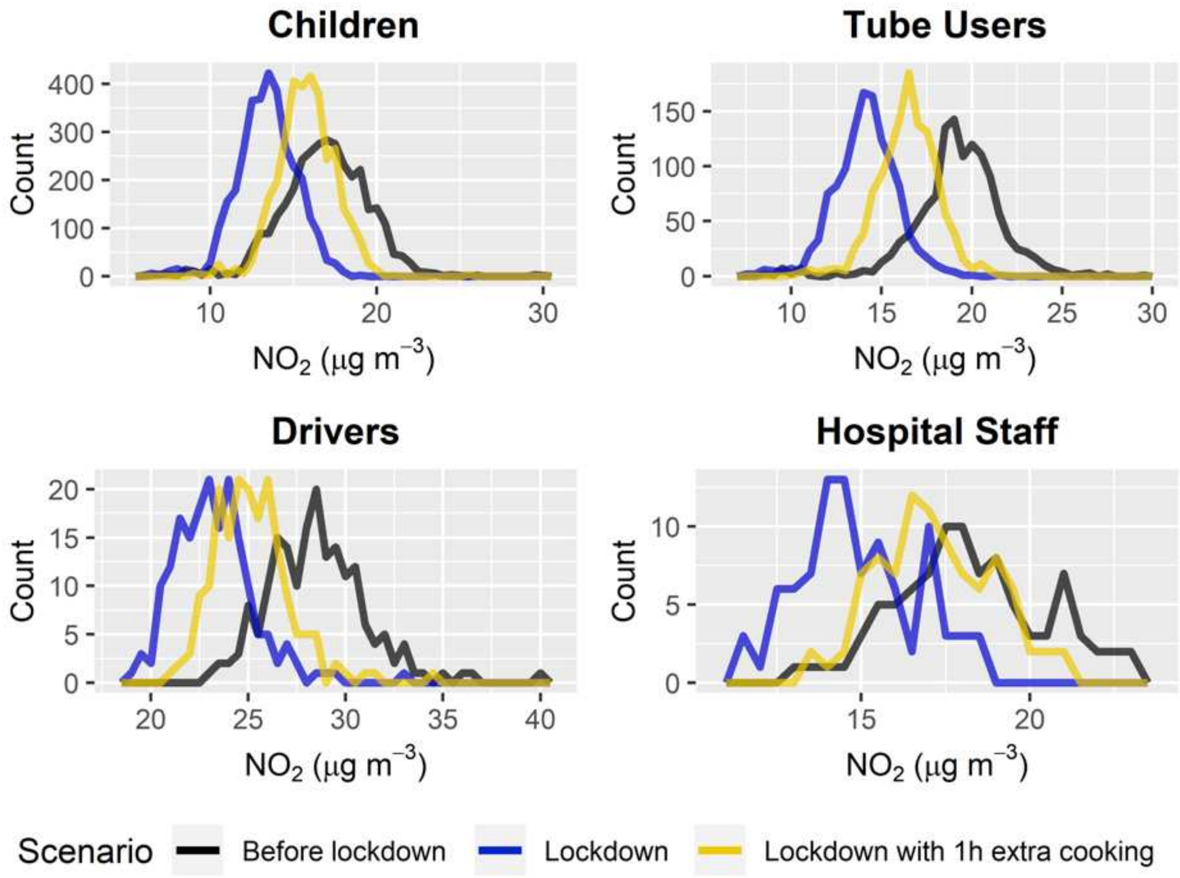

Figure 11. Histograms comparing modelled human exposure to NO_2 before and after the lockdown in different subgroups, with the "1h extra cooking" scenario also included.

Reproduced from Williams, M. on behalf of Environmental Research Group, *King's College London*, 2020. "The effect of covid-19 lockdown measures on air quality in London in 2020".

Ozone exposure and indoor impacts

As with $PM_{2.5}$, the post lockdown period saw several periods of elevated ozone concentrations with contributions from transboundary transport producing hourly concentrations up to 129 µg m^{-3} (~65 ppb). This in itself, coupled with the fact that people will probably have opened windows more than usual, would be enough to increase short term population exposures to ozone. However, increased outdoor ozone concentrations can also produce increased in indoor concentrations, through building ingress, and in turn driven chemical reactions between ozone and VOCs the latter from sources such as indoor cleaning and DIY products.

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. There is likely to have been more cooking than usual and cooking and cleaning both produce pollutants indoors that are well known to impact health, e.g. formaldehyde HCHO (classified by the International Agency for Research on Cancer (IARC) as carcinogenic to humans), as well as particulate matter. So potentially, there could have been increased concentrations of these secondary pollutants indoors and longer exposures to them, although there is no observational evidence to confirm this.

A modelling study was carried out (Carslaw, *University of York*, 2020) assuming that outdoor ozone concentrations had increased by 30%, whilst outdoor NO_x concentrations had decreased by 35%. These outdoor concentrations were then used to drive indoor chemistry following air exchange at a typical residential rate of 1 air change per hour. Indoor ozone concentrations were predicted to increase by ~50% under these conditions. The model was also used to simulate cleaning with a 10-minute use of a limonene-based product at 16:00 h, such that the indoor limonene concentration reached 300 ppb. The ozone-VOC chemistry produced increased concentrations of formaldehyde and PM as shown in Figure 12.

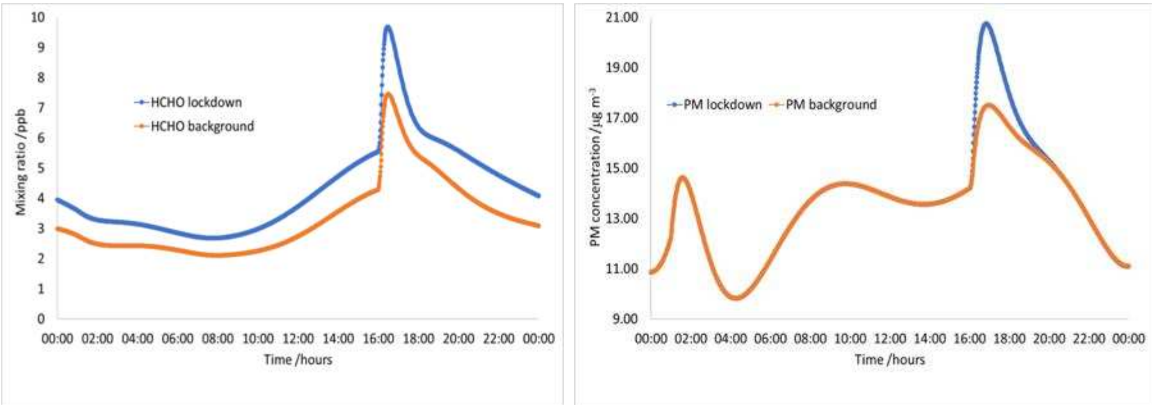

Figure 12. Modelled indoor formaldehyde (left) PM and (right) and for a typical residence pre- and during lockdown, including illustrative 10-minute emissions of limonene (a common ingredient in cleaning products) at 1600.

Reproduced from Carslaw, N. *University of York*, 2020. “Short summary for AQEG: What changes do you anticipate in indoor air quality as a result of the Covid-19 pandemic?”

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

General Issues

This question has been interpreted in understandably different ways in the evidence submitted, including which months are considered as summer and for how long the altered emissions associated with COVID-19 will continue. This summary given here is focussed on June to August 2020, as the next three months.

A range of atmospheric chemistry models have been run for the current period March-April 2020 as well as year-long runs using meteorological fields from previous years. The spatial scale and resolution (6 km to 200 km) of the models vary from local and regional to global. For PM and NO_2, the peaks in concentrations occur close to emission sources and fine spatial resolution is very important, while for O_3, the atmospheric background and therefore regional and transboundary effects contribute significantly to ozone.

Model results aiming to simulate summer air pollution are naturally highly dependent on the emission scenarios evaluated – some runs attempt to reproduce the estimated emission changes, whilst others are largely sensitivity studies testing a range of changes, and in some cases are limited just to changes in NO_x emissions. Regional models, whilst having higher resolution, have to deal with boundary conditions, which for some model runs are unchanged so may not capture the import of pollution from outside the model domain where large emission reductions have also occurred in recent months. The overall change in ozone over a season is crucially dependant on the weather at that time. As such model studies can only examine case study events, or assess the scale of change that would have occurred for a previous year.

Likely impacts on summertime air quality

Noting the issues above, the main messages from the submissions are:

Modelling of summertime air pollution should account for the substantial reductions in NO_x concentrations that have been reported over the last few months for countries and regions with large reductions in transport and industrial emission sources. Modelled scenarios using reductions in NO_2 concentrations at the surface in the range 20 to 50% would match observations. For the UK, the observed reductions in NO_2 in large urban centres range from 25% to 35% (cf. Chapter 2 and Table 2) and are likely to persist while emissions remain supressed.

- The key to estimating the changes in NO_x over the next three months, and by extension gaining more clarity over summertime pollution, is the rate at which emissions increase over the period in response to relaxation of lock-down.

- Percentage concentration changes are very approximately linear with the percentage emission changes (there has been a slight change in the partitioning of NO and NO_2 as NO_x emissions have declined).

Changes in $PM_{2.5}$ are less clear, with model results varying from very little change to decreases of around 20%.

- Reductions in primary PM emissions cause a direct reduction in measured PM of a magnitude less than or equal to the reduction in the emissions.
- Secondary nitrate aerosol is not limited by availability of NO_x in most parts of the UK but rather by ammonia, and there is little evidence of any reductions of ammonia during COVID-19. Indeed, PM events since the lock-down have continued and are dominated by NH_4NO_3.
- Transboundary transport of PM and precursors from mainland European sources, and the associated meteorology, will continue to play an important role in episodes of elevated concentrations of PM.
- Meteorology will be an important factor in the pollution climate of the UK during summer 2020. Anticyclonic conditions in the cooler springtime favour PM events, especially of NH_4NO_3 and these have been a dominant feature of UK weather over April and May 2020.
- Changes in emissions of PM that are linked to household activities, for example outdoor fires and BBQs are not evaluated in any of the evidence submissions, but could plausibly increase should a substantial fraction of the UK workforce remain at home over the summer.

The direction of change in O_3 differs between models, model resolution, geographical location and between different types of episodes used to test ozone changes. However, the evidence, both from measurements and models is for smaller predicted changes relative to those seen in ambient NO_x.

- On average across the UK the likely overall change in ozone relative to business as usual is small, (<10%)
- Ozone is predicted to increase in central areas of the UK and south east England, in urban areas more widely, near airports and major roads, as a result of repartitioning of O_X into O_3 and increased photochemical production of O_3 due to reduced NO emissions (i.e. reduced “urban decrement”).
- This increase in ozone may partially be compensated for by decreased VOCs emissions, reducing the production rate of O_3, although there is very limited observational or activity evidence that VOC emissions have decreased significantly during lockdown.
- Peak ozone during summertime anticyclonic episodes could potentially see higher ozone concentrations in some locations, although the scale of this increase, and its locations are highly dependent on the prevailing meteorology.

Wider regional scale reductions in O_3 production seem likely if reduced emissions of NO_x and VOC continue broadly across Europe and this may lead to decreases in O_3

in rural areas, with benefits to crop production and ecosystems. Again, the expected changes are small (<10%).

- Meteorology, as is the case for PM, will be an important factor in the frequency and magnitude of O_3 episodes during summer 2020. Anticyclonic conditions favour ozone production and episodes and have been a dominant feature of UK weather over April and May 2020.

Two example model predictions of ozone under changed COVID-19 lockdown emissions are shown in Figures 13 and 14. These do not attempt to estimate the seasonal change in ozone, or how overall annual statistics for ozone might change in 2020. Each model contrasts a reference meteorological case study event from the past in summer using firstly existing emissions and then with best estimates of possible reduced emissions of NO_x and VOCs. The differences that would arise for ozone are tested on periods recently when high ozone had previously occurred in the UK, with the models using historical meteorological fields.

Figure 13 shows the O_3 simulated at 13:00 on 27th August 2019 using the Met Office AQUM model for a control scenario with standard emissions, and with 0.3 x standard NO_x (i.e. -70% NO_X). The changes in O_3 are variable, with the most noticeable change being an increase over London where the supressing effect of urban NO emissions are reduced. However, the same model simulated no increase in O_3 over London for a similar episode in June 2017.

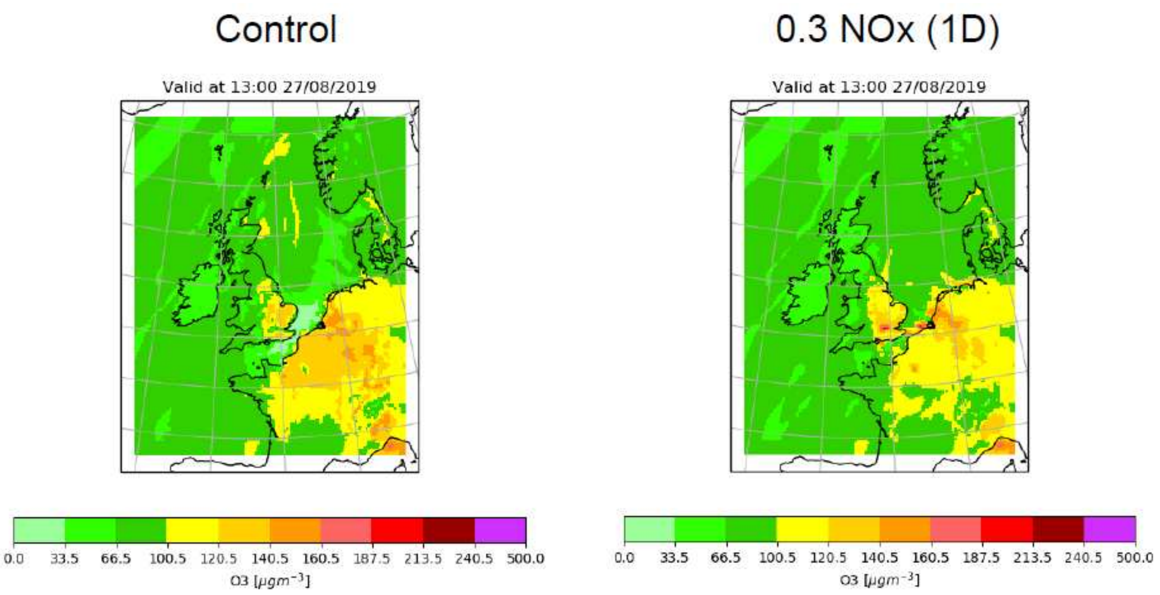

Figure 13. Estimated ozone concentration for a control scenario and a run with 0.3 x NOx reduction (-70% NO_X). Model run with meteorology from a known ozone event on 27th August 2019.

Reproduced from Agnew, P., Bright, V.B., Coward, K., Drummond, B., Hort, M.C., Malavelle, F., Molina-Jimenez, P., Nelson, N., Sherratt, B., Smith, E. *Met Office*, 2020. "Modelled impact of COVID-19 restrictions on pollutant emissions and summertime air quality in the UK."

Figure 14 shows the ozone response in the GEOS-Chem chemical transport model at noon on 25th August 2019 to three scenarios (-10%, -30% and -50% NO). The changes in ozone are variable, with no single direction of response. Higher concentrations of ozone are predicted across central regions of the UK with lower concentrations over Scotland and much of south east England, including London. It should be noted that the model resolution is ~30 km so reflects regional-scale responses to changes in NO emissions rather than localised ones.

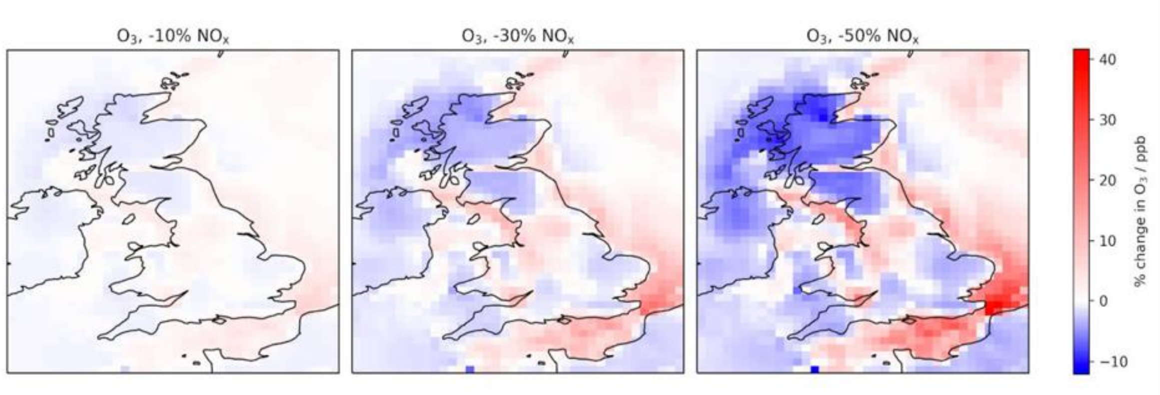

Figure 14. Estimated noontime changes in ozone for -10%, -30% and -50% NOx reductions, model run with meteorology from a known high ozone event on 25th August 2019.

Reproduced from Fakes, L. and Evans, M.J., *University of York*, 2020. “Predicting the influence COVID-19 restriction on air quality pollutants for summer 2020 with an atmospheric chemistry transport model.”

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?

A response to this question has been provided by the Committee on the Medical Effects of Air Pollutants (COMEAP) on the basis of the (limited) evidence received and using other contemporary academic and technical literature available at the time of writing. Membership and Secretariat is listed below. Further details can be found at https://www.gov.uk/government/groups/committee-on-the-medical-effects-of-air-pollutants-comeap

Committee on the Medical Effects of Air Pollution - Members

Professor Frank J Kelly, Chair, Imperial College London

Professor Alan R Boobis, Imperial College London

Dr Nicola Carslaw University of York

Ms Ruth Chambers (Lay member)

Professor Jonathan Grigg, Barts and the London School of Medicine, Queen Mary University of London, and Royal London Hospital)

Professor Anna Hansell, University of Leicester

Professor Roy Harrison, University of Birmingham

Dr Mike Holland, Freelance consultant

Professor Debbie Jarvis, Imperial College London

Dr Mark Miller, University of Edinburgh

Professor Gavin Shaddick, Statistics, University of Exeter

Mr John Stedman, Ricardo Energy and Environment

Dr Heather Walton, King's College London

Professor Paul Wilkinson, London School of Hygiene and Tropical Medicine

Professor Duncan Lee, University of Glasgow

Professor Mathew Heal, University of Edinburgh

Committee on the Medical Effects of Air Pollution - Secretariat

Ms Alison Gowers, Public Health England

Dr Karen Exley, Public Health England

Dr Christina Mitsakou, Public Health England

Dr Philippa Douglas, Public Health England

Dr Artemis Doutsi, Public Health England

Most regulated air pollutants are respiratory irritants and/or pro-inflammatory agents. Epidemiological time-series studies find associations between elevated concentrations of air pollution and increases in adverse respiratory outcomes, such as mortality and respiratory hospital admissions. These include exacerbations of chronic respiratory diseases, such as chronic obstructive pulmonary disease (COPD) and asthma; such exacerbations are often driven by viral infections. Long-term exposure to air pollution is associated with increased morbidity and chronic diseases, some of which have been identified as increasing the risk of severe COVID-19 symptoms. Given this, it would not be surprising if there was a link between exposure to air pollution and the occurrence or severity of COVID-19 infection, but currently there is no clear evidence on this or on the magnitude of any effect.

There is some evidence to suggest that nitrogen dioxide (NO_2), particulate matter (PM) and ozone (O_3) may increase susceptibility to respiratory infections or worsen disease prognosis. The evidence is suggestive in several areas, although there are insufficient studies or mixed evidence for specific combinations of endpoints, infection types, age groups or pollutants USEPA, 2016, 2019, 2020).

A number of epidemiological and other studies investigating possible links between air pollution and COVID-19 mortality have been reported in recent weeks. Most of these, including most of the studies mentioned here, are currently available as “pre-prints” and have not yet been subjected to the rigours of scientific peer review. Two ecological epidemiological studies have investigated the association between long-term average concentrations of air pollutants and COVID-19 mortality in the US.

These reported associations with different traffic-related pollutants: one (Wu *et al.*, 2020) found a link with fine particle ($PM_{2.5}$) concentrations while the other (Liang *et al.*, 2020) found mortality to be associated with NO_2, but that the association with $PM_{2.5}$ was statistically non-significant. Attempts to control for confounding by other risk factors have been made in both of these studies, to some extent, but it is nonetheless difficult to rule out residual confounding because many of the risk factors for disease transmission and severity are likely to be correlated with concentrations of air pollutants. A number of other studies which report correlations without appropriate attempts to adjust for confounding are also emerging in the literature but, because of their limitations, are not informative.

There are a number of possible mechanisms by which air pollutants could influence COVID-19 infection. These include non-specific impacts of inflammation on host immunity, specific impacts of pollutants on receptors such as angiotensin-converting enzyme 2 (ACE2) by which SARS-CoV-2 enters cells, and contribution of air pollution to cytokine production during infection thereby making a potential contribution to the cytokine storm that is a feature of Acute Respiratory Distress Syndrome seen in severe COVID-19 disease (Conticini *et al.* 2020).

An *in vitro* study in which high concentrations of traffic-derived particulate matter were applied to human airway epithelial cells appears to demonstrate a proof of principle that particulate pollution can up-regulate ACE2 (Miyashita *et al.* 2020). It is also possible that fine particles could act as a carrier for virus particles. A study in Northern Italy reported the detection of SARS-CoV-2 RNA on particulate air pollution (Setti *et al.*, 2020). However, there are a number of weaknesses in this study, including that contamination of the samples cannot be ruled out. Furthermore, only extremely low copy numbers were seen in those samples that had viral RNA.

Only undetectable or very low concentrations of SARS-CoV-2 RNA were found in aerosol samples taken in outdoor locations of Wuhan, except in locations in which large numbers of people gathered or passed, suggesting that infected carriers in the crowd were the likely source (Liu *et al.* 2020). The detection methods used in both of these studies do not provide information as to the viability of the virus. Further investigation into all of these possible mechanisms is urgently required.

Studies of the associations of COVID-19 disease with both past and contemporary air pollution exposure are limited by an, as yet, incomplete understanding of the factors controlling the transmission and progression of the disease, and especially individual risk factors. Elucidation of these factors by future research will greatly increase the capacity of studies of interactions with air pollution to detect and quantify any causal associations with the incidence and severity of the disease.

In summary, although there is, as yet, no clear empirical evidence that exposure to air pollutants increases the likelihood or severity of COVID-19 infection, knowledge of the impacts of air pollution on health suggests that this is likely. In addition, infection may temporarily increase subsequent responses to air pollution, in those with pre-existing illnesses. Potential interactions between air pollution and COVID-19 may be relevant to the future management of the pandemic in the UK and elsewhere

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

Much remains unknown surrounding the airborne transmission of the SARS-CoV-2 virus, and aerosol science is a key tool in elucidating some of the unknowns. There is strong circumstantial evidence that aerosol generated by mechanisms such as coughs, sneezes, breathing, resuspension of material, etc. provides a route for transmission of the virus. In support of this, there are a number of studies which have reported RNA signatures, identified by PCR, of the airborne virus, using air samplers/filters, collecting samples from room ventilation etc., although these studies do not mean the virus remains viable (and hence able to cause disease) when in aerosol.

Regarding the persistence of viable virus in the atmosphere, there have only a few studies so far, each at only one relative humidity (RH) . 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.

Even though the first indoor studies investigating viability have suggested the virus remains infectious for between 2 and >16 hours, and potentially longer on surfaces, the magnitude of the inhaled dose required for infection is as yet unknown. The particle size is also a key determinant of atmospheric lifetime. Large droplets (~100 μm) sediment rapidly within 1-2 m from source. Smaller respirable particles (< 5-10 μm) can be transported over longer distances (e.g. most recently in the context of SARS-CoV-2, Bourouiba L., 2020, estimated transport up to at least 7- 8 m).

Coughs and sneezes generate far more small particles of respirable size than the large droplets that sediment out and it is also suggested that these particles are produced in normal breathing, possibly enhanced by singing and talking. Consequently, aerosolised virus creates a hazard both through direct inhalation as well as by deposition to surfaces, the latter leading to other pathways of human intake.

One study has identified RNA from SARS-CoV2 virus internally mixed within urban particulate matter, but the impacts of this on viability/infectivity and the mechanism and range of transport are unclear. A study in a Wuhan hospital showed airborne SARS-CoV-2 RNA associated with particles from < 0.25 to 2.5-10 μm, a range with widely varying atmospheric properties, but capable of persistence in the air over many hours (and hence transport over distances of the order kilometres), but did not determine the presence of live virus. By analogy with similar organisms, it is likely that the virus will be deactivated relatively rapidly by reaction with atmospheric oxidants such as ozone and hydroxyl (OH). Attachment to larger particles would tend to slow this process due to the diffusion limitation of transfer of ozone to the particle surface. Attachment to larger particles would also increase the deposition velocity of particle-bound viruses and shorten their lifetime/transport distance. The well-recognised oxidative potential of airborne particles may also lead to decay and loss of viability of the virus.

A key factor is the importance of atmospheric dilution, which relates directly to inhaled dose. Far higher concentrations build up in a stagnant indoor environment than in a well-ventilated building or in an outdoor open space. Hence good ventilation and dispersion are important processes for reducing the inhaled dose, and hence the likelihood of infection. Evidence submitted by Carruthers *et al.* (CERC, 2020) illustrated the high dilution (by several orders of magnitude) of particles at 2 m away for a point source releasing at 1.5 m height, the extent of dilution being strongly dependant on mean airflow and turbulence. The impacts of face masks to limit the spread of virus from exhaled breath has been reviewed in a recent article by Prather *et al.*, 2020, the authors strongly advocating their use.

A number of these aspects are the subject of current study worldwide, with active programmes of research at a number of UK universities specialising in aerosol research, as described in the submissions by the Universities of Bristol (Reid, J., *University of Bristol*, 2020) and Manchester (Alfarra *et al., University of Manchester*, 2020).

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Annex 1.

AQEG received evidence submissions from the following organisations.

Air Quality Consultants
Arup Group
Atmo Ltd
BlockDox Ltd
CACP, University of Hertfordshire
Cambridge Environmental Research Consultants (CERC)
Cheshire East Council
Chartered Institute of Environmental Health
Condair Ltd
Department of Agriculture, Environment and Rural Affairs, Northern Ireland
Environmental Defense Fund, Europe
Environmental Protection UK
Fareham and Gosport Borough Council
Greater London Authority
Hull City Council
Kings College, London
Kirklees Council
Lancaster University
Lawnside Ltd
Leeds City Council
Met Office
Mid Devon District Council
National Centre for Atmospheric Science
Phlorum Ltd
Plymouth Marine Laboratory
Premier Diagnostics Ltd
Reading University
Ricardo Energy & Environment
Skanska Technology Ltd.
Swansea University
Tewkesbury Borough Council
Transport Scotland
UK Centre for Ecology and Hydrology
University of Birmingham
University of Bristol
University of Cambridge
University of Derby
University of East Anglia
University of Edinburgh
University of Leeds
University of Leeds (Institute for Transport Studies)
University of Leicester
University of Leicester (National Centre for Earth Observation)

University of Liverpool (multiple departments)
University of Manchester
University of Nottingham
University of Southampton
University of York (Wolfson Atmospheric Chemistry Laboratories)
University of York and NASA Goddard Space Flight Center
Wakefield Council