Covid-19, Air Quality and Mobility Policies:
Six European Cities

March 2021

Experts in air quality
management & assessment

Document Control

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Executive Summary

The response to the 2020/21 Covid-19 pandemic in Europe has been unprecedented in modern times, with major European cities shutting down economic and social activity, or locking down, for extended periods of time. While the extent and severity of lockdowns varied between cities, all experienced an improvement in air quality as a side effect. Transport and Environment (T&E) commissioned Air Quality Consultants (AQC) to:

1. analyze air quality data for six (initially seven, see below) European cities – Berlin, Brussels, Budapest, London, Madrid, Milan and Paris – to assess the air quality improvement between typical levels and those experienced during the most stringent phase of lockdown
2. estimate the contribution that traffic makes to typical air quality in the cities, and
3. attempt to replicate the air quality improvement by either converting the vehicle fleet to zero exhaust vehicles (ZEV) or increasing walking, cycling and home working, through a series of policy scenarios.

AQC used air quality monitoring data (mainly) supplied by the city authorities to assess the "real" reductions in nitrogen dioxide (NO_2) concentrations, by removing the effects of weather and other regional impacts. This used a boosted regression tree (BRT) model and revealed reductions in mean NO_2 of between 3.2µg/m^3 (Budapest) and 27.3 µg/m^3 (Central London), with Paris showing the highest city-average improvement, at 20.6 µg/m^3.

The traffic emissions were estimated by subtracting urban background concentrations from those measured at traffic stations, and assuming the difference is due to emissions from road vehicles. Using data on fleet make up, traffic levels and emission factors (locally sourced for Berlin and London, data purchased from emisia for the other cities), the traffic emissions were assigned to the different vehicle classes. Note that, due to issues with data from the identified urban background station, it was decided that the outputs for Milan were not suited to the methodology used in the analysis and so the results for Milan have not been included in this report.

Two types of scenarios were investigated: Two that rely on a switch to ZEVs only, and three that look at a combination of ZEVs with modal shift and demand reduction. The findings show that in all cities - except Madrid and Paris - both types of scenarios can achieve reductions equivalent to lockdown levels: focusing on cars only, between 42% (Budapest) and 92% (London) of all car-km need to be shifted to ZEVs. When using a combination of ZEVs with more walking, cycling, public transport and reduced car use, the necessary shift to zero-emission cars can be achieved more rapidly, with a need to shift between 6% (Budapest) and 74% (London) of all car-km to zero exhaust emissions.

In two of the cities (Madrid and Paris), which have seen particularly strong reductions of traffic-related NO_X emissions during most stringent phase of lockdown, only a mix of both strong modal shift and a switch to ZEVs more widely (including also vans and trucks) can deliver the targeted reductions. In Paris 67% of all vehicle-km need to be switched to ZEVs. In Madrid, 10% of all km travelled by light and heavy-goods vehicles as well as 94% of car-km need to convert to ZEV.

An additional sixth scenario was included to provide a longer-term outlook on the switch to ZEVs. If all cars, vans, buses, motorcycles and trucks in the selected cities were zero exhaust emissions, then NOx emissions from traffic would be eliminated completely and $PM_{2.5}$ emissions reduced by between 22% to 66%, although this range is closely related to whether or not locally generated emissions data were used.

The analysis was completed in October 2020 and reviewed in March 2121, and was based on the most stringent phase of lockdown policies in Europe.

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1 Introduction

Purpose and aims of the study

1.1 Poor air quality remains one of the leading, current risks to human health and the environment in Europe. While great strides have been made in reducing air pollution, with some pollutants reduced to barely measurable levels, there remain significant challenges and, with them, significant health impacts for the European population.

1.2 Interventions to control the Covid-19 pandemic have massively disrupted social and economic activity across Europe and have drastically reduced road traffic volumes. This is likely to have reduced emissions of traffic-related air pollution, as illustrated in the EEA's Covid-19 impacts tracker[1] and numerous independent studies[2]. There has been widespread media coverage, both of the air quality improvements caused by the pandemic, and also of potential relationships between exposure to air pollution and the severity of health responses to the virus[3]. The benefits of cleaner urban air have received renewed public interest and there is, therefore, an opportunity to examine in greater detail, the interrelationships between mobility policies, air quality, and health.

1.3 Countries and cities emerging from the initial, most stringent stages of their 'lockdowns' presented an opportunity to reimagine urban transport. As economic and social activity increases post-Covid-19 pandemic, there remains a danger that the public avoids public transport in favour of private cars. Alongside this, there may be increased pressure to relax, or at least delay, the introduction of emission control measures such as Low Emission Zones. Likewise, there is a unique opportunity to demonstrate the impact that action on polluting forms of transport can have, and the health benefits that such actions bring, in order to stimulate a demand that such outcomes can be achieved and maintained through more radical transport and mobility policies. This study is intended to support the latter and help inform long-term solutions in the context of a more virus-conscious world.

1.4 In December 2019, the European Commission published the European Green Deal, a roadmap for achieving key environmental outcomes in the EU by 2050[4]. It proposes that the EU achieves net zero carbon emissions by that point and also creates a "zero pollution environment" in Europe. It is not entirely clear what "zero" means in relation to air pollution; the stated ambition of bringing the EU air quality Limit Values into line with the WHO Guideline Values would not necessarily

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[1] https://www.eea.europa.eu/themes/air/air-quality-and-covid19/monitoring-covid-19-impacts-on

[2] Including AQC's own: https://www.aqconsultants.co.uk/CMSPages/GetFile.aspx?guid=1222ff30-3c9f-4189-b353-2f2ee50edab1

[3] For example: https://www.bbc.co.uk/news/health-52351290

[4] https://ec.europa.eu/info/strategy/priorities-2019-2024/european-green-deal_en

eliminate all air emission sources. The Green Deal commits to the adoption of a Zero Pollution Action Plan for Europe in 2021 and so further details will emerge over the coming months. However, the stark impact made by lockdown policies and the need to achieve net zero carbon across the Community has the potential to shift the debate towards higher ambition. Nevertheless, the use of clearly stated policy measures, supported by robust and transparent analysis will assist in making that shift both more likely and ambitious.

1.5 The objective of this study is to demonstrate how practical and sustainable mobility policies could replicate the air quality benefits of the most stringent Covid-19 lockdown periods within the context of the European Green Deal. The study will provide:

- A demonstration of the real impact of the most stringent lockdown policies on urban air quality versus business as usual, using real data from key exemplar cities, adjusted for meteorology; and
- A robust assessment of the impact of low pollution mobility shifts in the public, private and commercial fleets relative to the lockdown impacts and business as usual.

Choice of cities

1.6 The aim of this study was to analyse a representative sample of cities in Europe rather than to be fully comprehensive. The project budget constrained this to a total of seven cities covering a variety of climactic conditions and policy approaches to the Covid-19 pandemic. The seven cities chosen were:

- Berlin
- Brussels
- Budapest
- London
- Madrid
- Milan
- Paris

1.7 However, the assessment methodology used in this study could be applied to any city, assuming suitable air quality and traffic data can be identified.

1.8 This report sets out the analysis methodology used, the assumptions made and associated uncertainties, the policy scenarios analysed and the outputs for the cities. More extensive information on the air quality data obtained from the cities can be found in the Appendices.

1.9 Note that analysis results are shown for only six of the seven cities. The methodology employed for this project is crucially dependent on air quality monitoring data, both to assess the magnitude of the change in air quality during the most stringent phase of lockdown and to establish the traffic contribution to air quality in the cities. During analysis, issues arose with regard to the monitoring data supplied for Milan, in particular the high concentrations reported at the urban background station in relation to the roadside stations. Comparison with data for the other cities, for which the background to roadside e relationship was relatively consistent, showed Milan as a significant outlier, which had a large impact on the analysis of the traffic contribution. In consultation with Transport and Environment and its stakeholders, it was agreed that **the results for Milan should be excluded from this report.**

2 Analysis methodology

Determining the Extent of Reduced Urban NO_2, NOx, and $PM_{2.5}$ Emissions Associated with Covid-19

Data Collection

2.1 Measured hourly mean concentrations of NOx, NO_2, O_3, and $PM_{2.5}$ were obtained for all monitoring sites in and, in the case of O_3, around each city; these were identified or suggested by the city representatives themselves, or could be identified from published databases. Concurrent hourly measured meteorological data were also collected for each city from suitable representative sites. The sources of data for each city are set out in Table 1. Data were collected for the five calendar years 2015 to 2019, as well as for the period Jan 2020 until 13th May 2020. Only those sites which met the following data capture criteria were included in the analysis:

- At least 75% valid observations between 1st January 2015 and 31st December 2020;
- At least 75% valid observations between 1st January 2020 and 15th March 2020; and
- At least 75% valid observations between 16th March 2020 and 13th May 2020.

2.2 All of the sites within a city which met these criteria for NOx and NO_2, and/or $PM_{2.5}$ were included in the first stage of the analysis. It was considered most appropriate to define a single background O_3 monitoring site and meteorological monitoring site as representative of each city. In those cases where more than one site could have been selected, the site was chosen based on professional experience, taking account of the siting of each available instrument.

2.3 Table 2 shows the number of available monitoring sites in each city which met the above criteria. Maps showing the locations of sites are given in Figure 1 to Figure 6 and details of all sites are given in Appendix A2.

Table 1: Sources of Air Quality and Meteorological Monitoring Data Used for Each City

City	Data Sources Used
Berlin	Data provided directly by the Senate Department for the Environment, Transport and Climate Protection
Brussels	Data provided directly by the Belgian Interregional Environment Agency
Budapest	Data provided directly by the Greenpeace Hungary and BKK Centre for Budapest Transport
London	UK Automatic Urban and Rural Network, Air Quality England network, Londonair network, UK Meteorological Office.
Madrid	Data provided by the EMT Madrid & taken from the published database from the department of Air Quality
Paris	Data taken from the Airparif website

Table 2: Number of Sites with Suitable Data Capture in Each City

City	Number of Monitoring Sites	
	NOx + NO_2	$PM_{2.5}$
Berlin	17	10
Brussels	7	3
Budapest	5	9 (PM10)
London	86	14
Madrid	23	6
Paris	23 (NOx), 22(NO_2)	5

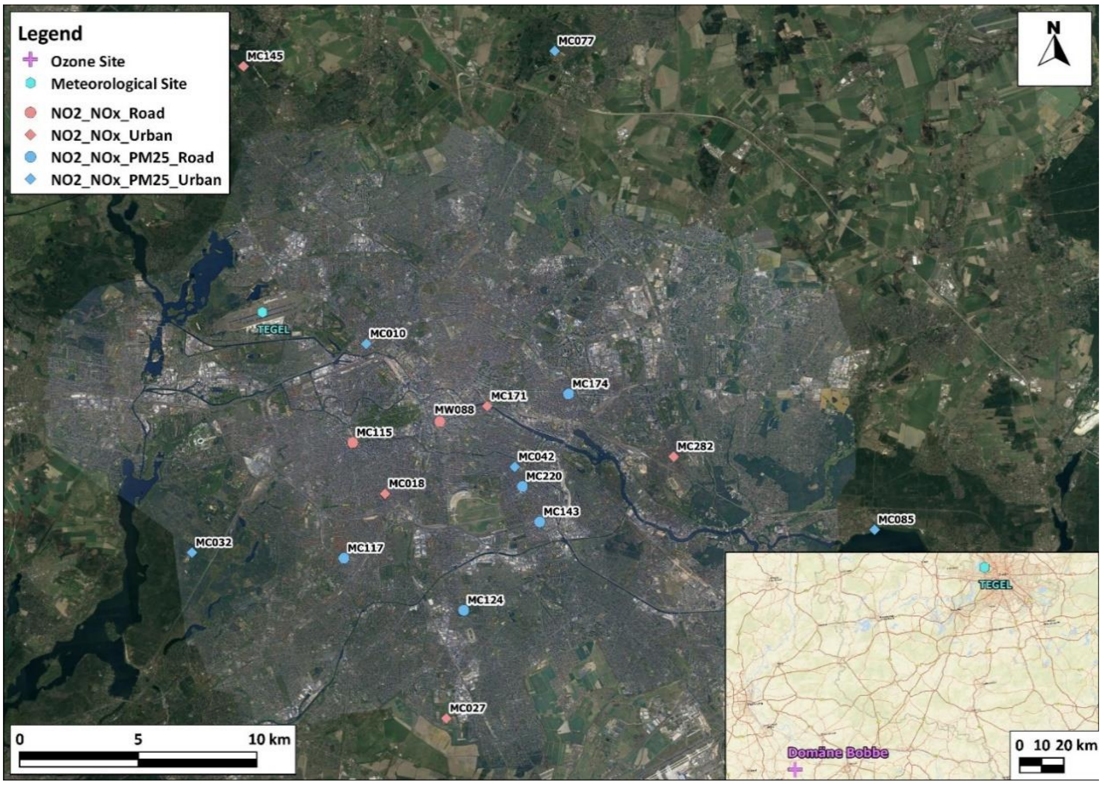

Figure 1:Air Quality Monitors in Berlin

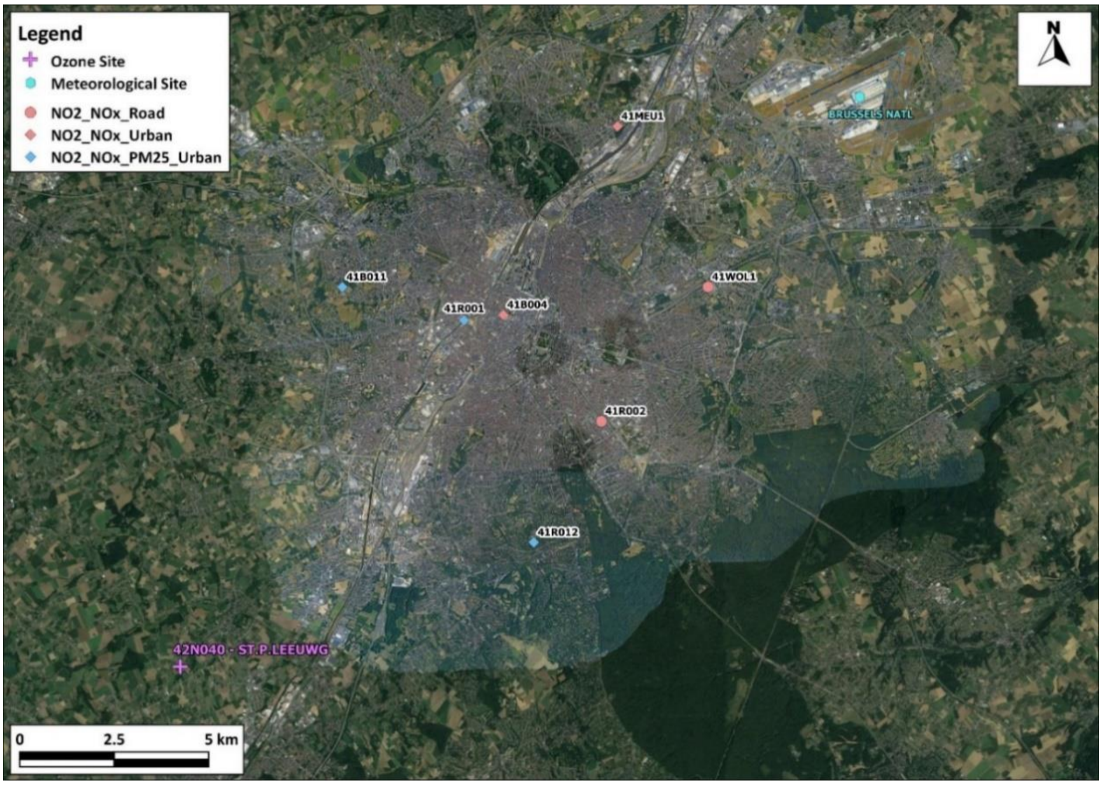

Figure 2: Air Quality Monitors in Brussels

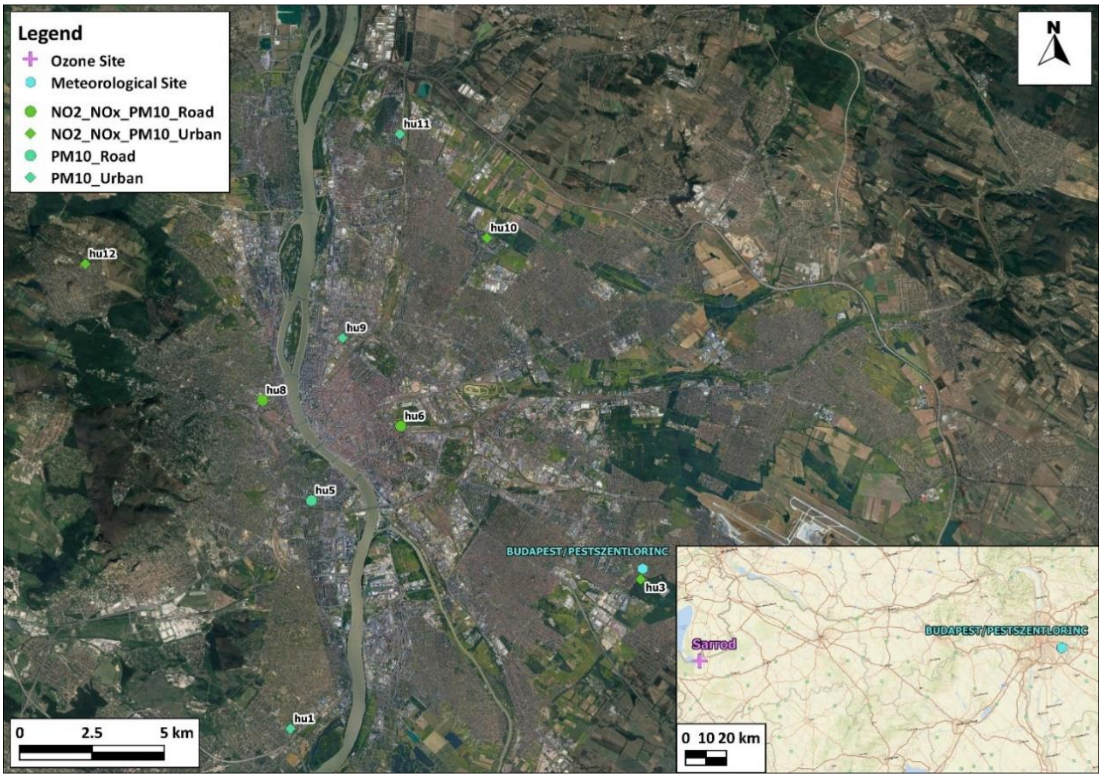

Figure 3: Air Quality Monitors in Budapest

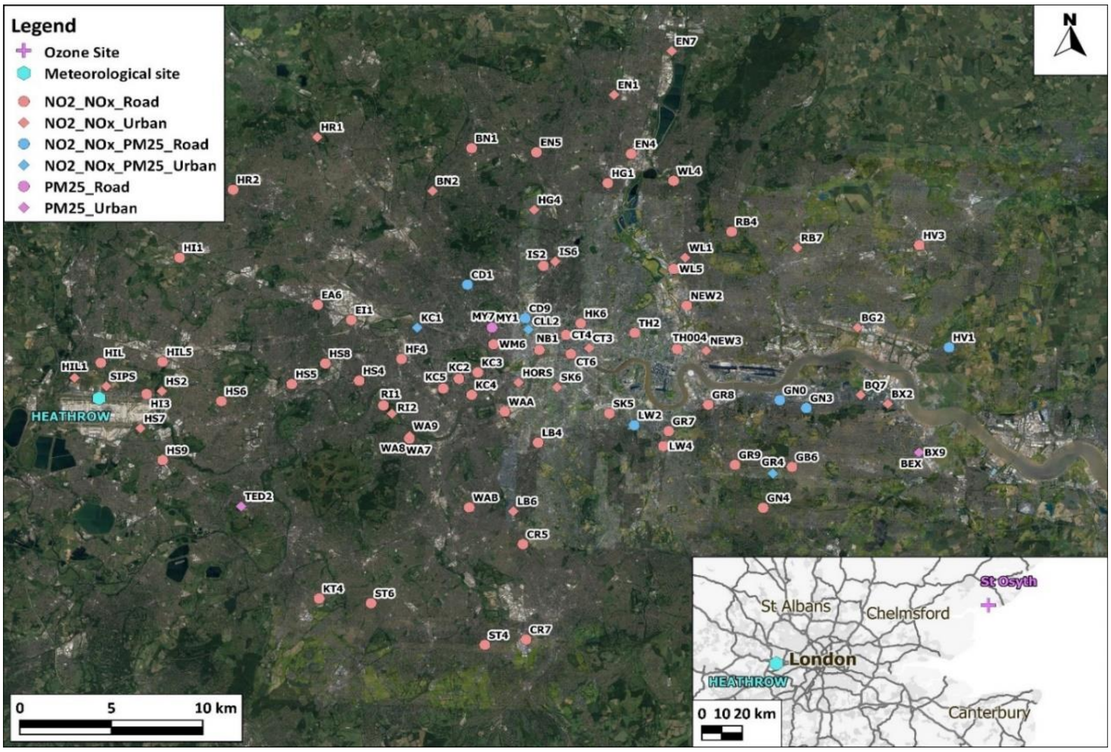

Figure 4:Air Quality Monitors in London

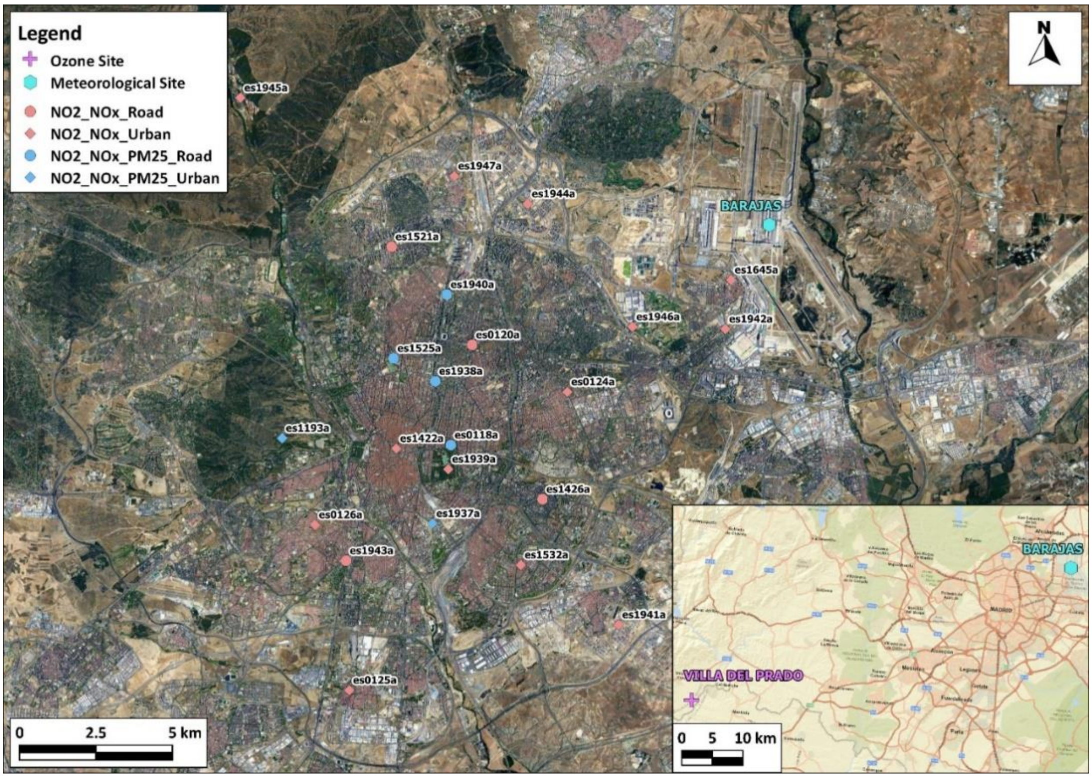

Figure 5: Air Quality Monitors in Madrid

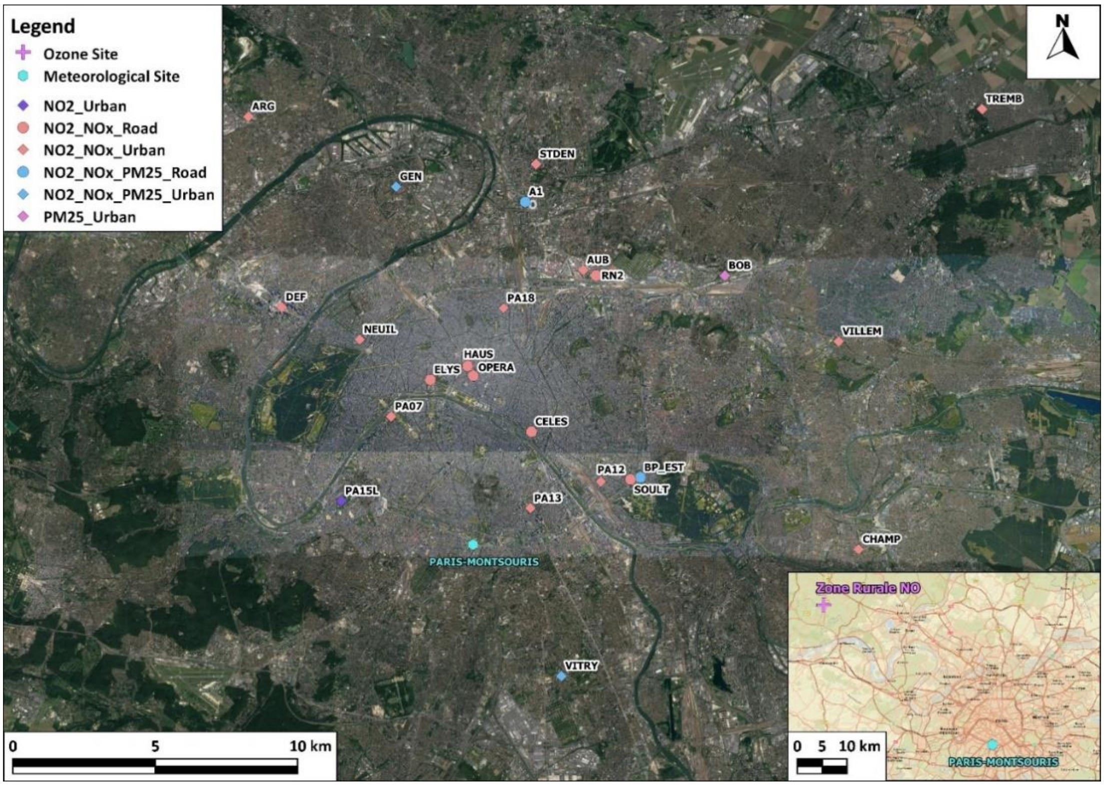

Figure 6: Air Quality Monitors in Paris

Traffic Data

2.4 Typical pre-pandemic traffic flows, with a detailed breakdown of the fleet composition, for the local road networks, along with traffic flows during the most stringent lockdown periods, were requested from each city's representative (see Table 1). Where local road network information was not available, COPERT data provided by Emisia[5] were used instead.

Accounting for Meteorological and Routine Temporal Variability

2.5 Ambient pollutant concentrations are significantly affected by meteorology. Furthermore, the period during the start of lockdown in many countries was affected by notably atypical weather, e.g. as reported by (AQEG, 2020). It is, thus, necessary to normalise the measurements for these effects in order to determine the underlying trend in concentrations which might be driven by changes to emissions.

2.6 This analysis follows on from that described by Carslaw and Taylor (2009), Carslaw et al. (2012) and AQC (2020). Boosted Regression Tree (BRT) models have been constructed to predict the dependence of NOx, NO_2 and $PM_{2.5}$ concentrations on each of the parameters listed in Table 3. The models have been constructed by comparing, within a machine learning environment, the measured hourly-mean concentrations with the concurrent descriptor parameters (Table 3), over the full 5+ years of measurements. The BRT models, which are specific to each individual monitoring site, thus predict how each of the parameters in Table 3 influences concentrations at that monitor, both alone and in combination with one another. All of the calculations have been performed using 'openair' (Carslaw and Ropkins, 2012).

2.7 The BRT models were then used to normalise weather (and the other parameters in Table 3) by running multiple (200) simulations across the range of different variables, and then averaging across these results. This process is sometimes known as 'deweathering' although it should be noted that in this case the parameters which have been accounted for include routine temporal factors (thus accounting, by way of example, for systematic differences in emissions on different days of the week) and also for regional background O_3 concentrations.

2.8 The rationale for including regional background O_3 as a descriptor parameter in the models is not straightforward. This is because O_3 concentrations at the roadside monitors will be systematically different from the regional background and because the Covid 19 lockdown might reasonably be expected to have changed the regional background O_3. However, previous work (AQC (2020) and (AQEG, 2020)) has shown that while the roadside O_3 concentrations reduced during lockdown (in response to reduced local NO emissions) BRT-adjusted background O_3 concentrations appear to have been relatively unaffected. It is also relevant to note that inclusion of regional background O_3 within the

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[5] https://www.emisia.com/about-us/

BRT models resulted in markedly improved model performance for both NO_2 and NOx. Since the NOx concentration is independent of the O_3 concentration, this suggests that regional background O_3 is acting as a proxy for other factors, most likely related to long-range transport, which cannot otherwise be controlled for within the models.

2.9 Because the BRT models are built using machine learning, repeating the same model-build process multiple times using identical input parameters results in marginally different outputs each time. The degree of divergence between different sets of outputs provides an indication of the general ‘stability’ of the models. Where there are significant changes in concentrations over the model build period which cannot be explained in terms of the parameters in Table 3, this tends to amplify the divergence between model runs. In essence, the models struggle to describe the observations when those observations are not dependent on the available descriptor parameters. This tends to be the case for pollutants which are dominated by long-rage transport and can also be caused by using unratified measurements (in which case the ability of the model is constrained by the quality of the measurements). The approach taken has been to repeat each analysis three times and thus produce three separate sets of BRT-adjusted concentrations. A visual check of these three BRT-adjusted time-series has been carried out to confirm that the degree of variance is qualitatively small. The average from these three datasets has then been reported.

Table 3: Descriptor Parameters Used to Build BRT Models

List of Parameters Considered
wind speed, wind direction, air temperature, relative humidity, hour of day, day of week, week of year and background O_3 (with O_3 concentrations excluded from the $PM_{2.5}$ BRT models)

2.10 In general, pollutants without a significant long-range component, such as NOx and NO_2, can be described reasonably well using BRT models built using these descriptor parameters. This means that once these parameters are controlled for, BRT-adjusted concentrations become relatively consistent; in practice the BRT-adjusted daily mean concentration on any day of the year is likely to be very similar to the BRT-adjusted concentration measured on any other day. This relationship fails where an additional factor, which the model cannot control for, affects concentrations. A step-change reduction in emissions (for example caused by Covid-19 restrictions) would reasonably be expected to cause a step-change reduction in BRT-adjusted roadside concentrations. Any change in BRT-adjusted concentrations over a given time series must be caused by factors which the BRT-models cannot account for. Where the time series shows stable BRT-adjusted concentrations pre-lockdown, with a step change to lower, but stable, BRT-adjusted concentrations during the most stringent phase of lockdown, then it is reasonable to infer that this change has been caused by the lockdown.

2.11 In the case of $PM_{2.5}$, concentrations are significantly affected by long-range processes which are not described well by the parameters in Table 3. The same set of conditions described using the parameters in Table 3 might coincide with very different ambient concentrations, simply because the key controlling factor is not included in the listed parameters. This means that the models are not able to make precise predictions of the dependence of concentrations on each of the model input parameters (the models attempt to infer a causal relationship where none exists). Unfortunately, it has not been possible to determine appropriate descriptor parameters to reliably control for changes to $PM_{2.5}$ concentrations. It is, thus, expected that the approach is more limited for $PM_{2.5}$ than it is for NOx and NO_2.

Defining the Study Period

2.12 The study relies on comparing BRT-adjusted concentrations measured before the Covid-19 lockdown with those measured during the most stringent phase of lockdown. Lockdown began at different times in different cities, and also began to be eased at different times. Table 4 sets out the periods over which BRT-adjusted concentrations have been averaged in each city in order to define pre-lockdown and most stringent lockdown mean concentrations.

Table 4: Monitoring Periods of Interest

City	Pre-lockdown Period	Most stringent lockdown Period
Berlin	01/01/2020 – 15/02/2020	23/03/2020 – 20/04/2020
Brussels	01/01/2020 – 29/02/2020	18/03/2020 – 30/04/2020
Budapest	01/01/2020 – 29/02/2020	28/03/2020 – 30/04/2020
London	01/01/2020 – 29/02/2020	23/03/2020 – 30/04/2020
Madrid	01/01/2020 – 29/02/2020	14/03/2020 – 30/04/2020
Paris	01/01/2020 – 07/02/2020	17/03/2020 – 30/04/2020

Identifying the Influence of Road Traffic

2.13 It is helpful to distinguish between the increment to concentrations caused by emissions from roads that might lie adjacent an individual monitoring site, and that which is caused by emissions from all of the roads in the city. While NOx and NO_2 concentrations measured close to busy roads tend to be dominated by local emissions, the aggregated influence of all other roads across the city can still be appreciable. The focus of this analysis has been the total increment to concentrations caused by all traffic emissions within each city.

Defining ‘Background’ Concentrations

2.14 The first step has been to separate the sites into those which represent ‘background’ conditions, well away from any roads or other local emission sources, and those sited close to busy roads. Initially this has been based on the site identifiers included in each monitoring network (Table 1). However, aerial and street-level imagery has been used to verify these groupings. The objective has been to determine the concentrations measured in locations well away from any roads or other emissions sources. Some of the sites, nominally classified as background, were considered to be too close to roads or other emission sources (e.g. a bus depot) to provide an ideal indication of non-traffic background concentrations and were excluded. The sites which were retained, and used to define background conditions in each city, are given in Table 5.

Table 5: Sites used to Define Background Conditions in Each City

City	Sites used to Define Background NOx and NO_2 Concentrations
Berlin	MC282, MC085, MC077, MC171, MC145, MC027, MC018, MC042, MC032
Brussels	41B011, 41B004, 41R012, 41MEU1
Budapest	hu12
London	IS6, HG4, GR4, HORS, HIL1, RI2, NEW3, WL1, SIPS, BX2, BX1, EN7, CT3, BG2, BEX, BQ7, BN2, HS2, HR1, KC1
Madrid	es1939a
Paris	STDEN, PA15L, VILLEM, TREMB, NEUIL, GEN, PA13, PA07, AUB, ARG, DEF, CHAMP, VITRY.

2.15 At many of the background monitoring sites sited well away from roads, the BRT-adjusted measurements for NOx and NO_2 show a relatively constant time-series throughout 2020 (i.e. what is essentially a flat line of constant daily-mean concentrations each day). This is because, once the expected response to the parameters in Table 3 is accounted for, there is no residual change in measured concentrations. Examples of this are Sites MC027 in Berlin and BN2 in London (see Appendix A3). Some of the background monitors which are less 'ideal' in that they are still measurably affected by transport emissions, show a clear 'step change' in BRT-adjusted concentrations at around the time of the most stringent phase of Covid-19 lockdown. Examples of this are Sites es1939 in Madrid and PA07 in Paris (see Appendix A3).

2.16 While there will undoubtedly have been changes to non-transport emissions caused by the Covid-19 lockdowns, which are discussed further in Section 3, the principal changes appear to have been related to transport. It has thus been assumed that the best representation of background concentrations which are unaffected by the influence of road traffic is given by the BRT-adjusted concentrations measured at background sites during the most stringent phase of lockdown. This is because those background sites which are affected by nearby road traffic will have experienced a much smaller traffic effect during the lockdown (while at those sites not affected by roads, the BRT-adjusted concentrations measured during lockdown are indistinguishable from those measured pre-lockdown).

2.17 For Berlin, London and Paris, the BRT-adjusted pre-lockdown background NOx and NO_2 concentrations show an apparent gradation across the city, with higher concentrations toward the centre, and lower concentrations toward the edges (see Figure 7 to Figure 9). In order to provide the most representative indication of background concentrations at each roadside site, the site-specific BRT-adjusted pre-lockdown background NOx and NO_2 concentrations have been interpolated using kriging.

2.18 For Brussels, Budapest and Madrid, there are insufficient background monitors to define the spatial gradation in non-traffic concentrations across the city. The approach for these three cities has been to take either the single available value, or the average of all available values and assume no spatial variation in this non-traffic background field.

2.19 For $PM_{2.5}$, there are insufficient monitors to reliably interpolate a background field and so the approach has been to take either the single available value, or the average of all available values.

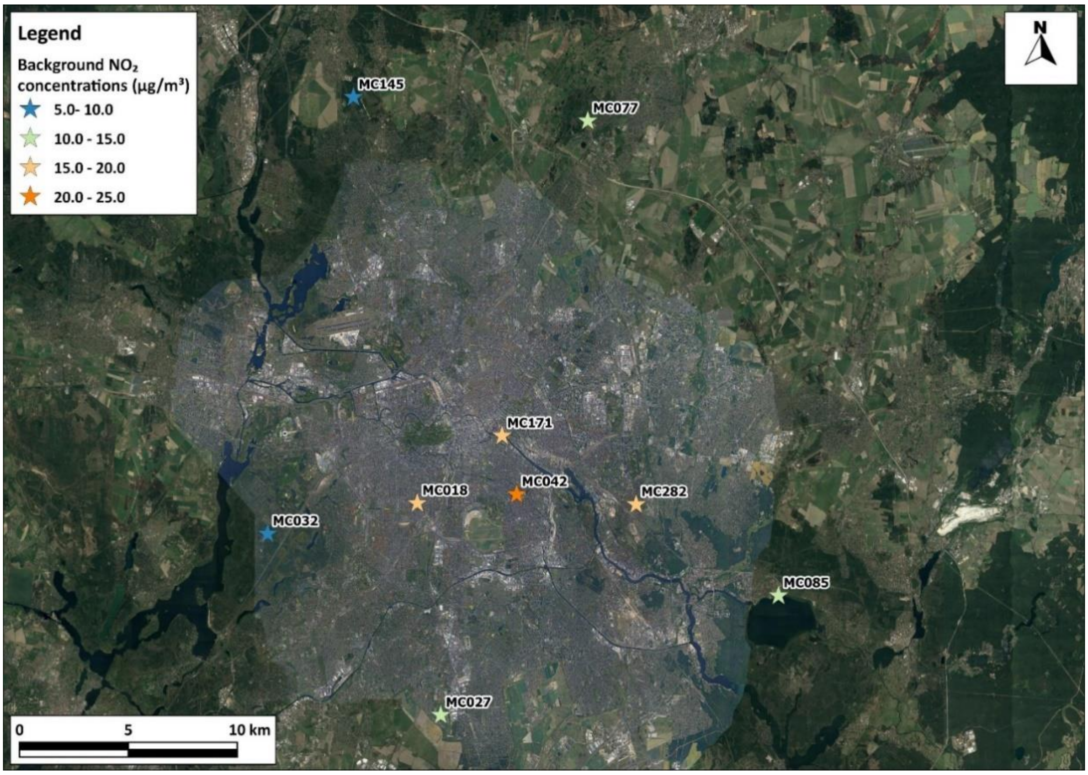

Figure 7: BRT-adjusted pre-lockdown background NO_2 in Berlin

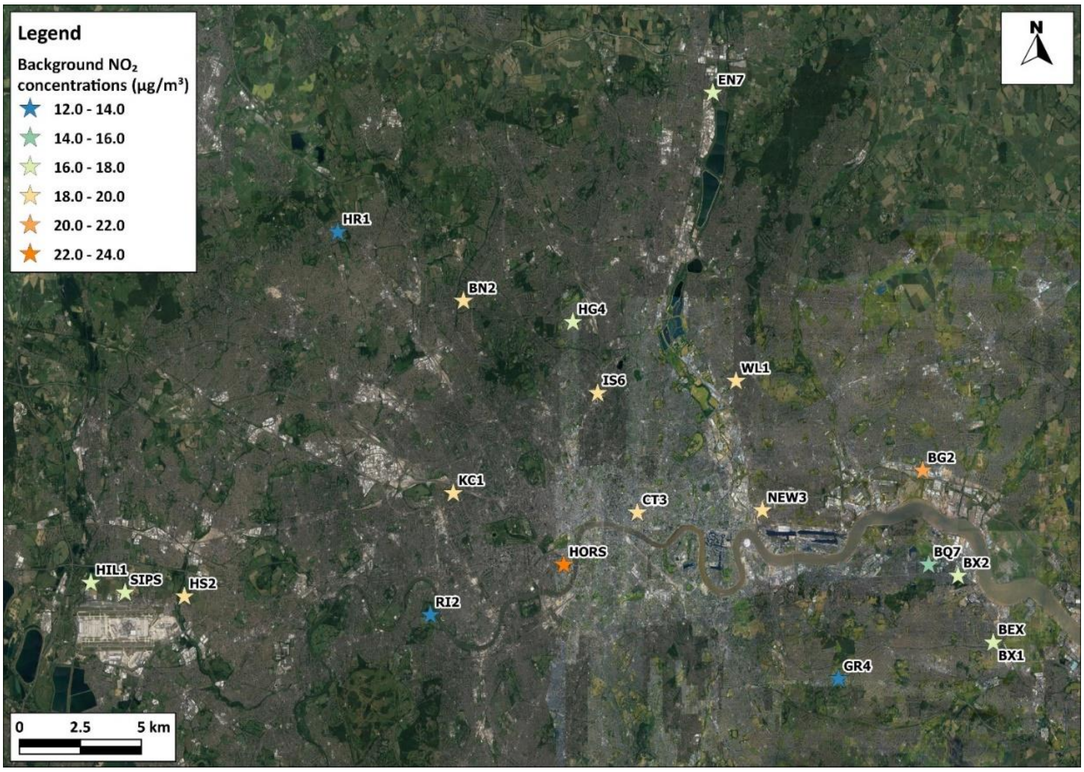

Figure 8: BRT-adjusted pre-lockdown background NO_2 in London

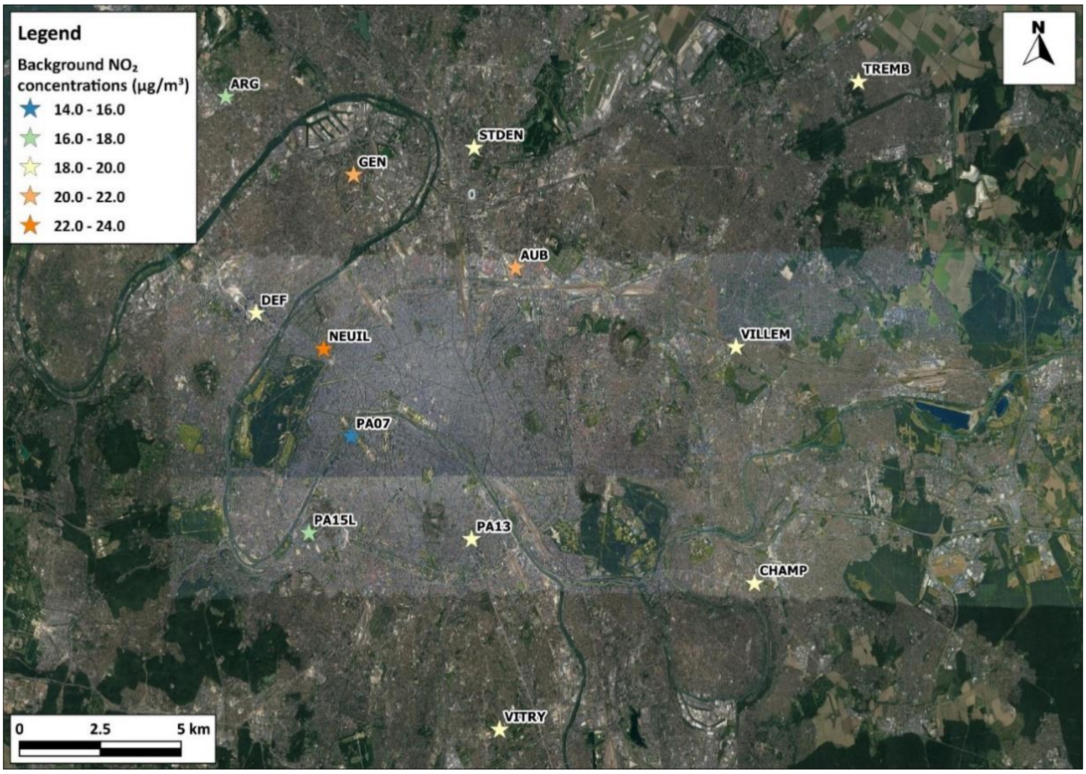

Figure 9: BRT-adjusted pre-lockdown background NO_2 in Paris

Defining the Road Traffic Increment

2.20 There were a number of roadside sites which were considered unsuitable for this analysis. Most of these were in London, where the greater number of monitors provided an opportunity to refine the analysis by excluding those instruments which were not ideal for the study. These were: site HI3, which was removed owing to its proximity to the runway of Heathrow airport; sites NEW2, WAA, and WA8, which were removed owing to concerns regarding the representativeness of the reported data; and sites GR8, HS9, HS6, and WAB which were removed owing to large contiguous gaps in the time series which, while not triggering the data capture criteria, made it unlikely that the BRT models would be able to accurately define trends. For Paris, site PA12, which is nominally classified as a background site, was reclassified as road-influenced for the purposes of this analysis owing to its proximity to a nearby road.

2.21 For each roadside monitor, the BRT-adjusted pre-lockdown background concentration has been subtracted from:

A) the mean of BRT-adjusted concentrations measured pre-lockdown; and

B) the mean of BRT-adjusted concentrations measured during the most stringent phase of lockdown.

These values have been taken to represent the road traffic increment of concentrations during these two periods.

Calculating the Effect of the most stringent phase of Lockdown on Air Quality

2.22 Comparing the mean of the BRT-adjusted measurements made during most stringent phase of lockdown with the equivalent measurements pre-lockdown shows the effect that the lockdown has had on total ambient concentrations. Comparing the road traffic increment of concentrations measured during lockdown with the equivalent measurements pre-lockdown shows the effect that the lockdown has had on this road traffic increment of concentrations.

2.23 The calculated improvements at an individual site are subject to considerable uncertainty (see Section 3). For this reason, it was considered appropriate to take the average improvements calculated across multiple monitoring sites within a city. However, in the case of Berlin, London and Paris, a clear spatial pattern in these calculated improvements is evident, with larger reductions in the centre of each city and smaller reductions outside of the centre. These patterns can be seen in Section 4. The decision was made to separate these three cities into inner and outer areas and calculate the average improvement seen at all monitoring sites within each of these zones. It should be noted that the absence of similar patterns in other cities is more likely to reflect the availability, and distribution, of monitoring sites, rather than a significant difference in activity patterns. It is also relevant to calculate the city-wide changes observed in Berlin, London and Paris without this additional spatial differentiation.

Calculating the Vehicle-specific Contributions to NOx and $PM_{2.5}$ Emissions in Each City

2.24 In order to calculate the effect that specific transport scenarios would have on emissions in each city, it is first necessary to calculate the relative contribution that different types of vehicles make to transport emissions of NOx and $PM_{2.5}$ in that city. This has been based on the best available information for each city. As explained in previous sections, discussions were held with the organisations responsible for air quality in each city. In the case of Berlin and London, this resulted in the identification or provision of detailed city-specific traffic activity and emissions data. For other cities, city-specific transport and activity data could not be provided, or otherwise identified within the timescale of the data analysis, and so less precise data were used. The approach taken in each city is set out below.

Berlin

2.25 The Senate Department for Environment, Transport and Climate Protection in Berlin provided access to its official emissions inventory for Berlin. This includes traffic flows and calculated NOx and $PM_{2.5}$ emissions by vehicle type for all main roads in the city. The data cover the 2015 base year and a forecast for traffic and emissions in 2020 (with these predictions made prior to the Covid-19 pandemic and thus unaffected by it). Link-specific emissions were calculated by the Senate Department based on link and period-specific service descriptors using the Handbook Emissions Factors for Road Transport (‘HBEFA’) model. HBEFA is used by the Federal Government of Germany for reporting transport emissions to the European Commission and also underpins the Senate Department’s official inventory for Berlin. Emissions are provided by vehicle type for NOx, exhaust $PM_{2.5}$ and non-exhaust $PM_{2.5}$.

2.26 As explained in Paragraph 2.23, different parts of Berlin appear to have experienced markedly different changes to air quality during the most stringent phase of Covid-19 lockdown. It is thus helpful to calculate emissions separately for separate parts of the city. There is, however, no firm basis on which to define the boundaries of parts of the city which experienced different changes (since these were measured at discrete monitoring sites). It is, however, noted that all of the largest reductions in concentrations during lockdown occurred within the Berlin Low Emission Zone (LEZ) but outside of the residential neighbourhood of NeuKölln. The approach has been to define the ‘central’ area for the emissions calculations as those areas within the LEZ but not within NeuKölln, and to define the ‘outer’ area for this study as the remainder of the city. It should be noted that this arbitrary division has only a small effect on the results, which does not seek to place fine dividing lines around the fleet intervention scenarios. However, dividing the emissions inventory in this way does allow a recognition that the fleet in the centre of the city is systematically different to that in the outer neighbourhoods and that the effect of the Covid-19 lockdown was also different in the centre than elsewhere.

2.27 Emissions from all of the major roads contained in the Berlin emissions inventory have been aggregated according to whether they fall in within the central area (defined as within the LEZ but outside of out NeuKölln) or within the outer area (defined as the rest of the inventory). Emissions have also been calculated for the city as a whole, which are those from the entire inventory.

London

2.28 Emissions in London are published as part of the London Atmospheric Emissions Inventory (LAEI)[6], covering all major and minor roads in the city. The most recent year for which emissions have been published is 2015. The LAEI also includes link-specific annual average traffic volumes by vehicle type, as well as annual average link speeds, for all major roads in London; as well as total traffic by vehicle type and average speed across all minor roads within each 1 km x 1 km grid square of the city. The most recent year for which these flows are available is also 2015. The UK Government publishes a model called Trip End Model Presentation Program ('TEMPro')[7] which predicts the rate of change in traffic activity over time within each UK local authority (the London Boroughs in this case). This has been used to calculate the rate of change in traffic flows between 2015 and 2020 within each London Borough; and thus the total traffic flows in 2020. These datasets do not take account of the Covid-19 pandemic and give a best estimate of the annual average traffic flows that would have occurred in 2020 without the pandemic.

2.29 The UK Government also publishes an emissions factor database (the Emissions Factors Toolkit – EFT[8]) which is based on the COmputer Programme to calculate Emissions from Road Transport ('COPERT') model and takes account of local fleet composition data collected by the UK Department for Transport and, specifically for London, by Transport for London for different zones within the city. The COPERT model has been used by the UK Government, and many other member states, for reporting transport emissions to the European Commission. The EFT (v9.0) has been used to calculate emissions from each major road link and area-averaged collection of minor roads within London in 2020 (the LAEI also includes data for some roads outside of London and these have been excluded). As with the outputs for Berlin, emissions have been output separately for different vehicle types, and for NOx, exhaust $PM_{2.5}$, and non-exhaust $PM_{2.5}$.

2.30 As explained in Paragraph 2.23, different parts of London appear to have experienced different changes to air quality during the most stringent phase of Covid-19 lockdown. The LAEI already separates London roads into three zones "central", "inner", and "outer". All of the monitoring sites used to characterise the larger relative changes observed in the centre of London are within the "central" zone as defined in the LAEI. The changes in measured concentrations during lockdown

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[6] https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory--laei--2016

[7] https://www.gov.uk/government/publications/tempro-downloads

[8] https://laqm.defra.gov.uk/review-and-assessment/tools/emissions-factors-toolkit.html

of those sites within the "inner" LAEI zone were virtually indistinguishable, on average, from those in the "outer" LAEI zone (for example, the average percentage change in measured NO_2 concentrations in the "inner" LAEI zone is within half of one percent of that in the "outer" LAEI zone). It was thus determined that there was no benefit to this study in separating London into three zones. Link (and grid-cell) emissions were, thus, aggregated according to two zones. Emissions in central London are those released in the LAEI "central" zone. Emissions in outer London are the sum of those released in the LAEI "inner" and "outer" zones.

Brussels, Budapest, Madrid, and Paris

2.31 In the case of Brussels, Budapest, Madrid, and Paris, city-specific emissions data and city-specific traffic flows and/or fleet compositions could not be identified within the timescale of the analysis. The approach adopted was to rely on datasets relating to total urban traffic emissions in 2018 for each country. These emissions have been calculated using the COPERT model and are based on the estimated fleet composition and average peak-hour and off-peak link speeds within the urban areas of each respective country. These data are collected by Emisia, who produce the COPERT model, and take account of data sources including the European Commission's official statistics (Eurostat), the European Commission Statistical Pocket Book, the European Automobile Manufacturers' Association (ACEA), the Motorcycle Industry in Europe (ACEM), the European Environment Agency's CO_2 monitoring database, the European Alternative Fuels Observatory (EAFO), and the United National Framework Convention on Climate Change (UNFCCC). While necessarily less precise than city-specific activity or emissions data, the COPERT data can be considered to provide a reasonable estimate of the total traffic emissions within each country and the relative contribution to that total made by different vehicle types. Focusing on the urban subset of these data provides a reasonable estimate of the emissions summed across all urban areas within the country. The source-apportionment of this urban total has been taken to represent that of each of the cities considered in this study (e.g. the relative contribution of passenger cars to the total traffic NOx emissions in urban areas of Belgium is taken to represent the relative contribution of passenger cars to the total traffic emissions in Brussels etc.). Data have been generated for each vehicle type and for NOx, exhaust $PM_{2.5}$, and non-exhaust $PM_{2.5}$.

Testing Different Emissions Scenarios

2.32 The observed improvements in BRT-adjusted NO_2 concentrations have been taken as the "targets" to be met by the different transport interventions. The observed improvements in the road traffic increment of BRT-adjusted NOx concentrations have been taken as a proxy for these NO_2 targets. In other words, has been assumed that if the traffic increment to NOx concentrations can be reduced to the levels observed during most stringent phase of lockdown, then the effects that the lockdowns had on NO_2 concentrations will also be replicated. As explained in Section 3, this approach does not allow for changes in primary NO_2 emission proportions associated with different traffic interventions, but it does provide the most practical means of linking traffic emissions with the observed reductions in NO_2 concentrations.

2.33 As explained in Paragraph 2.23, the observed reductions in concentrations have been averaged across whole cities, or large parts of each city. Within each of these areas, it has been assumed that there is a linear relationship between total NOx emissions from road transport and the traffic increment to NOx concentrations. While it would be possible to achieve the same reductions in concentrations by differentially targeting individual roads, the current study is concerned with area-wide changes to the vehicle fleet.

2.34 Because there is a linear relationship between NOx emissions from traffic and the traffic increment of NOx concentrations, this means that the relative observed reductions in the traffic increment of NOx concentrations can be read directly as relative reductions in NOx emissions from traffic.

Emissions Scenario Testing

2.35 The outcome from the emissions calculations described in Paragraphs 2.25 to 2.31 is an inventory, for each city or city zone, showing the relative contribution made by each vehicle type to total NOx emissions, as well as to $PM_{2.5}$ emissions for the exhaust and non-exhaust components separately. It has been assumed that the scenarios outlined in Paragraph 4.6 apply equally across all roads and all vehicles subtypes. For example, a 10% reduction is car vehicle-kilometres is taken to mean a 10% reduction in vehicle-kilometres driven by small Euro 6 petrol cars driving on predominantly congested roads and equally as a 10% reduction in vehicle-kilometres driven by large Euro 4 diesel cars driven on predominantly free-flowing roads, etc.. As explained in Section 3, some of the scenarios considered might reasonably be expected to reduce congestion and, thus, emissions from the remaining fleet, but it has not been possible to consider this effect within the analysis.

2.36 The assumption of a linear response across all vehicles and all roads means that there is also a direct linear relationship between vehicle-kilometres driven (for example by all passenger cars) and total emissions (for example by all passenger cars). This means, for example, that a 10% reduction in vehicle-kilometres driven by passenger cars will cause a 10% reduction in NOx emissions from passenger cars etc.

2.37 It is also relevant to note that fully electric vehicles have no exhaust emissions and so, in terms of NOx, there is assumed to be no difference between, for example, replacing 10% of the passenger car vehicle-kilometres with home working, and replacing 10% of passenger car vehicle-kilometres with electric vehicles. Both interventions will remove NOx emissions from 10% of the active fleet and, within the current model, reduce total NOx emissions from passenger cars by 10%. Furthermore, while the scenarios which consider a shift toward public transport have the potential to increase vehicle-kilometres driven by buses, the assumption has been made that all new trips generated in this scenario would be made by electric buses. Thus, in terms of NOx emissions, there is no practical difference between switching to home working and switching to public transport.

2.38 The approach has simply been to calculate the relative reduction in the emissions from each vehicle type which is required to bring about the observed reductions in NOx emissions from the road vehicle fleet as a whole. For example, if passenger cars make up 50% of NOx emissions from the entire vehicle fleet, then emissions from passenger cars would need to reduce by 70% in order to achieve a reduction of 35% (i.e. 50% x 70%) in total road traffic emissions.

2.39 The percentage reduction in NOx emissions from each vehicle type which would be required to achieve the scenarios outlined in Paragraph 4.6 have been calculated in this manner. The vehicle class-specific reduction in NOx emissions is then reported as a vehicle-class-specific reduction in vehicle-kilometres.

2.40 As explained in Paragraph 4.5 the BRT models were able to show the effect of the most stringent phase of lockdown on the traffic contribution to NO_2 and NOx concentrations but not $PM_{2.5}$. The approach for $PM_{2.5}$ has been to calculate the reduction in emissions which would be caused by meeting the NO_2 targets under each of the scenarios considered. For example, if 70% of the passenger car fleet would need to switch to electric vehicles in order to match the reductions in NO_2 observed during lockdown, then the focus has been to calculate the effect on $PM_{2.5}$ emissions of switching 70% of the car fleet in this way. These calculations have been carried out using the city-specific and city-zone-specific emissions calculated for each vehicle type and assuming that:

- interventions which switch non-electric modes to equivalent electric modes will remove all exhaust emissions while having no effect on non-exhaust emissions;
- interventions which switch non-electric modes to home working, walking, or cycling will remove both the exhaust and non-exhaust emission components; and
- interventions which promote the uptake of public transport have the potential to increase non-exhaust emissions from the bus fleet (but not exhaust emissions for the reasons given in Paragraph 2.37).

2.41 The extent to which increased public transport use would increase the vehicle-kilometres driven by buses will depend on a large number of factors and it has not been possible to account for this. Calculations have been made using two contrasting assumptions so as to provide a sensitivity test to the results. The first, and main, set of results assumes that there will be no increase in bus vehicle-kilometres associated with an increase in public transport use. The second assumes that every 10 car vehicle-kilometres saved will cause one additional bus vehicle- kilometre. In practice, because all of the additional bus-kilometres are assumed to be made by electric buses, the additional emissions relate to the non-exhaust component only. Calculated reductions in $PM_{2.5}$ emissions in the worst-case scenario for buses were all between 0.7% and 2.5% smaller than those based on assuming no additional bus trips. Within the context of this analysis, such small differences are considered to reflect a spurious level of precision and, as such, only the $PM_{2.5}$ emission calculations based on no additional bus trips have been presented.

3 Uncertainty and Limitations

3.1 This study has used the best available information to provide an indication of how the reductions in traffic emissions caused by the most stringent phase of Covid-19 lockdowns might be replicated using specific, managed fleet interventions. The analysis is intentionally high-level and thus, while the results can be considered as reasonable best-estimates, they should not be viewed as definitive or precise. The principal sources of uncertainty are outlined below.

Analyses of Ambient Measurements

Measurements

3.2 The results of the study are underpinned by the ambient concentration measurements made in each of the six cities. These measurements will be subject to a degree of uncertainty. More importantly, they represent individual sites which are each subject to location-specific influences. The results from these monitoring sites have been used to define changes to air quality across each city as a whole. These city-wide or zone-specific averages will be affected by the distribution of the monitoring sites from which they are derived; meaning that if additional monitoring sites were available for a city, then the overall averages would change. In the case of Paris, there appear to be clear and consistent patterns within each city zone (i.e. central vs outer), and it seems likely that the available measurements provide a reasonable representation of the zone-average effects of Covid-19. Similarly, the large number of monitors in London adds confidence to the calculated means. In the case of those cities with fewer monitoring sites, particularly where these show quite different results from each another (as is the case in Budapest), then the calculated city-wide averages are less certain.

Approach to Weather-normalisation

3.3 Normalisation for weather effects is essential for a study such as this but does not provide precise data. In particular, the machine learning algorithms give different predictions from each model run. While care has been taken to minimise uncertainty the weather-normalisation process will inevitably have introduced some degree of error.

3.4 There is also uncertainty introduced by the measurements of the parameters used to normalise the NO_2, NOx and $PM_{2.5}$ concentrations. In particular, the outcomes of the analysis are affected by the choice of meteorological site and O_3 measurement site. Testing with alternative datasets has shown that while specific values at individual air quality monitors are affected by site choice, the overall conclusions of the study are unlikely to be affected. In particular, it is noted that neither the meteorology, nor the O_3 data, are intended to capture individual street-level conditions but rather the overall environmental context for the city; on this basis, the precise location of the monitor is less important.

Subtraction of Concurrent Regional Background Concentrations

3.5 In order to determine how much of the observed reductions in concentrations are related to transport emissions it is necessary to estimate the non-transport component of concentrations at each roadside monitoring site. It is not possible to measure the non-transport component directly and so the approach has been to rely on measurements made during most stringent phase of lockdown at monitoring sites well away from roads.

3.6 A problem with this approach is that none of the background monitoring sites are perfectly placed to represent the background concentration field at any of the roadside sites. They are either sufficiently distant from the roadside monitor that the non-road background concentration field will differ between the two points; or the background monitors themselves may be affected by local emissions from road traffic or other sources. Where possible (for Berlin, London, and Paris), an interpolated background field has been used in order to minimise the first of these issues, as far as possible, but the interpolated background field remains limited by the siting of the background monitors from which it was derived.

3.7 To minimise the influence of traffic emissions on the measured background concentrations, the study has extrapolated the BRT-adjusted background concentrations measured during the most stringent phase of Covid-19 lockdowns. This is appropriate because BRT-adjusted daily-mean concentrations are largely constant throughout a calendar year - so long as no significant, external events (such as the Covid-19 lockdowns themselves) cause a disruption. However, this approach assumes that non-transport emissions were unaffected by the lockdowns, which is unlikely to be the case. It seems likely, for example, that emissions from heating in the central areas of cities might have reduced during the most stringent phase of lockdowns, while heating emissions in the suburbs increased, reflecting the redistribution of the population. This effect may have caused the extent of reductions in traffic emissions to be overestimated in the centre of cities and underestimated in the suburbs. In practice, this artefact is unavoidable, but the effect is likely to be small when compared with other sources of uncertainty.

Emissions Calculations

Activity Data

3.8 For Berlin and London, it has been possible to use traffic volumes and fleet compositions, as well as either average link speed, or service categories, taken from the official emissions inventory of the city. These data are considered to be robust but will still be subject to some uncertainty; particularly since, in each case, the data are predictions of activity levels in 2020 (without Covid-19 lockdowns) which were extrapolated from counts and predictions made in earlier years.

3.9 For the other five cities, it has been necessary to rely on national-level fleet and speed data (for urban settings) held by Emisia. These data are primarily intended for national-level reporting (for example, to the European Commission). While still providing a robust basis for high-level emissions calculations, these activity data are less precise than those available in Berlin and London.

Emissions Factors

3.10 The emissions factors used for Berlin are derived from HBEFA, while for all other cities the COPERT emissions factors have been used[9]. The different approaches have made the best use of the available information for each city. All road transport emissions factors are subject to uncertainty and, while both COPERT and HBEFA are ultimately derived from the same European Research for Mobile Emission Sources (ERMES) emissions database, each treats the data in different ways and is subject to different uncertainties. Overall, both emissions models provide equally valid results which can be considered as fit for purpose in terms of this study.

3.11 Non-exhaust emissions of $PM_{2.5}$ have been calculated within each model. The data which underpin the non-exhaust emissions factors is relatively old, and does not differentiate between different technologies and other features which might be expected to affect emissions (for example the use of regenerative braking). Given the uncertainties around the emissions factors, no attempt has been made to differentiate between non-exhaust emissions from conventional and electric vehicles. In other words, it has been assumed that switching from a conventional to electric vehicle will have no effect of non-exhaust $PM_{2.5}$ emissions. In practice, electric vehicles may have lower emissions from brake wear, on average, than conventional vehicles, but there remains some uncertainty regarding the effect of electrifying the fleet on average vehicle weight, which is linked to road and tyre wear rates. Assuming equal non-exhaust emissions from electric and conventional vehicles is the most robust approach currently available.

Congestion Effects

3.12 It has been assumed that there is a linear relationship between the total vehicle-kilometres for a particular vehicle type on a given road and the NOx emissions from that vehicle type on that road. In practice, this relationship will be non-linear because altering the total flow of vehicles will affect driving characteristics. Put simply, reducing traffic will often reduce congestion and reduce emissions from all remaining vehicles. It has not been possible to take account of this effect in the study but the same air quality benefits could be seen with smaller reductions in vehicle-kilometres for those scenarios which relate to increased home working etc. It is also possible that changes to working patterns, allowing more flexible start and end times, might reduce congestion effects and emissions for the same nominal traffic volumes. These details fall outside of the scope of this study.

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[9] For London, COPERT was used by application of the UK EFT, which embeds COPERT emissions factors with London-specific fleet compositions.

Primary NO_2

3.13 The target reductions have been based on measurements of NO_2 and NOx concentrations. The effects of managed fleet interventions have then been calculated in terms of NOx only. Achieving the observed reductions in NOx emissions should also achieve the observed reductions in NO_2 concentrations as long as the proportion of primary NO_2 emissions does not change. However, in practice, the Covid-19 lockdowns, and the managed fleet interventions, are both likely to be associated with a change in the fleet-average proportion of primary NO_2. This is because, for example, primary NO_2 emissions from passenger cars tend to be very different to those from buses, and selectively removing one vehicle type without removing the other will alter the fleet-average emission proportion. So long as the managed interventions have a similar effect on primary NO_2 proportions as occurred during the Covid-19 lockdowns (for example if the main effects were on car traffic), then it is still appropriate to assume a linear relationship between NO_2 and NOx in terms of achieving the transport-emissions-derived targets. Where the interventions target an alternative part of the fleet, then this may introduce some error.

Secondary $PM_{2.5}$

3.14 The calculations of $PM_{2.5}$ emissions have focused solely on primary particles. The managed transport interventions explored would all reduce emissions of NOx and also ammonia from road traffic. This has the potential to reduce the formation of secondary $PM_{2.5}$, but it has not been possible to include this additional benefit within the calculations, and the benefits to $PM_{2.5}$ of the different scenarios will have been under-predicted.

Lockdown Equivalence for $PM_{2.5}$

3.15 The effects of the most stringent phase of lockdown in each city have been calculated from observed NO_2 and NOx concentrations. The managed interventions have then been quantified based on these observed changes. The effects of the interventions on $PM_{2.5}$ emissions have been calculated, but as explained in Appendix A1, it has not been possible to calculate the effect of the lockdowns on $PM_{2.5}$ emissions. In practice, the reductions to $PM_{2.5}$ emissions during most stringent phase of lockdown are likely to be very similar (on a proportional basis) to the changes in NOx emissions. Many of the managed intervention scenarios involve a switch from conventional to electric vehicles and while this removes all local NOx emissions, it is only expected to remove a portion of the local $PM_{2.5}$ emissions. This means that, although the effects of the managed interventions on $PM_{2.5}$ emissions have been calculated, these interventions would be unlikely to be sufficient to recreate the reductions in traffic-related $PM_{2.5}$ emissions which occurred during lockdown.

Overall Uncertainty

3.16 It is not possible to quantify the overall uncertainty which is inherent in the results of this study. However, the data sources and assumptions made are fit for the purpose of providing high-level indications of how the air quality improvements experienced during the most stringent phase of Covid-19 lockdowns might be replicated using managed transport interventions.

4 Analysis scenarios

4.1 The first step in the analysis was to construct a baseline scenario, essentially modelling the changes in air quality brought about by Covid-19 response policies. The detailed methodology for this is described in Section 2 and was based on local measurements of nitrogen dioxide (NO_2) before and during the most stringent phase of lockdown. A de-weathering process was used to remove the influence of meteorology and other external factors. NO_2 was used as it is both a pollutant of prime concern for air quality policy and because there is a strong correlation between ambient concentrations in urban areas and traffic flow. It was assumed that changes in measured concentrations during most stringent phase of lockdown were due to changes in the road traffic contribution; uncertainties relating to this assumption are discussed in Section 3.

4.2 The initial work, thus, produced a baseline scenario which the subsequent analysis attempts to replicate through targeted changes in mobility policy. In consultation with T&E, six analysis scenarios were developed, building on the baseline scenario, with one scenario tested in two variants. These scenarios attempted to represent some of the ways in which changes to city mobility polices could attempt to replicate the air quality improvements seen in the baseline scenario; the scope of the project restricted the range of options tested which, clearly, could have been very extensive.

4.3 In each scenario, changes in the urban vehicle fleet have been simulated that either meet, or attempt to meet, the air quality benefit seen in the baseline scenario. The data available allows an estimate in terms of vehicle-kilometres and the results (in Section 4) are shown as the percentage change in vehicle-kilometres required, or the effect of a 100% change where the target (i.e. the reductions seen in the baseline scenario) is not met. An assumption that each vehicle in a class travels the same distance each year would mean that a proportionate change in vehicle-kilometres is equivalent to the same proportionate change in vehicle numbers. This is clearly not the case in reality – a similar result may be achieved by targeting only those vehicles travelling a greater distance – but the results do provide a guide to the level of change required.

4.4 In addition, the assumed change is applied equally across each vehicle class, i.e. the vehicle-kilometre reductions are applied equally, in percentage terms, to all Euro standard emission categories. It may be that applying a greater proportional change to older vehicles is more effective although, in the case of diesel cars, it is debatable whether, in reality, there is a significant difference in NOx emissions from all but the newest vehicles. However, data were not available to be able to undertake this analysis for all the cities selected for study.

4.5 The same analysis of ambient measurements which was carried out for NO_2 was also repeated for $PM_{2.5}$ (or PM_{10} where insufficient $PM_{2.5}$ measurements were available). However, $PM_{2.5}$ and PM_{10} concentrations are heavily influenced by long-range transport which is not straightforward to account for within statistical deweather models. Furthermore, minor differences in instrument type can confound any comparison between roadside and background measurements. The results of the analysis of ambient $PM_{2.5}$ measurements is described in Appendix A1, but the effect of city lockdowns on traffic-related air quality can be seen much more clearly in the NO_2 and NOx measurements than in the $PM_{2.5}$ measurements. The ambient $PM_{2.5}$ measurements have not, therefore, been used to define the baseline, Instead, the change in $PM_{2.5}$ emissions from road transport associated with achieving the NO_2 baseline was calculated for each scenario. It was assumed that electric vehicles[10] (EV) had the same emission rate for non-exhaust $PM_{2.5}$ as their internal combustion engine (ICE) equivalents, and so only the exhaust component changed, unless conversion was to non-emission transport options, such as walking and cycling.

4.6 The six scenarios modelled, A-F, were as follows.

- **Scenario A, ICE cars to EV**: The proportion of ICE passenger car kilometres that would need to be converted to zero tailpipe emission, in order to reach the baseline.

  Rationale: The sales of electric cars are increasing faster than projected and seem to withstand the effect of the pandemic (see T&E analysis).

- **Scenario B, ICE cars, vans and trucks to EV**: Conversion of 10% of heavy good vehicle (HGV) and light good vehicle (LGV, i.e. vans) vehicle-kilometres to EV (constrained due to likely availability of EV options), with the remining benefit to be achieved by converting ICE passenger car kilometres to zero tailpipe emission.

  Rationale: A shift of 10% of vehicle-kilometres for LGV and HGV is ambitious but not impossible, especially when looking at cities leading the transition (scenario based on T&E modelling).

- **Scenario C1, ICE cars to EV and non-transport or non-emission transport**: Conversion of 10% of ICE car vehicle-kilometres to teleworking, cycling or walking and all buses to EV, with the remaining benefit to be achieved by converting ICE passenger car kilometres to zero tailpipe emission. A sensitivity analysis was undertaken whereby the first part of the scenario resulted in an increase in bus kilometres (i.e. more services were run), but that these new buses were also EV. This resulted in a negligible increase in $PM_{2.5}$ emissions.

  Rationale: The success of modal shift from car traffic to other modes has been limited in practice, even if many cities pursue ambitious goals. Vienna, one of Europe's leaders in this regard, has achieved a reduction of 6.3% between 2010-2015 on all roads and 11.2% in the city centre. The effect of teleworking and Covid-19 still remains to be seen. We

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[10] For the purposes of this project, EV could include both fully battery powered electric vehicles and hybrid vehicles running in full electric mode, i.e. it assumes zero tailpipe emissions.

therefore assume a realistic but ambitious shift of 10% of car-km for the short to mid-term. As regards e-buses, T&E has been analysing the market and currently assumes that 100% zero emission bus sales across Europe are possible in the 2020s and hence a 100% zero-emission fleet in leading cities could be achieved in the next years.

- **Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport**: The same as for Scenario C1, except 10% of LGV and HGV kilometres were also converted to EV.

  Rationale: as for C1.

- **Scenario D, Long term ICE phase out**: Convert 10% of car kilometres to teleworking, walking and cycling, followed by an equal part of car, bus, LGV and HGV kilometres to EV.

  Rationale: the EU aims for a long-term shift to net-zero CO_2 and zero pollution; hence all ICEs must be phased-out in the longer term.

- **Scenario E, class by class switch to EV**: convert, by turn, 100% of all ICE cars, LGVs and HGVs to EV.

  Rationale: test model effectiveness and assess the absolute influence of each vehicle class on air quality in the cities.

4.7 The outcome for these scenarios is presented, city by city, in Section 4.

5 Baseline and Scenario Analysis Outputs

5.1 This section describes the outputs from the baseline and scenario analyses described in Section 2. The scenarios themselves are described in Section 4. Each city is described in turn. The outputs of the baseline scenario are described in terms of both the percentage reduction in the traffic NO_X emissions and the absolute reduction in NO_2 concentrations between the pre-lockdown and most stringent phase of lockdown, based on the de-weathered trends in monitoring data. It was not possible, with the data available, to undertake the same baseline analysis for $PM_{2.5}$ although it was possible to estimate the reduction in $PM_{2.5}$ emissions from each of the scenarios.

5.2 The outputs of the baseline analyses were used to set a target reduction in NO_X emissions which changes to mobility policy should try to replicate. The analysis methodology and available data means that the required changes are set out in terms of the reduction in passenger car kilometres (or all vehicle kilometres, in the case of Scenario D) necessary to achieve the same reductions in traffic NOx emissions. This reduction varies in each scenario, depending on the preceding stages in the analyses. For example, Scenario A is simply the conversion of conventional, i.e. petrol- and diesel-powered, passenger cars (ICE) to full electric (EV), thus the entire reduction of NOx emissions relies on passenger cars. For Scenario B, emissions are reduced first by converting 10% of LGV and HGV kilometres to electric, and then achieving the remaining required emission reduction through passenger car conversion. Thus, the required conversion is slightly lower for Scenario B than for Scenario A.

5.3 Note that the scenarios do not presuppose the method of implementation for the scenarios. Converting 75% of passenger car kilometres to EV may mean converting 75% of the cars themselves, or the focus could first be on high use vehicles, which could mean that fewer vehicles in total need to change. Nor does the analysis necessarily rule out the use of hybrid drivetrains, although, clearly, these would not be as effective at reducing exhaust emissions as full electric (unless fully “geofenced”, i.e. restricted to electric only operation in the cities).

5.4 The outputs from the scenario analysis are presented in three tables for each city, with corresponding charts. These show firstly the required reduction in passenger car kilometres required to meet the baseline followed by the resultant decrease in $PM_{2.5}$ emissions should this be achieved, and then the reduction in emissions from each vehicle class were it to be fully converted to electric. The reductions in $PM_{2.5}$ are not as great as for NOx given that there are non-exhaust emissions of $PM_{2.5}$ associated with vehicles – brake, tyre and road wear – which is assumed to be the same for conventional and fully electric vehicles. This is discussed further in Section 3 on uncertainties.

5.5 For some cities, it was not possible to achieve the target emission reductions through some of the scenarios. In such cases, the results are shown as a greater the 100% requirement (>100%), with the impact of a 100% conversion of passenger vehicle kilometres on emissions shown in the same table. There are two main reasons driving a greater the 100% conversion; either the local lockdown was more severe, resulting in a higher emission reduction target for the analysis, or because the local vehicle fleet is particularly clean, i.e. the removal of each vehicle-kilometre yields a lower emission saving. These are not mutually exclusive and could both be a factor.

Berlin

5.6 Berlin is the largest city in Germany, as well as its capital. In 2019, it had a population of just under 3.8m[11], which made it, officially, the largest city in the EU. There is an LEZ in operation in the centre of the city which means that roadside concentrations in the centre can be lower than those in the outer areas.

Baseline scenario

5.7 Appendix A3 shows the measured and BRT-adjusted measured NO_2 and NOx concentrations between January and May 2020 in Berlin. The dashed lines show the daily mean concentrations as measured, while the six bold lines show the results from six different BRT model runs. As explained in Section 2, each of these runs has used identical input data, and variability between the different outputs represents alternative decisions made by the probabilistic machine learning approach. More confidence can be had in results averaged across multiple BRT runs which predict very similar results. Where the BRT runs predict very different results, the results can be considered less certain (e.g. compare the results for Sites MC085 and MC282 in Appendix A3).

5.8 At most of the sites, there is an apparent change in the raw, observed concentrations coinciding with the start of lockdown in March 2020. However, this becomes much clearer in the BRT-adjusted data which, for all roadside sites, show an apparent step-change in concentrations roughly coinciding with the Covid-19 outbreak. The nature of the BRT adjustment means that the precise date of this step change is less relevant than the size of the step. Appendix A3 also shows how the BRT adjustment has effectively flattened what is otherwise a highly variable observed time-series, but that there are clearly observations which have presented the BRT models with an additional challenge, resulting in some significant deviations between the different BRT model runs.

5.9 Figure 10 shows the daily mean NO_2 concentrations averaged across all roadside sites, and across all urban background sites. It also shows the average of all daily mean BRT-adjusted concentrations[12]. Figure 10 is thus a compilation of all the site-specific results from Appendix A3. It shows that, taken on aggregate, there was a small reduction in concentrations measured at background sites which could not be explained by normalising for weather[13]. Thus, the BRT-adjusted data for urban sites in Figure 10 is marginally lower at the end of the time-series than at the start. The picture is quite different for roadside sites, where, taken on aggregate, there is a

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[11] https://www.statistik-berlin-brandenburg.de/publikationen/stat_berichte/2020/SB_A01-05-00_2019h02_BE.pdf

[12] i.e. the average across all individual BRT model runs (as opposed to a separate adjustment to the multi-site average data)

[13] The normalisation has accounted for more than just weather but for ease of reading, the term weather has been used.

clear step-change to lower concentrations during March, over and above that which can be explained by changes to the weather. The BRT models cannot ascribe this step-change directly to the Covid-19 lockdown, but show this is the residual trend when all known routine influences are accounted for. It is thus a reasonable assumption that this residual change was caused by the lockdown.

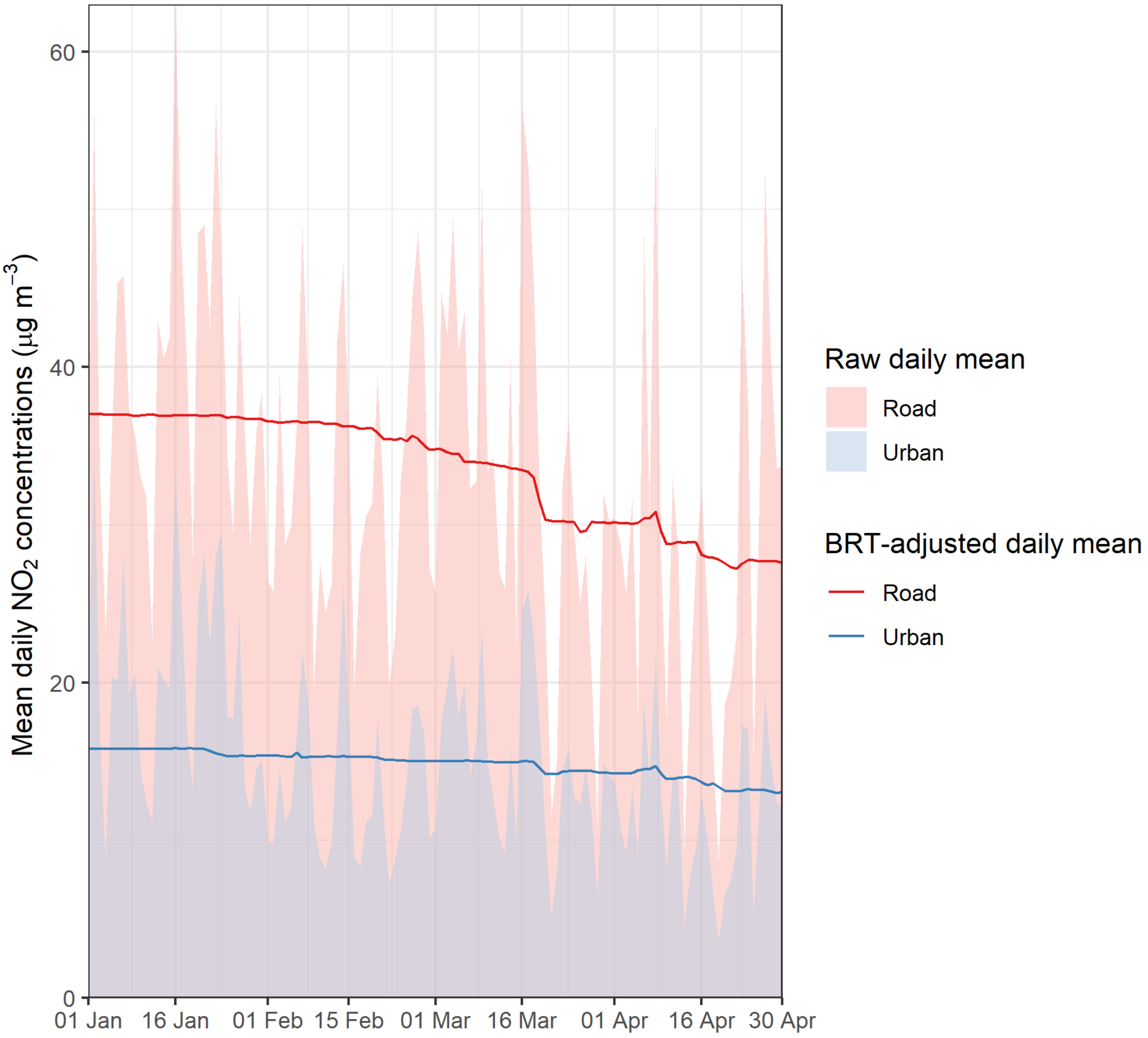

Figure 10: Mean Daily NO_2 Averaged across All Roadside Sites and across All Urban Background Monitoring Sites in Berlin

5.10 As explained in Section 2, the site-specific period averages before and during most stringent phase of lockdown have been calculated across all six BRT model runs. Appropriate non-traffic background values have then been subtracted to show the change in the road increment. The relative changes in the road increment to NO_2 concentrations at the seven sites which are suitable for this analysis are shown in Figure 11. There appears to be a spatial pattern in Figure 11, with the smallest reductions (<40% removal of the traffic contribution to NO_2 concentrations) in the outer areas of the city, and much larger changes in the centre. This pattern is also highlighted in Figure 12, which shows the observed reductions as a function of the mean concentration. It appears from Figure 12 that there is some consistency in the observed changes, with all reductions in the outer areas sitting within a small range (reductions of 24% to 36% in the traffic contribution to NO_2 concentrations), while both monitoring sites in the centre also show a similar pattern to each other (59% and 64% reductions).

5.11 While it is possible that the apparent distinction between the different city zones is driven by other factors, a similar pattern has been observed in London and Paris (as shown later in this section). It seems most likely that the reductions in traffic emissions were appreciably greater in the centre of the city than in the outer areas. The observed reductions have, thus, been averaged as shown in Figure 12, with one mean value calculated for the outer areas and another calculated for the inner city.

5.12 The overall patterns for NOx are, predictably, very similar to those shown for NO_2 and are not presented separately. Averages have been calculated for NOx in the same way as NO_2 and, as explained in Section 2, the calculated changes in NOx form the effective targets for the emissions calculations. This is on the basis that achieving the observed reductions for NOx would also achieve the observed reductions for NO_2.

5.13 The air quality changes due to the lockdown period are significant, with a mean reduction in NOx emissions from traffic of 28%, resulting in a mean drop in NO_2 concentrations of between 6.2 (outer) and 9.7 (inner) $\mu g/m^3$. However, this reduction is smaller in comparison to some of the other cities, and could reflect a relatively less severe lockdown. a cleaner vehicle fleet, or both. Berlin was, along with London, one of only two cities where local emission inventory data were available for this project; this will have introduced an inconsistency in the analysis which makes comparison between cities difficult. However, the use of local data, where available, was preferable to the use of more generic data, such as available from Emisia.

Figure 11: Map of Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Berlin

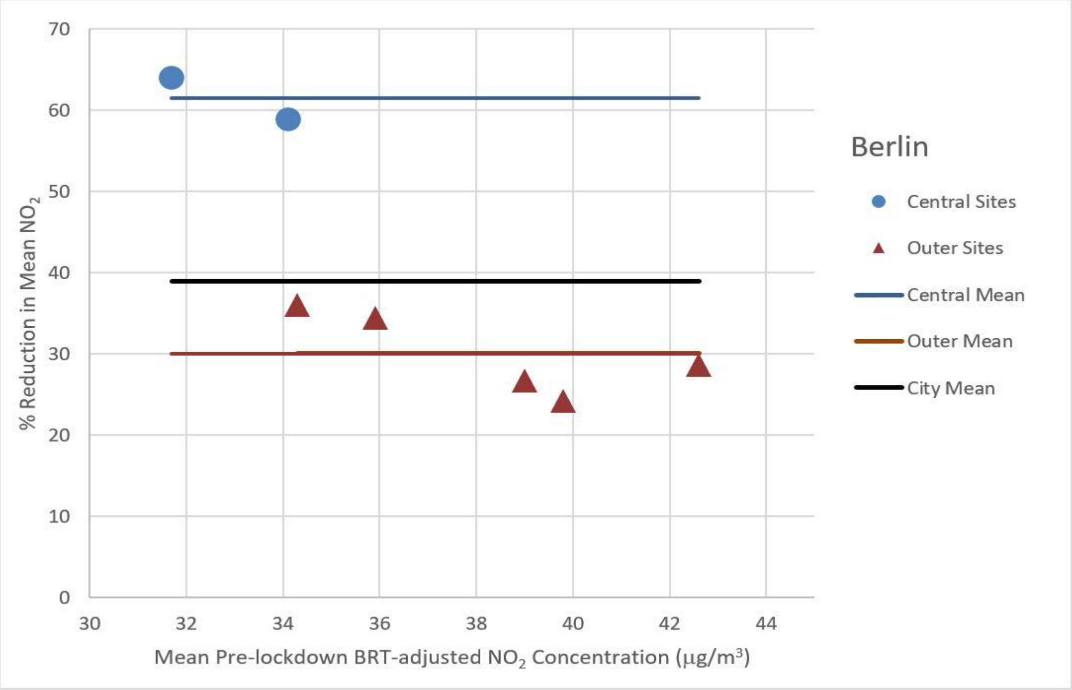

Figure 12: Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Berlin

Table 6: Baseline scenario outputs; Berlin

	Zone	% reduction in traffic contribution	Mean pre-lockdown concentration (µ/m³)	Mean lockdown concentration (µ/m³)	Mean change (µ/m³)	Roadside sites included
NO₂	Inner	61	32.9	23.3	-9.7	2
	Outer	30	38.3	32.1	-6.2	5
	City-wide	39	36.8	29.6	-7.2	7
NOx	Inner	47	61.7	43.4	-18.3	2
	Outer	20	80.7	69.1	-11.6	5
	City-wide	28	75.3	61.8	-13.5	7

Scenario Analysis

5.14 Table 7 shows that the emission reductions seen during the most stringent phase of lockdown are, theoretically, attainable through the mobility policies represented by the analysis scenarios. As expected, the level of conversion is higher for inner city areas than for outer, reflecting the greater emission change seen in the inner zone during lockdown.

Table 7: Outputs for scenario analyses, showing the conversion of ICE passenger car km to EV to meet target NOx emission reduction; Berlin

Scenario	Zone	Reduction in passenger car km
Scenario A, ICE cars to EV	Inner	84%
	Outer	37%
	City-wide	51%
Scenario B, ICE cars, vans and trucks to EV	Inner	79%
	Outer	31%
	City-wide	45%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	Inner	59%
	Outer	10%
	City-wide	26%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Inner	54%
	Outer	4%
	City-wide	20%
Scenario D, Long term ICE phase out[14]	Inner	44%
	Outer	15%
	City-wide	23%

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[14] The results for scenario D apply to all vehicle classes, not just passenger cars

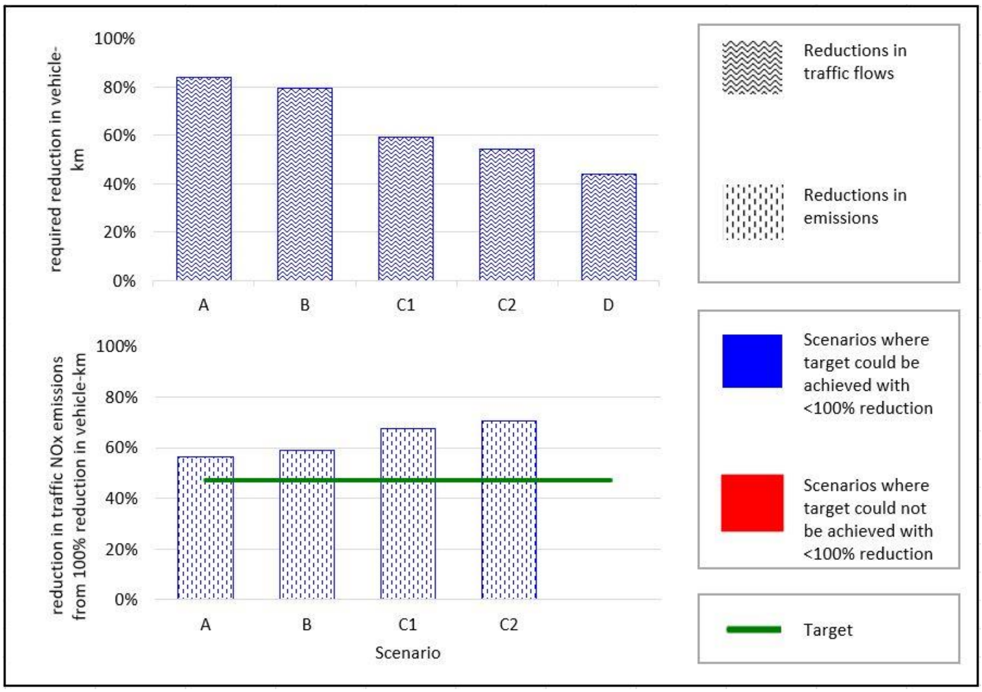

Figure 13: **Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Central Berlin**

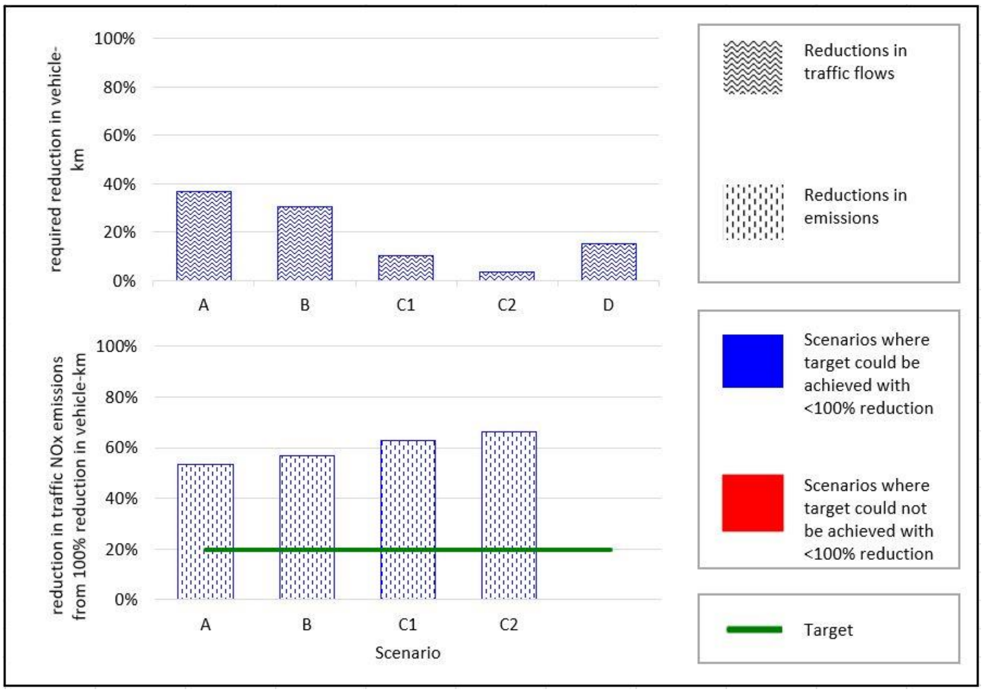

Figure 14: **Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Outer Berlin**

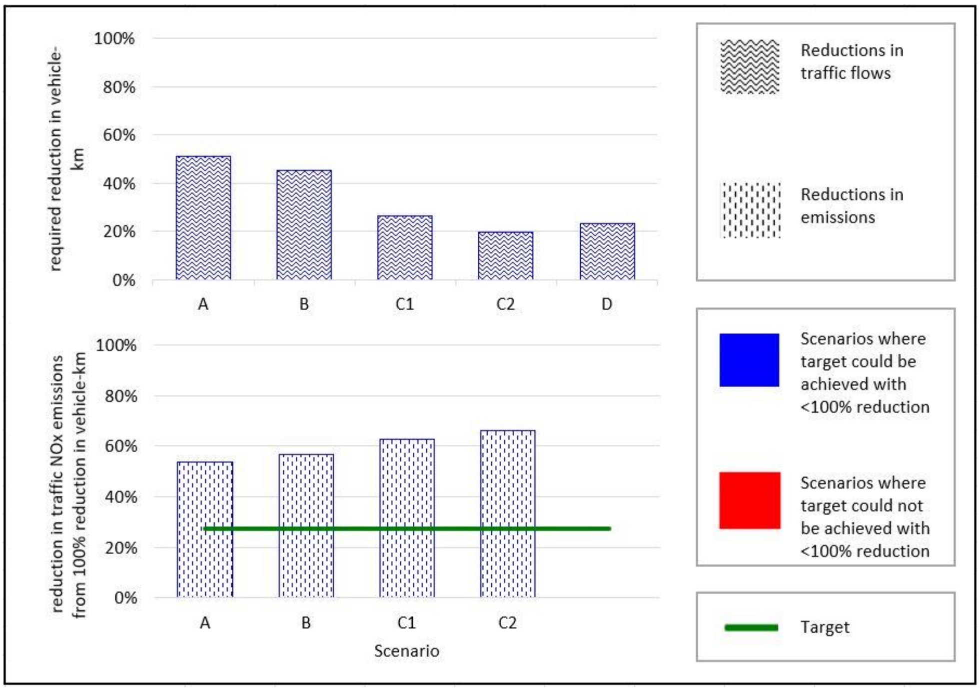

Figure 15: **Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Berlin City-wide**

5.15 Reductions in $PM_{2.5}$ emissions from full implementation of the scenarios are relatively modest, possibly reflecting a cleaner ICE vehicle fleet in current conditions. It is possible, given current levels of uptake, not just of EV and hybrid vehicles but higher Euro 6 standard ICE cars, that tailpipe emissions from road transport are not the largest source of $PM_{2.5}$ emissions in many European cities by 2030 and beyond. It is, therefore, not surprising that emission reductions achieved through conversion from ICE to EV may be relatively modest.

5.16 Table 9 shows the impact of converting each vehicle class, by turn, to EV, and the relative importance on each class in terms of emissions. The relative burden of reducing transport NOx emissions inevitably falls on passenger cars as they make up over half of the total emissions. However, for $PM_{2.5}$, LGVs become more important.

Table 8 Outputs for Scenario Analyses, showing traffic $PM_{2.5}$ emission reductions for achieving each scenario; Berlin

Scenario	Zone	Reduction in $PM_{2.5}$ emissions, traffic component
Scenario A, ICE cars to EV	Inner	11%
	Outer	5%
	City-wide	6%
Scenario B, ICE cars, vans and trucks to EV	Inner	12%
	Outer	5%
	City-wide	7%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	Inner	15%
	Outer	9%
	City-wide	11%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Inner	16%
	Outer	9%
	City-wide	11%
Scenario D, Long term ICE phase out	Inner	18%
	Outer	10%
	City-wide	12%

Table 9: Scenario E, class by class 100% switch to EV, emission reductions from the transport component; Berlin

Pollutant		Central	Outer	City-wide
NOx	Cars	56%	53%	54%
	LGV	15%	16%	16%
	HGV	11%	17%	16%
	Bus & coach	17%	13%	14%
	Motorcycles	0.6%	0.5%	0.5%
$PM_{2.5}$	Cars	14%	12%	13%
	LGV	10%	9%	9%
	HGV	2%	3%	3%
	Bus & coach	2%	2%	2%
	Motorcycles	0%	0%	0%

Brussels

5.17 Brussels is the capital of Belgium, with a population in 2019 of 1.2 million in the Brussels Capital Region, or 2.5 million in the wider metropolitan area. It is also part of a larger conurbation, including Ghent, Antwerp Leuven and Walloon Brabant. There are only two roadside monitoring locations in the city, both of which show relatively low concentrations, given the size of the city, in comparison to other cities in Europe.

Baseline scenario

5.18 Appendix A3 shows the measured and BRT-adjusted measured NO_2 and NOx concentrations between January and May 2020 at all monitoring sites in Brussels. Figure 16 shows the NO_2 results averaged across each of the roadside sites, and across each of the urban background sites. The most stringent phase of Covid-19 lockdown appears to have caused a step change in BRT-adjusted concentrations at both roadside and background (urban) sites. A key feature in the raw observed concentrations at all sites is a large spike in NOx and NO_2 concentrations in late January, but the fact that the BRT-adjusted data have removed this spike suggests that it was associated with ‘routine’ factors such as changes to the weather. There are only two roadside monitors in Brussels considered suitable for the analysis and, as shown in Figure 17, both recorded almost identical reductions in traffic-NO_2 during the lockdown (43% reduction at Site 41R002 and 42% reduction at Site 41WOL1). These two site-specific reductions have been averaged as shown in Figure 17 to represent the city-wide effect of the Covid-19 lockdown on traffic-related NO_2 concentrations. The changes for NOx have been calculated in the same way as shown for NO_2.

5.19 It is interesting to note the different pattern shown in Figure 16 for Brussels as compared to Berlin (Figure 10). The Berlin data suggest that most of the variability in roadside concentrations was driven by the local increment over background. The Brussels data suggest that most of the variability at the roadside is driven by variation in the background itself. In this respect, the Brussels data are similar to the other cities analysed, as described later in this section. It is also noted that, in terms of period averages, the background concentrations used for Brussels are not dissimilar to the other cities, including Berlin.

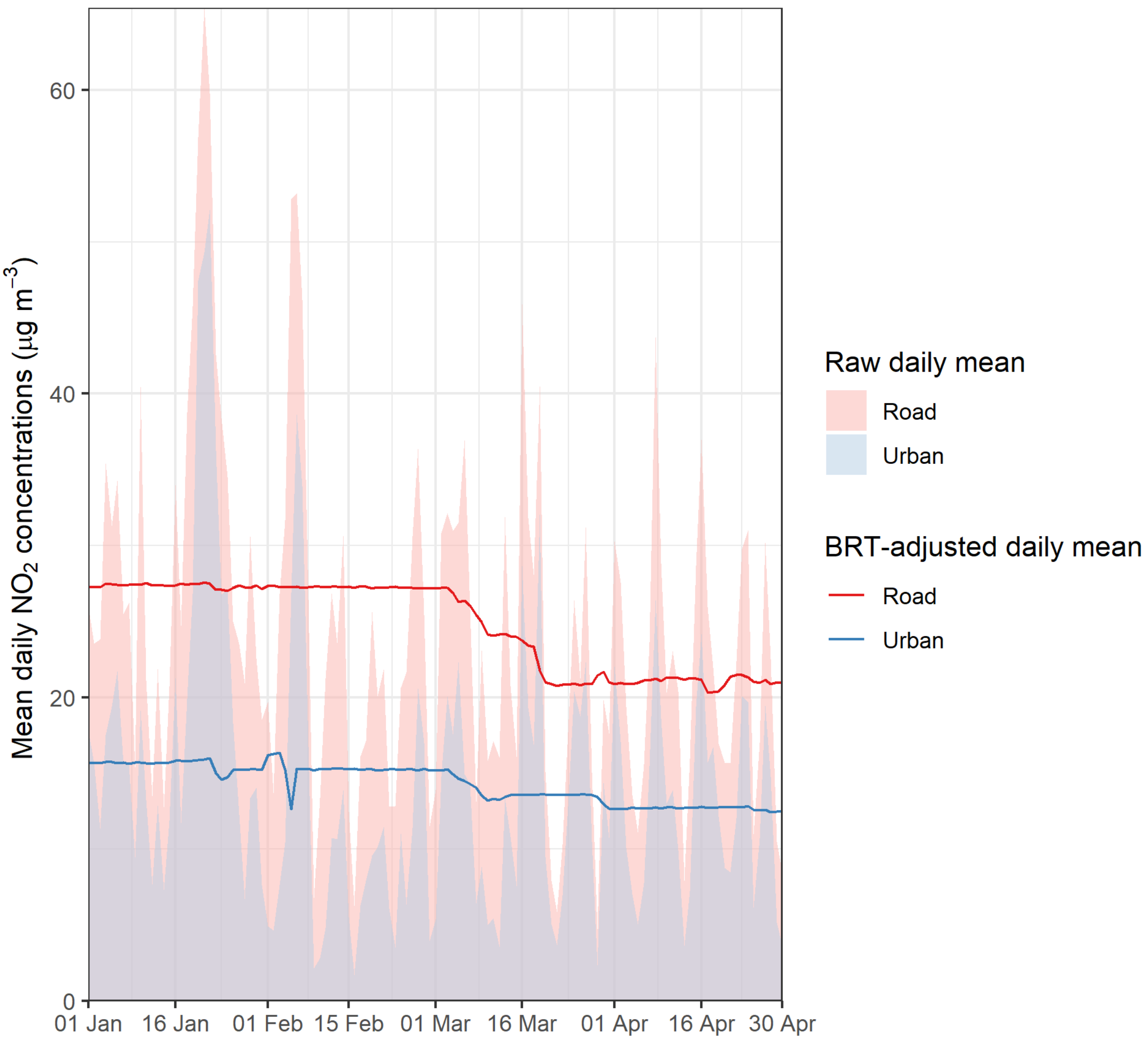

Figure 16: Mean Daily NO_2 Averaged across All Roadside Sites and across All Urban Background Monitoring Sites in Brussels

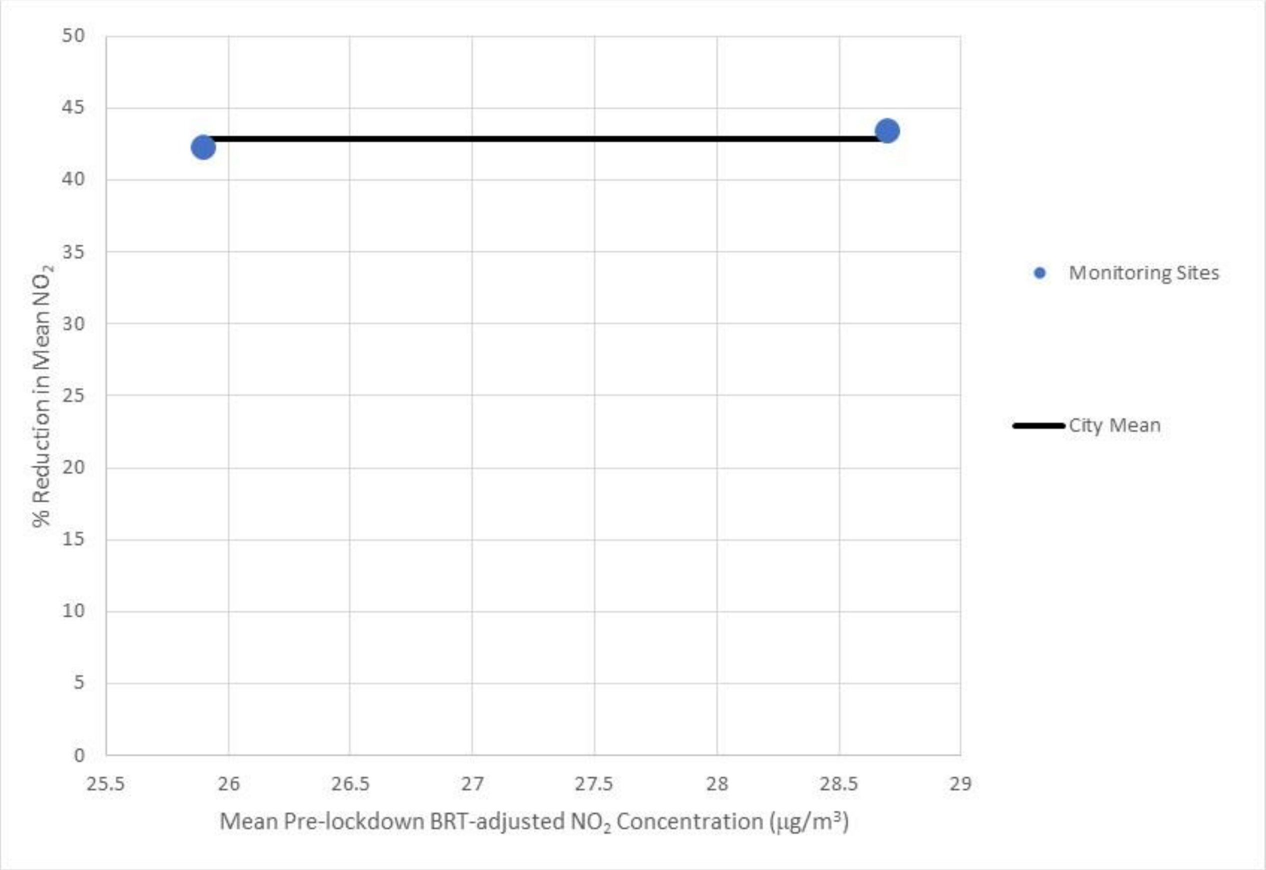

Figure 17: Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Brussels

Table 10: Baseline scenario outputs; Brussels

Pollutant	% reduction in traffic contribution	Mean pre-lockdown concentration (µ/m³)	Mean lockdown concentration (µ/m³)	Mean change (µ/m³)	Roadside sites included
NO_2	43	27.3	21.2	-6.2	2
NOx	35	43.7	35.1	-8.6	

Scenario Analysis

5.20 Table 11 shows that the emission reductions seen during the most stringent phase of lockdown are, theoretically, attainable through the mobility policies represented by the analysis scenarios.

Table 11: Outputs for scenario analyses, showing the conversion of ICE passenger car km to EV to meet target NOx emission reduction; Brussels

Scenario	Reduction in passenger car km
Scenario A, ICE cars to EV	72%
Scenario B, ICE cars, vans and trucks to EV	64%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	47%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	37%
Scenario D, Long term ICE phase out	32%

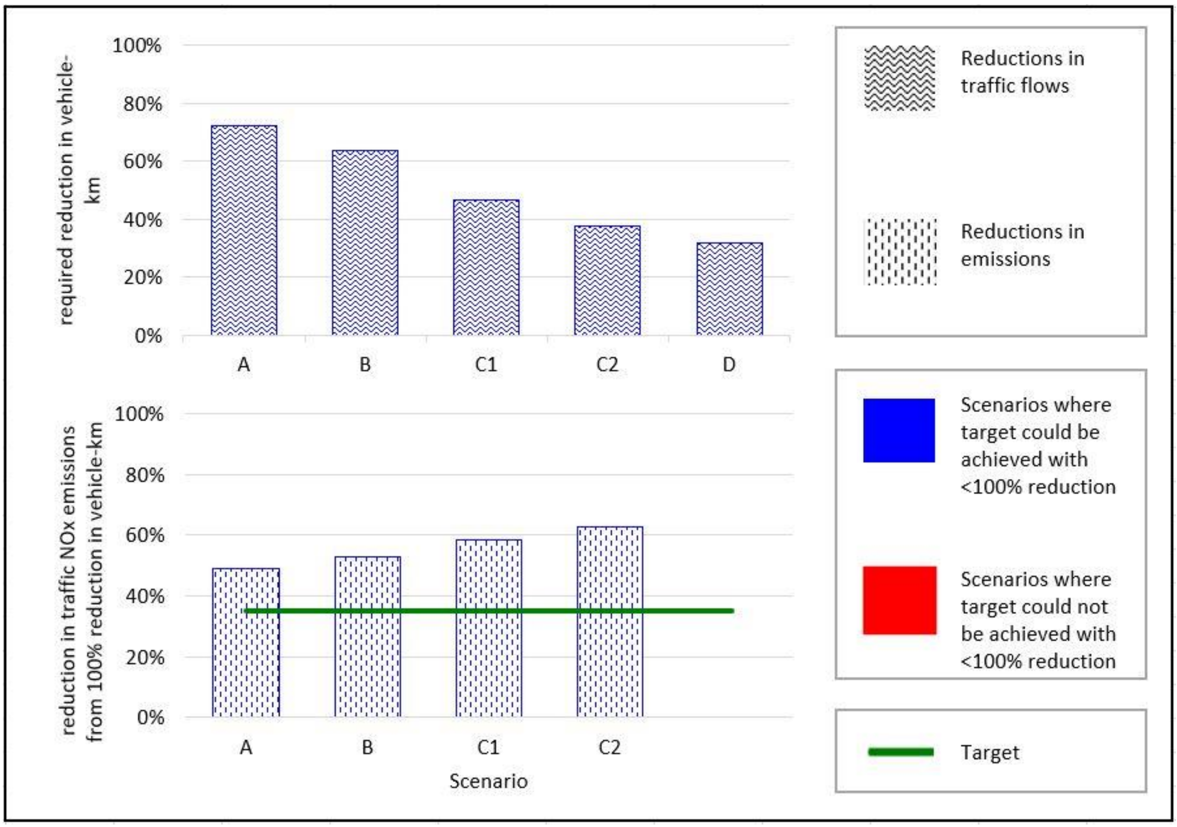

Figure 18: Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Brussels

5.21 Reductions in $PM_{2.5}$ emissions from full implementation of the scenarios are relatively modest and could reflect a cleaner ICE vehicle fleet in current conditions. Given current levels of uptake, not just of EV and hybrid vehicles but higher Euro 6 standard ICE cars, tailpipe emissions from road transport may not be the largest source of $PM_{2.5}$ emissions in many European cities by 2030 and beyond. It is, therefore, not surprising that emission reductions achieved through conversion from ICE to EV may be relatively modest.

Table 12: Outputs for Scenario Analyses, showing traffic $PM_{2.5}$ emission reductions for achieving each scenario; Brussels

Scenario	Reduction in $PM_{2.5}$ emissions, traffic component
Scenario A, ICE cars to EV	23%
Scenario B, ICE cars, vans and trucks to EV	22%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	23%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	22%
Scenario D, Long term ICE phase out	21%

5.22 Table 13 shows the impact of converting each vehicle class, by turn, to EV, and the relative importance on each class in terms of emissions. The relative burden of reducing transport NOx emissions inevitably falls on passenger cars as they make up over half of the total emissions. However, for $PM_{2.5}$, emissions from other source categories, most likely buses, are also likely to be important.

Table 13: Scenario E, class by class 100% switch to EV, emission reductions from the transport component; Brussels

Pollutant		Reduction in transport emissions
NOx	Cars	49%
	LGV	15%
	HGV	26%
	Bus & coach	10%
	Motorcycles	0.1%
$PM_{2.5}$	Cars	33%
	LGV	8%
	HGV	8%
	Bus & coach	3%
	Motorcycles	0.4%

Budapest

5.23 Budapest is the capital of Hungary and its largest city, with a population in 2017 of 1.7 million, rising to 3.3 million within the Budapest metropolitan area (which covers a much larger area than the city limits).

Baseline scenario

5.24 Appendix A3 shows the measured and BRT-adjusted measured NO_2 and NOx concentrations between January and May 2020 in Budapest. The BRT-adjusted concentrations at the background monitor are largely static throughout this period, and apparently unaffected by the most stringent phase of Covid-19 lockdown. The effects of the lockdown are also less evident at the roadside sites than at most sites in other cities. While the three BRT model runs show very similar results to one another, there are several episodes of elevated observed concentrations which cannot be explained; for example, the average BRT-adjusted NO_2 concentration during lockdown at Site Hu6 is affected by elevated, measured concentrations during April. Figure 19 shows the daily mean NO_2 concentrations averaged across both roadside sites, with the results from the background site superimposed.

5.25 The average reductions in traffic-NO_2 during the lockdown at the two roadside sites in Budapest are shown in Figure 20 and Figure 21. There is significant variability when comparing the two sites. Site Hu8, which is to the west of the city, is adjacent to a road which was partially closed during 2019 and this may have affected the ability of the BRT models to normalise the observations. Site Hu6, which is to the east, is set back more than 30 m from the main road (which will reduce the road signal) but close to a number of tram lines (which while not a source of NOx emissions, might cause local NOx emissions to be atypical), and it is possible that this setting has affected the observed reduction during lockdown. Despite the large variability between the two sites, it was considered appropriate to take an average from the two monitors to represent the city-wide improvements (Figure 21); this is because there are no other local empirical data to calculate these improvements. It is, however, clear that the uncertainty around this calculated average will be particularly large for Budapest, because of the variability between the two monitors. The patterns for NOx are very similar to those for NO_2 and have been averaged in the same way, as shown in Figure 21.

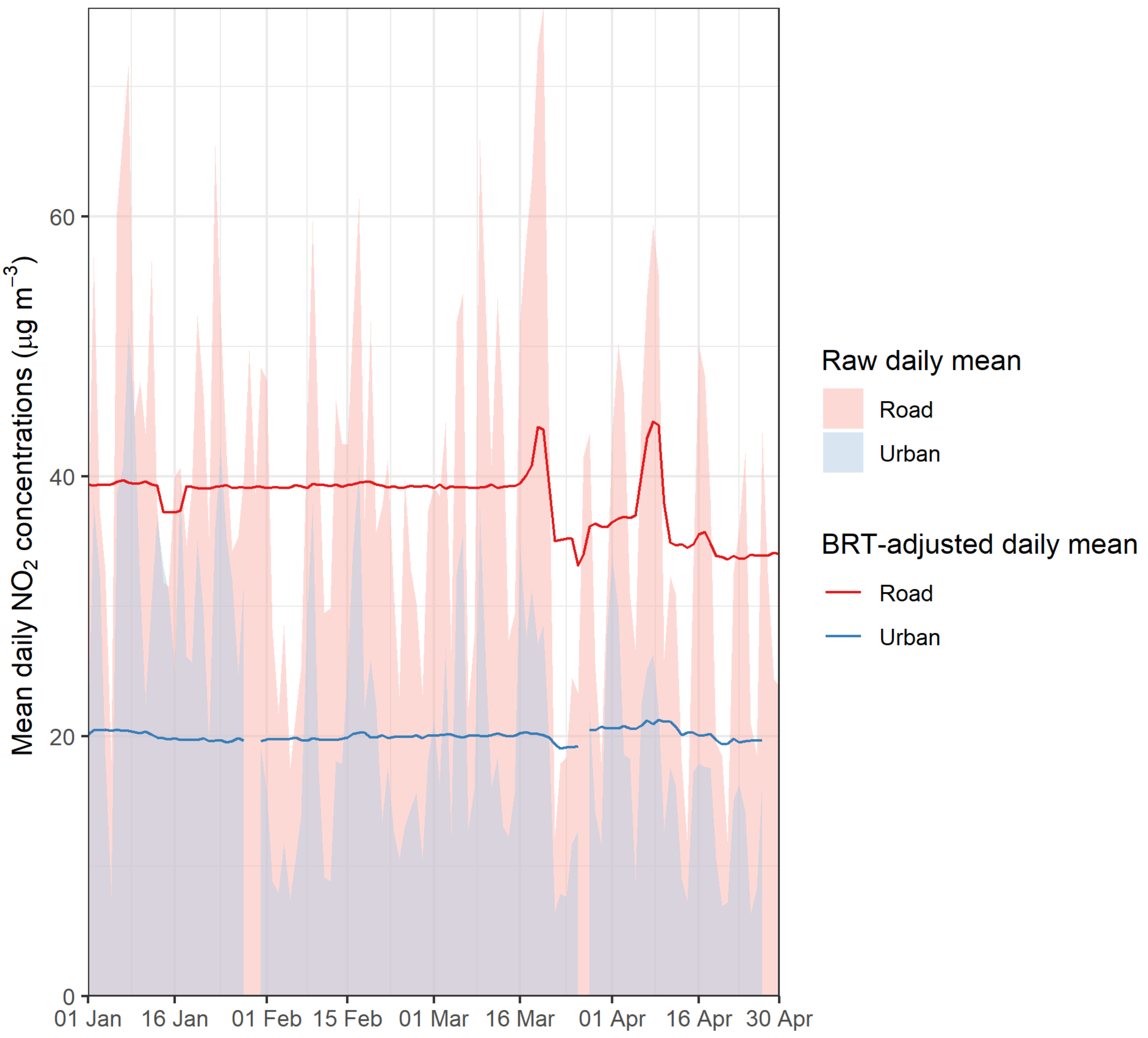

Figure 19: Mean Daily NO$_2$ Averaged across All Roadside Sites and across All Urban Background Monitoring Sites in Budapest

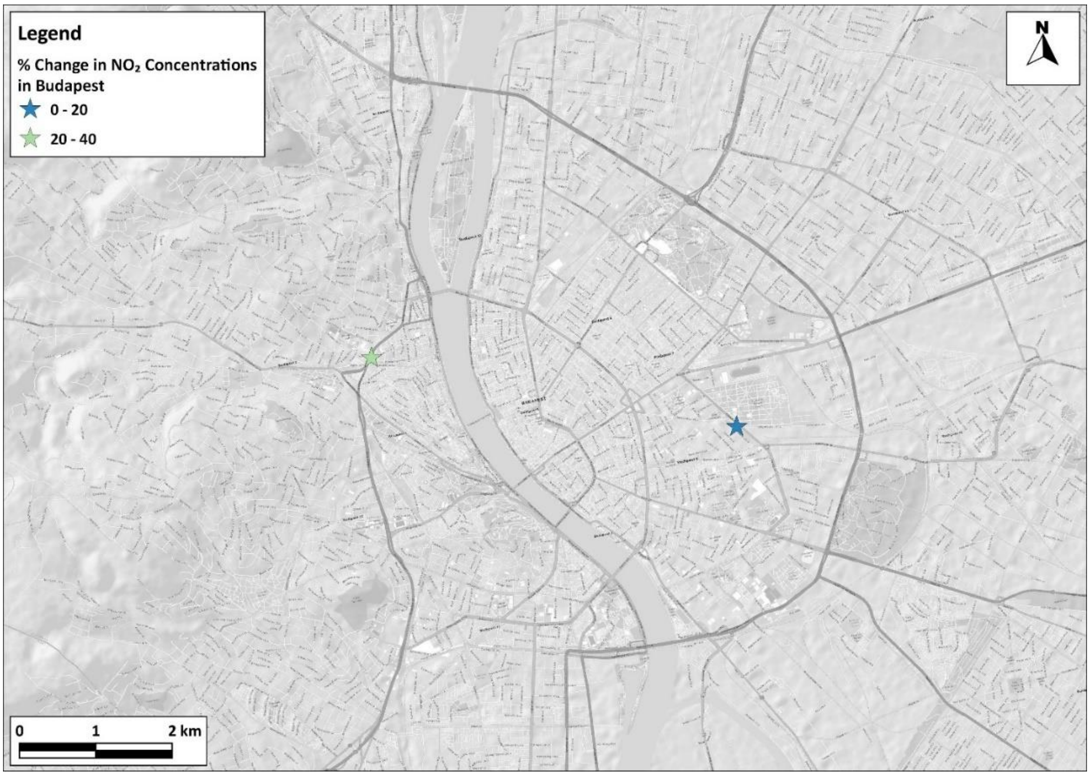

Figure 20: Map of Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Budapest

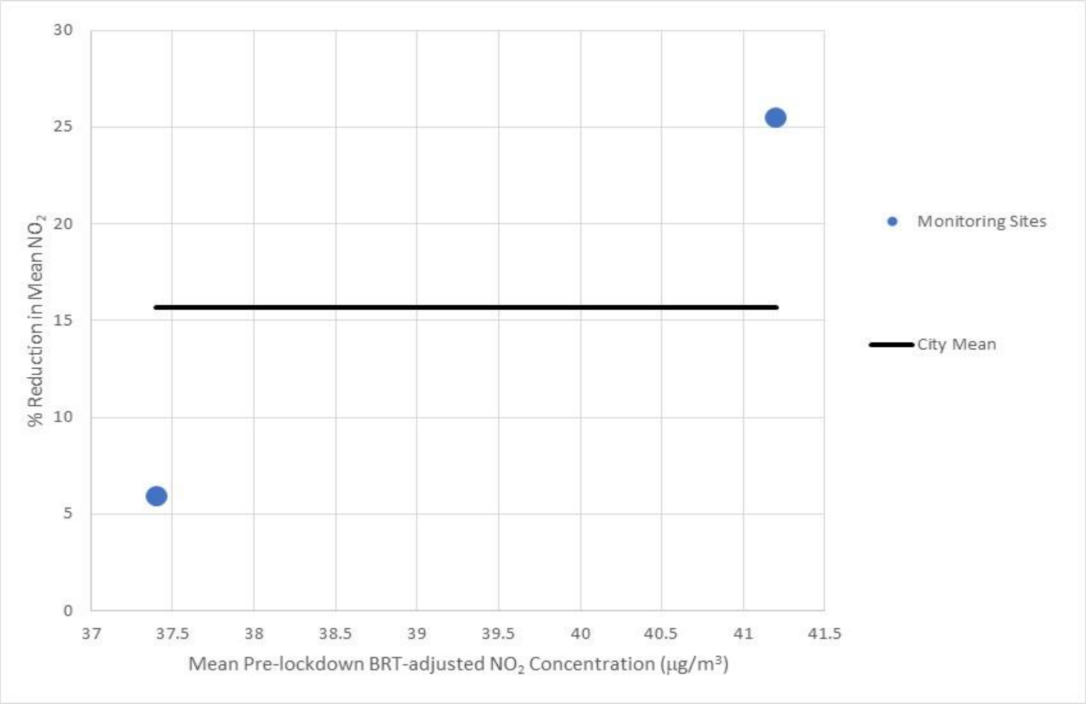

Figure 21: Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Budapest

Table 14: Baseline scenario outputs; Budapest

Pollutant	% reduction in traffic contribution	Mean pre-lockdown concentration (µ/m³)	Mean lockdown concentration (µ/m³)	Mean change (µ/m³)	Roadside sites included
NO_2	16	39.3	36.1	-3.2	2
NOx	14	74.3	67.4	-6.9	

Scenario Analysis

5.26 Table 15 shows that the emission reductions during the most stringent phase of lockdown are, theoretically, attainable through the mobility policies represented by the analysis scenarios. As the mean lockdown reductions were smaller than the other cities analysed, the target reductions are comparatively easier to achieve. For Scenario C2, the reduction can be achieved through conversion of HGVs and LGVs to EV, without the need to convert cars. This shows that reductions in transport NOx emissions, beyond the target reductions, are realistically possible.

Table 15: Outputs for scenario analyses, showing the conversion of ICE passenger car km to EV to meet target NOx emission reduction; Budapest

Scenario	Reduction in passenger car km
Scenario A, ICE cars to EV	42%
Scenario B, ICE cars, vans and trucks to EV	25%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	6%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Reduction in ICE HGV and LGV km required to meet target: 3%
Scenario D, Long term ICE phase out	11%

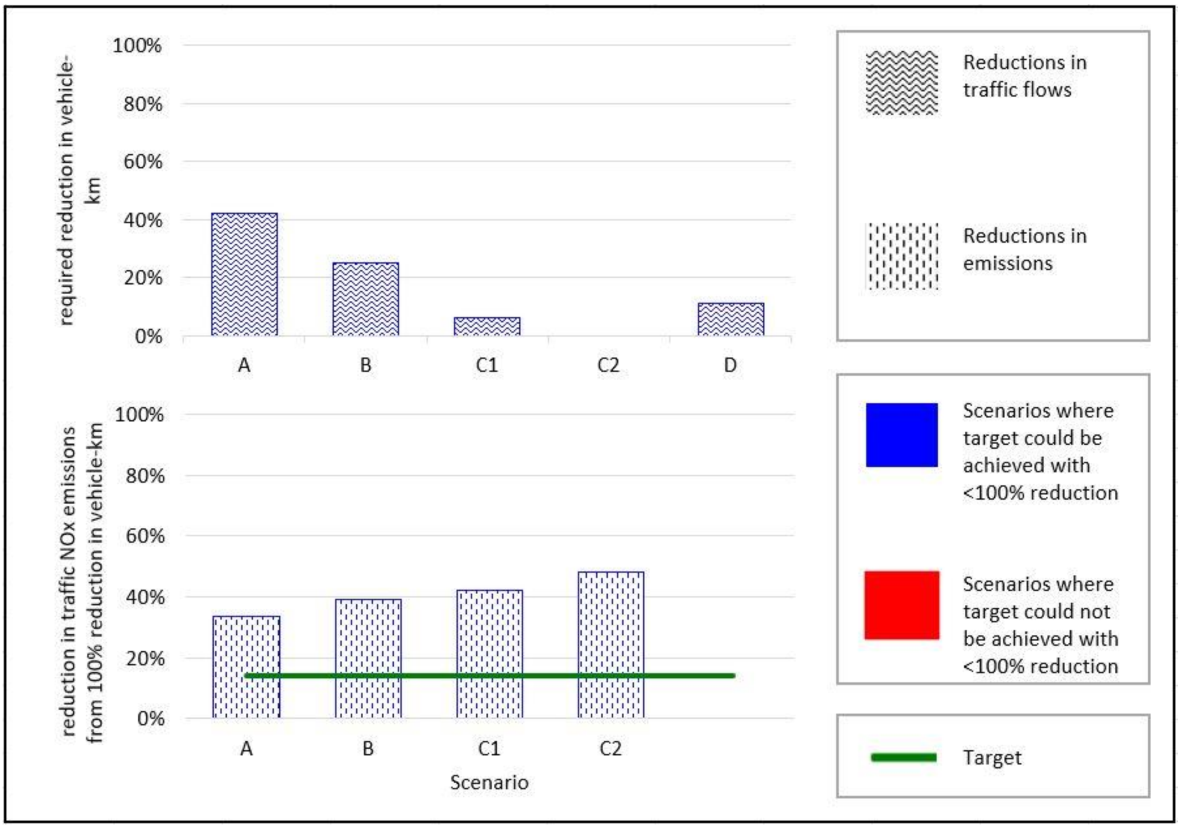

Figure 22: **Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Budapest**

5.27 Reductions in $PM_{2.5}$ emissions from full implementation of the scenarios are relatively modest, possibly reflecting a cleaner ICE vehicle fleet in current conditions. Given current levels of uptake, not just of EV and hybrid vehicles but higher Euro 6 standard ICE cars, tailpipe emissions from road transport may not be the largest source of $PM_{2.5}$ emissions in many European cities by 2030 and beyond. It is, therefore, not surprising that emission reductions achieved through conversion from ICE to EV may be relatively modest.

Table 16: **Outputs for Scenario Analyses, showing traffic $PM_{2.5}$ emission reductions for achieving each scenario; Budapest**

Scenario	Reduction in $PM_{2.5}$ emissions, traffic component
Scenario A, ICE cars to EV	10%
Scenario B, ICE cars, vans and trucks to EV	10%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	8%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	8%
Scenario D, Long term ICE phase out	11%

5.28 Table 17 shows the impact of converting each vehicle class, by turn, to EV, and the relative importance on each class in terms of emissions. The relative burden of reducing transport NOx emissions should be spread across all vehicle classes, as they each make up around one-third of the transport emissions total (if buses are included with HGVs). For $PM_{2.5}$, LGVs become the dominant source, followed by passenger cars. Both of these patterns are significantly different from the other cities analysed, and potentially show why Scenario C2 is so successful.

Table 17: Scenario D, class by class 100% switch to EV, emission reductions from the transport component; Budapest

Pollutant		Reduction in transport emissions
NOx	Cars	33%
	LGV	32%
	HGV	26%
	Bus & coach	9%
	Motorcycles	0.4%
PM2.5	Cars	25%
	LGV	31%
	HGV	7%
	Bus & coach	2%
	Motorcycles	0.8%

London

5.29 London is the capital city of the United Kingdom and also its largest city, with an estimated population in 2018 of 8.9 million. It has an extensive network of air quality monitoring stations with annual average NO_2 concentrations at roadside locations ranging from 23 to 68 $\mu g/m^3$. It has a city-wide Low Emission Zone and an Ultra-Low Emission Zone in the central area.

Baseline scenario

5.30 Appendix A3 shows the measured and BRT-adjusted measured NO_2 and NOx concentrations between January and May 2020 in London. The time series of raw (observed) data show similar sequences of peaks and troughs across most of the sites, reflecting regional pollution episodes. These short-term variations are mostly removed in the BRT-adjusted time series. This means that at many of the background sites which are relatively unaffected by road traffic, the BRT-adjusted data show a largely flat time series for 2020. In many cases, there appears to be some deviation in the BRT-adjusted NO_2 time series at background sites, but not in the equivalent NOx data. This is an interesting feature that warrants further investigation. In particular, it is noted that these differences are greater if O_3 is not included as a comparator variable within the BRT model.

5.31 Most of the roadside sites in London shown in Appendix A3 show step changes in BRT-adjusted concentrations coinciding approximately with the start of lockdown. Precise patterns, both in the scale of change and timing, are different at different sites. This is likely to be a combined effect of differences in activity patterns and also artefacts introduced by the machine learning algorithms. The large number of monitoring sites available in London, when compared to other cities, provides confidence in the overall means, but also highlights the limitations of relying on smaller numbers of sites. Had a small subset of the London sites been used then the overall patterns would have been the same, but the precise details would have been different.

5.32 Figure 23 shows the daily mean NO_2 concentrations averaged across all roadside sites, and across all urban background monitoring sites in London. It also shows the average of all daily mean BRT-adjusted concentrations. Similar to Berlin, it shows that, taken on aggregate, there was a small reduction in concentrations measured at background sites which could not be explained by normalising for weather (etc.) causing a small downward trend in the BRT-adjusted data for urban background sites. Similar to the other cities, there is a clear step change toward lower concentrations at roadside sites which occurred during March.

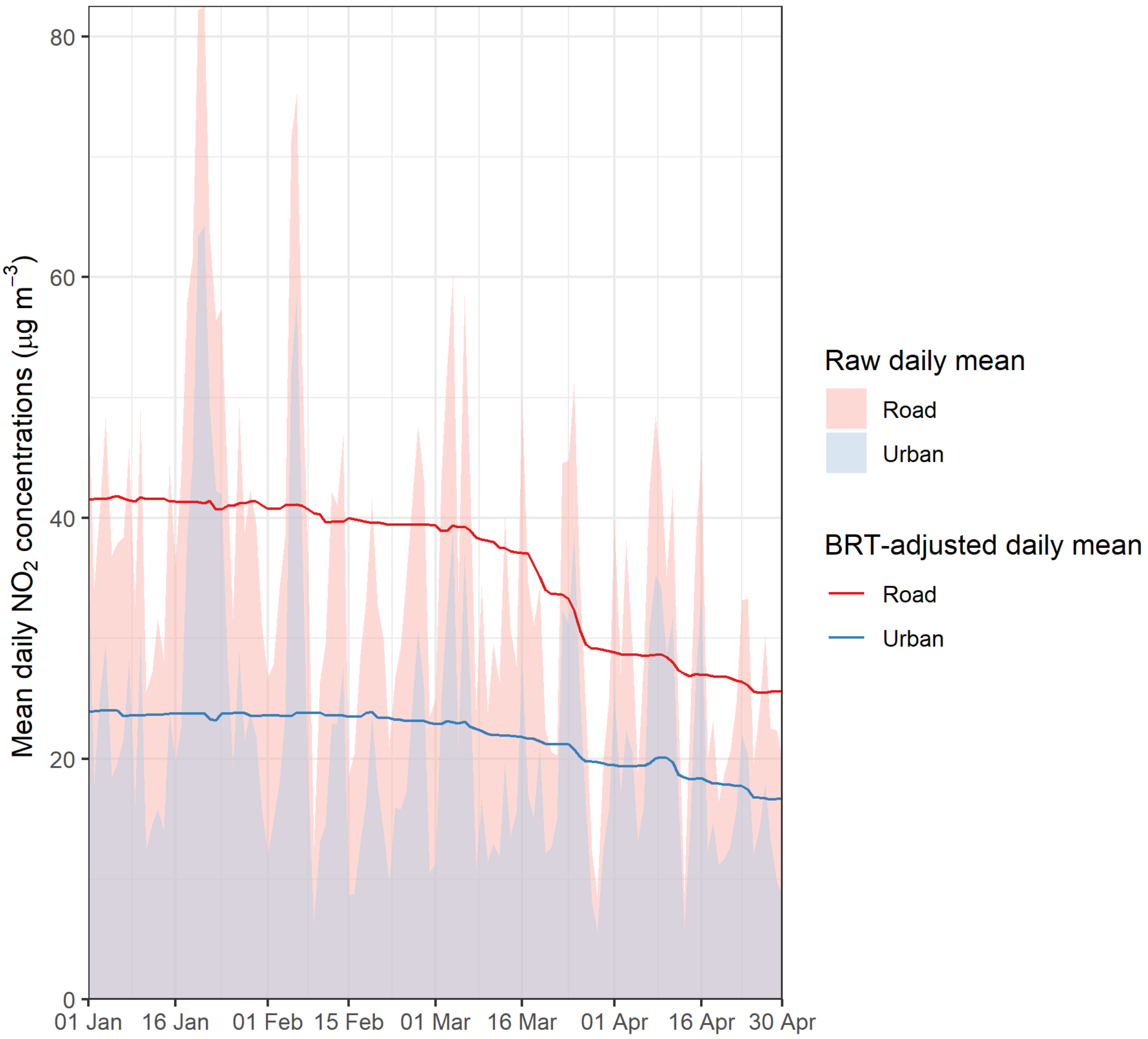

Figure 23: Mean Daily NO_2 Averaged across All Roadside Sites and across All Urban Background Monitoring Sites in London

5.33 There are fifty-one roadside sites in London considered suitable for use in this analysis. As for the other cities, the period averages before and during the most stringent phase of lockdown have been calculated across all three BRT model runs. Equivalent non-traffic background values have then been subtracted to show the change in the road increment. The relative changes in these road increments to NO_2 concentrations are shown in Figure 24. Spatial patterns in these calculated changes are less evident than seen in Berlin and Paris, but there does seem to be a general trend of larger changes toward the centre of the city and smaller changes in outer areas. These data are also shown in Figure 25, with monitoring sites within central London shown separately. The pattern appears to be that sites within central London consistently recorded relatively high (>60%) reductions in traffic-related NO_2. Some sites in outer areas have also seen equivalent, or even larger reductions, but these are relatively isolated, and at most sites outside of the centre the reductions have been less than 50%.

5.34 As with Berlin, the average reductions within central London and outer London have been calculated separately, as shown in Figure 25. The overall patterns for NOx are, predictably, very similar to those shown for NO_2 and are not shown separately. Averages have been calculated for NOx in the same way as shown for NO_2.

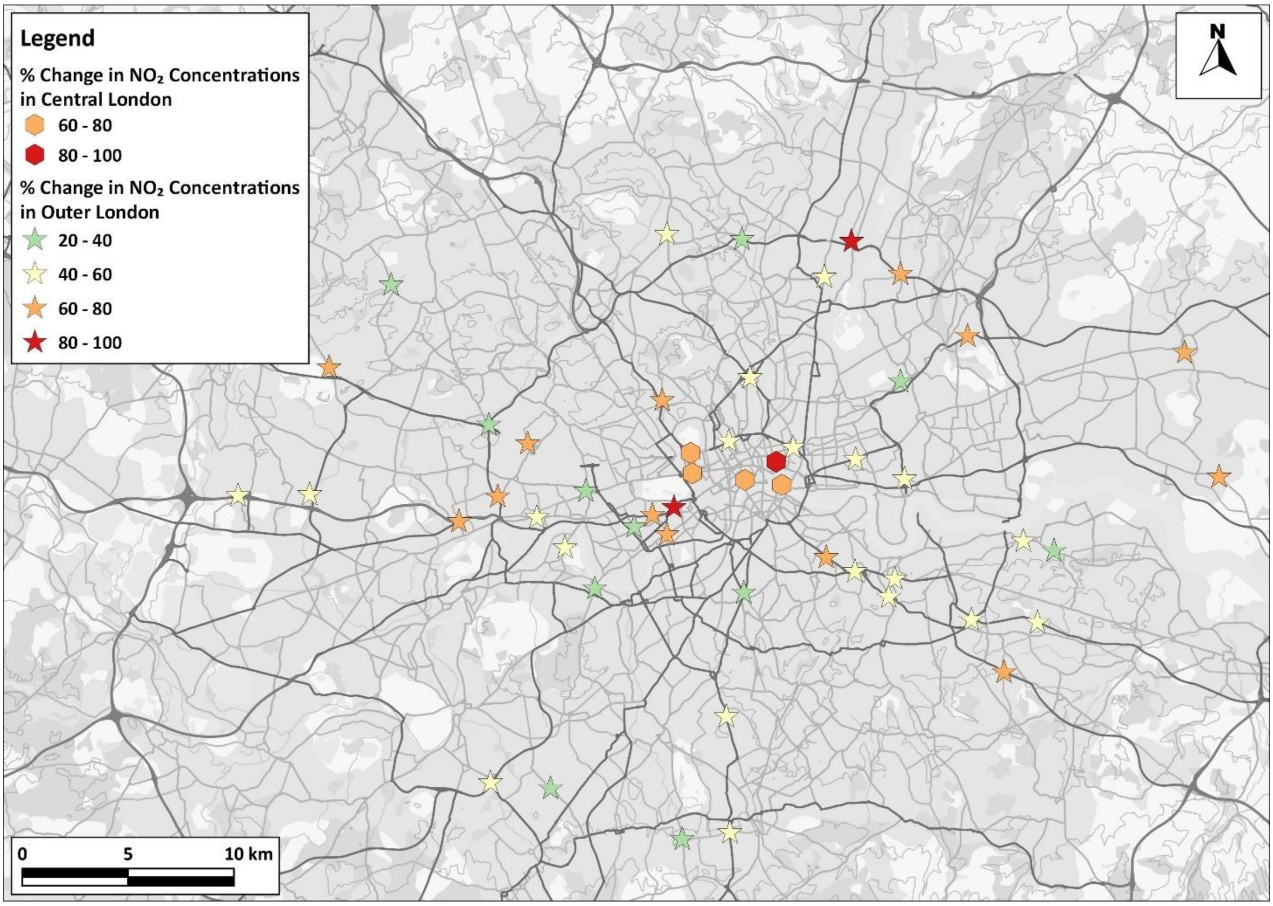

Figure 24: **Map of Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in London**

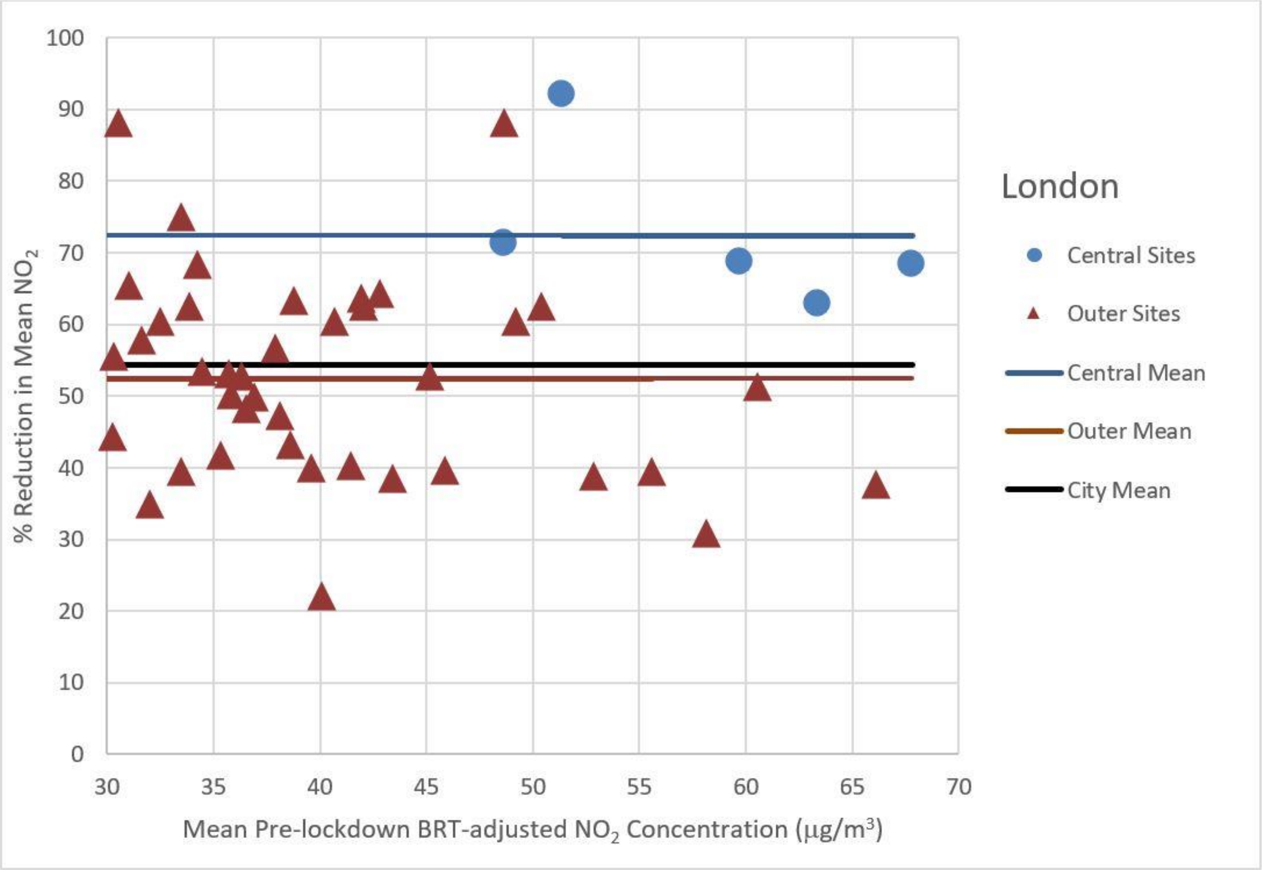

Figure 25: Relative Reductions in Mean Traffic-related NO2 During the Most Stringent Phase of Lockdown in London

Table 18: Baseline scenario outputs; London

	Zone	% reduction in traffic contribution	Mean pre-lockdown concentration (µ/m^3)	Mean lockdown concentration (µ/m^3)	Mean change (µ/m^3)	Roadside sites included
NO$_2$	Inner	72	58.2	31.0	-27.3	5
	Outer	52	38.8	27.7	-11.1	46
	City-wide	54	40.7	28.0	-12.7	51
NOx	Inner	66	129.1	63.6	-65.5	5
	Outer	47	79.9	54.8	-25.1	46
	City-wide	49	84.8	55.7	-29.1	51

Scenario Analysis

5.35 Table 19 shows that the emission reductions seen during the most stringent phase of lockdown are, theoretically, attainable through the mobility policies represented by the analysis scenarios, other than for the central area, although the levels of conversion needed are high. In the central area, the level of traffic reduction during lockdown was very high and, because of the ULEZ, the fleet is relatively clean. This means that the target is high, and the emissions reduction per vehicle converted is low. Thus, the targets are unattainable under all scenarios except D, where HGVs and LGVs make an equivalent proportionate contribution to the emission savings.

Table 19: Outputs for scenario analyses, showing the conversion of ICE passenger car km to EV to meet target NOx emission reduction; London

Scenario	Zone	Reduction in passenger car km	NOx emission reduction[15]
Scenario A, ICE cars to EV	Inner	>100%	27%
	Outer	87%	
	City-wide	92%	
Scenario B, ICE cars, vans and trucks to EV	Inner	>100%	30%
	Outer	81%	
	City-wide	86%	
Scenario C1, ICE cars to EV and non-transport or non-emission transport	Inner	>100%	42%
	Outer	69%	
	City-wide	74%	
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Inner	>100%	45%
	Outer	62%	
	City-wide	67%	
Scenario D, Long term ICE phase out	Inner	65%	
	Outer	44%	
	City-wide	46%	

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[15] Reduction in the traffic component of NOx emissions, if 100% of ICE cars are converted to EV, where greater that 100% would be required to meet the baseline scenario reductions.

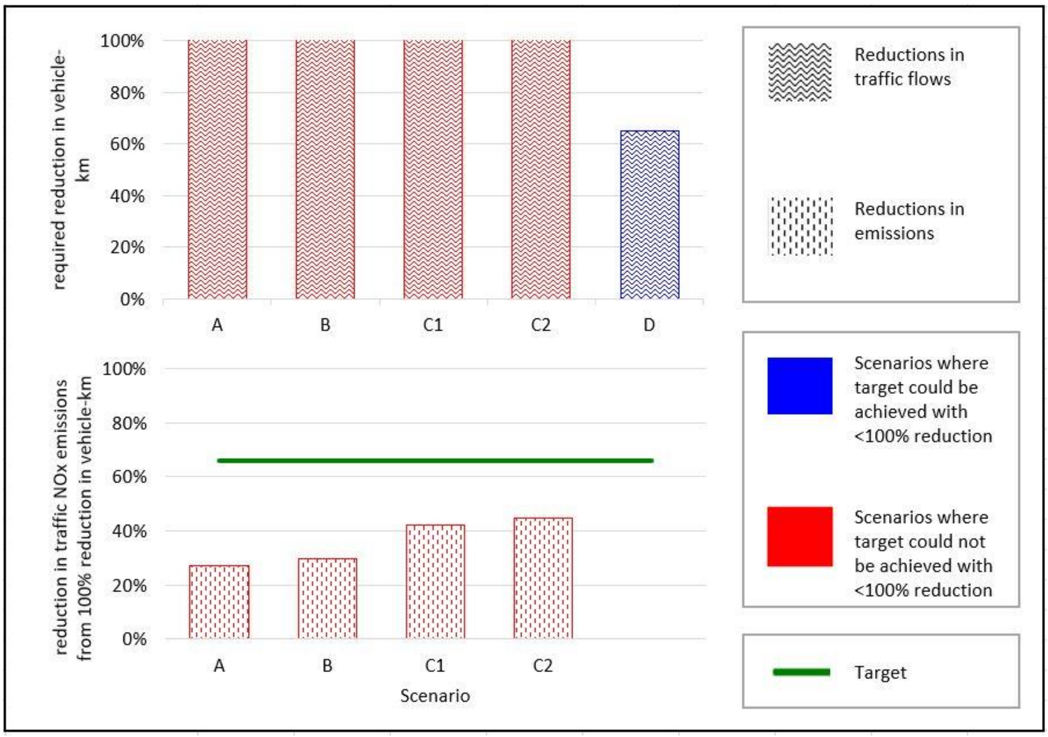

Figure 26: Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Central London

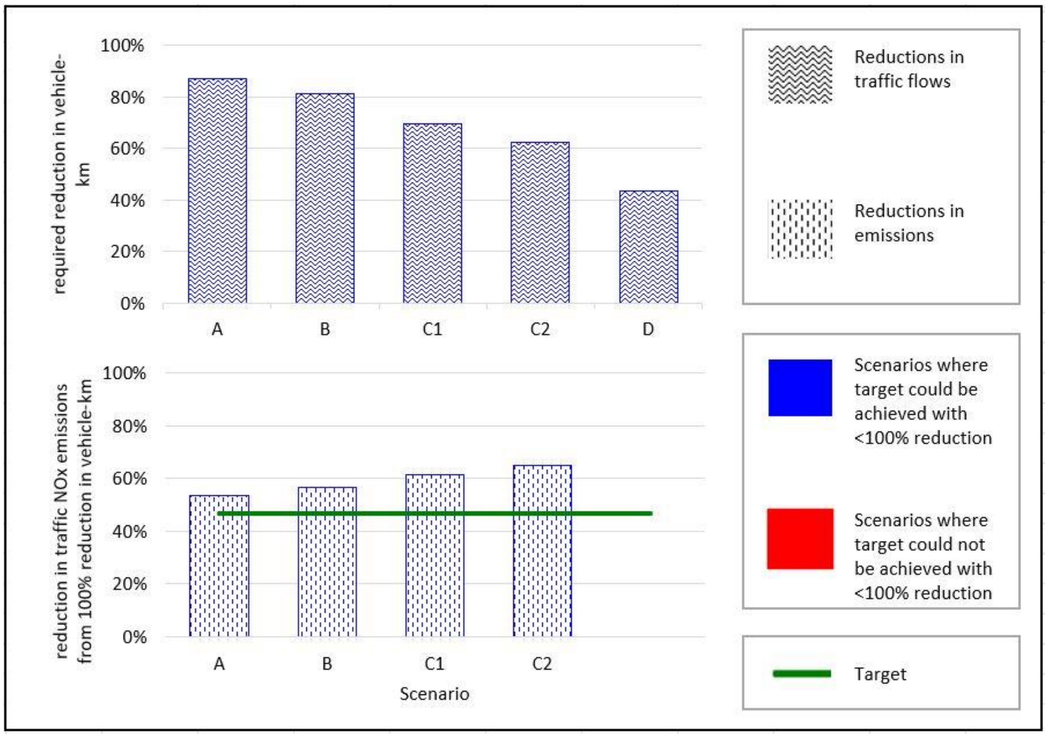

Figure 27: Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Outer London

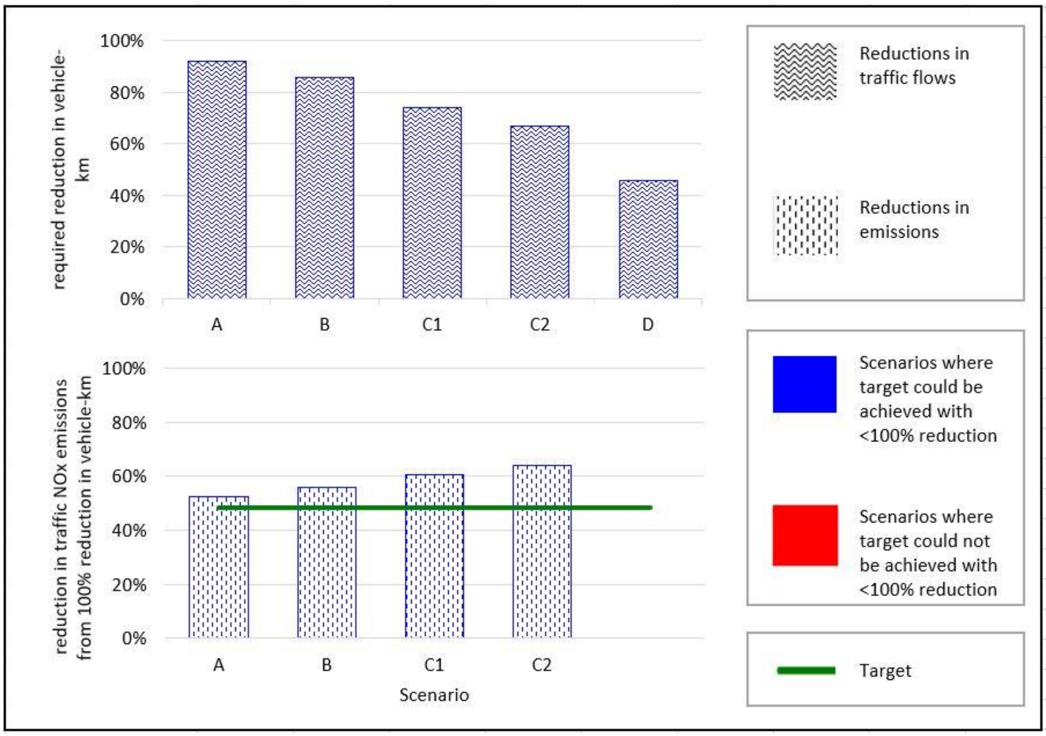

Figure 28: **Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; London City-wide**

5.36 Reductions in $PM_{2.5}$ emissions from full implementation of the scenarios are relatively modest and probably reflect a cleaner ICE vehicle fleet in current conditions. Given current levels of uptake, not just of EV and hybrid vehicles but higher Euro 6 standard ICE cars, tailpipe emissions from road transport may not be the largest source of $PM_{2.5}$ emissions in many European cities by 2030 and beyond. It is, therefore, not surprising that emission reductions achieved through conversion from ICE to EV may be relatively modest.

5.37 Table 21 shows the emissions impact of converting each vehicle class, by turn, to EV, and shows the relative importance on each class in terms of emissions. From this it can be seen that the relative burden of reducing transport NOx emissions inevitably fall on passenger cars as they make up over half of the total emissions.

Table 20: Outputs for Scenario Analyses, showing traffic $PM_{2.5}$ emission reductions for achieving each scenario; London

Scenario	Zone	Reduction in $PM_{2.5}$ emissions, traffic component
Scenario A, ICE cars to EV	Inner	13%
	Outer	12%
	City-wide	13%
Scenario B, ICE cars, vans and trucks to EV	Inner	13%
	Outer	12%
	City-wide	12%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	Inner	15%
	Outer	16%
	City-wide	17%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Inner	15%
	Outer	16%
	City-wide	16%
Scenario D, Long term ICE phase out	Inner	17%
	Outer	15%
	City-wide	16%

Table 21: Scenario E, class by class 100% switch to EV, emission reductions from the transport component; London

Pollutant		Central	Outer	City-wide
NOx	Cars	27%	53%	53%
	LGV	20%	22%	22%
	HGV	8%	12%	12%
	Bus & coach	15%	8%	8%
	Motorcycles	2%	0.6%	0.6%
	London Taxis	28%	5%	5%
$PM_{2.5}$	Cars	5%	14%	14%
	LGV	2%	3%	3%
	HGV	1%	2%	2%
	Bus & coach	2%	1%	1%
	Motorcycles	2%	0.6%	0.6%
	London Taxis	9%	1%	1%

Madrid

5.38 Madrid is the largest city in Spain, as well as its capital, and is the second largest in the EU, after Berlin (based on the defined city limits). Its population in 2018 was 3.2 million, rising to 6.5 million in the wider metropolitan area. A low emission zone has operated in the city centre since 2018, with a proposed phase out of all diesel-powered vehicles within the next decade.

Baseline scenario

5.39 Appendix A3 shows the measured and BRT-adjusted measured NO_2 and NOx concentrations between January and May 2020 in Madrid. There is a clear step-change in BRT-adjusted concentrations coinciding approximately with the start of lockdown at all of the sites; including the background (urban) site. The raw (observed) time series are also very simiar for all sites, showing elevated concentrations in January and late February, and consistently low concentrations during lockdown.

5.40 Figure 29 shows the daily mean NO_2 averaged across all of the sites in Appendix A3, separated into roadside and background. As with most of the other cities, much of the variabilty in the roadside observations is driven by changes in the background field. Figure 29 also highlights that, for Madrid, the effect of lockdown was seen not only at the roadside but also at background monitors.

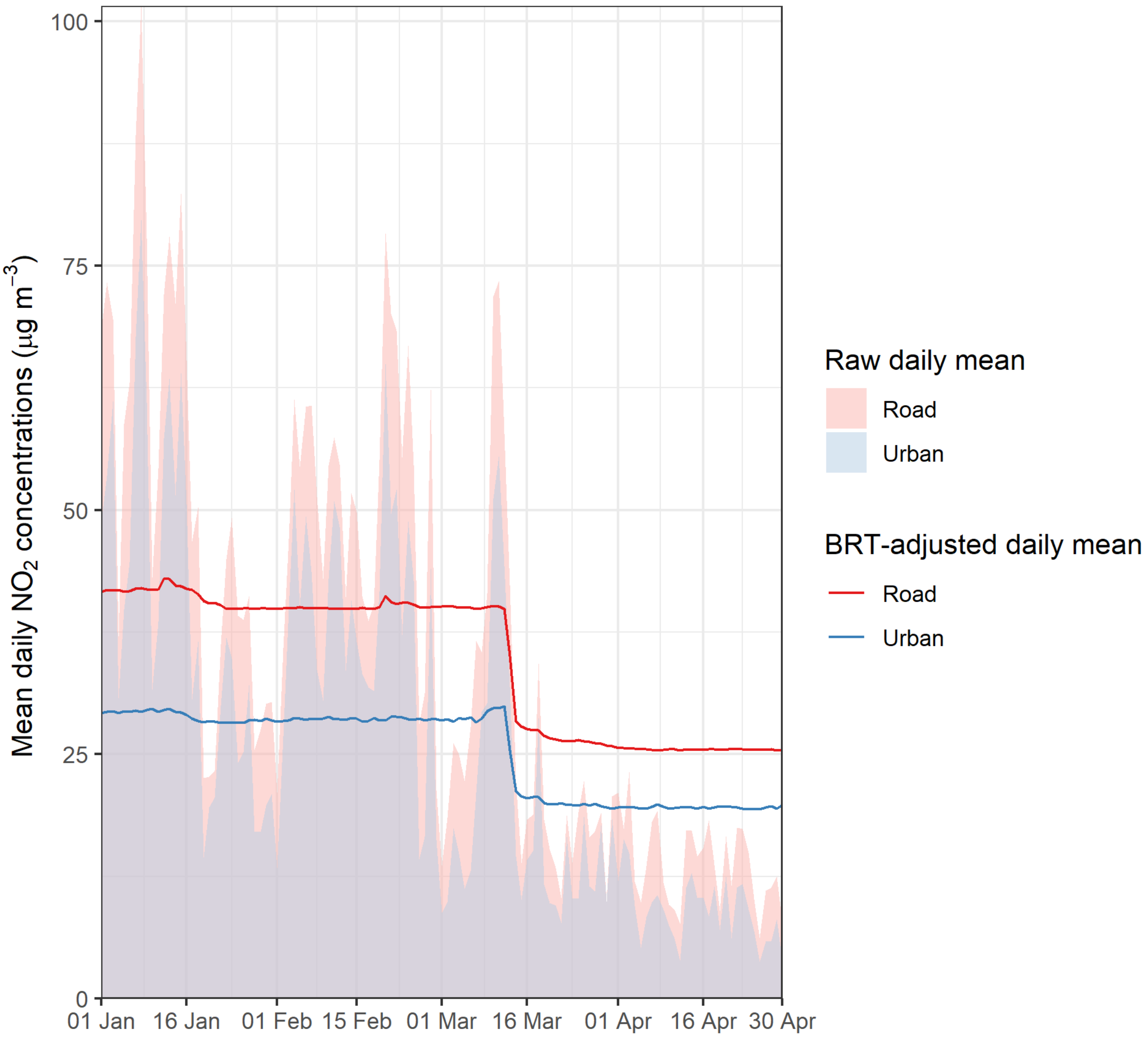

Figure 29: Mean Daily NO_2 Averaged across All Roadside and Urban Background Monitoring Sites in Madrid

5.41 As for the other cities, the site-specific period averages, before and during the most stringent phase of lockdown, have been calculated across all three BRT model runs. The during-lockdown background values have then been subtracted to show the change in the road increment. The relative changes in the road contribution to NO_2 concentrations during the lockdown at the eight suitable roadside sites are shown in Figure 30 and Figure 31.

5.42 All of the calculated reductions fall within the range of 50-86%, but the majority fall within a much narrower range (66% to 73%). Outside of this range are two sites (es0118a and es1940a) which showed reductions greater than 80%, and one (es1943a) which showed a reduction of less than 55%. Of the two sites showing notably large changes, one (ex0118a) is the most central, and is directly between the Calle de O'Donnell and Calle de Alcala, both of which are major arterial routes into the centre of the city. The large reductions seen here may fit with the patterns seen in Berlin, London and Paris, related to larger relative improvements in the city centre. The other site which reported a change greater than 80% is Site es1940a, which is beside the Plaza Castilla and Paseo de la Castellana, and which is another key route into the city centre. The site which showed an atypically small reduction (es1943a) is within a car park close to a station on the Madrid Metro (Plaza Eliptica), but there is no obvious reason for the discrepancy. There is no justification for calculating reductions for different zones in the city, and all of the changes seen at roadside sites have been averaged, as shown in Figure 31. The same calculations have then also been carried out for NOx, which showed very similar spatial patterns.

Figure 30: Map of Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Madrid

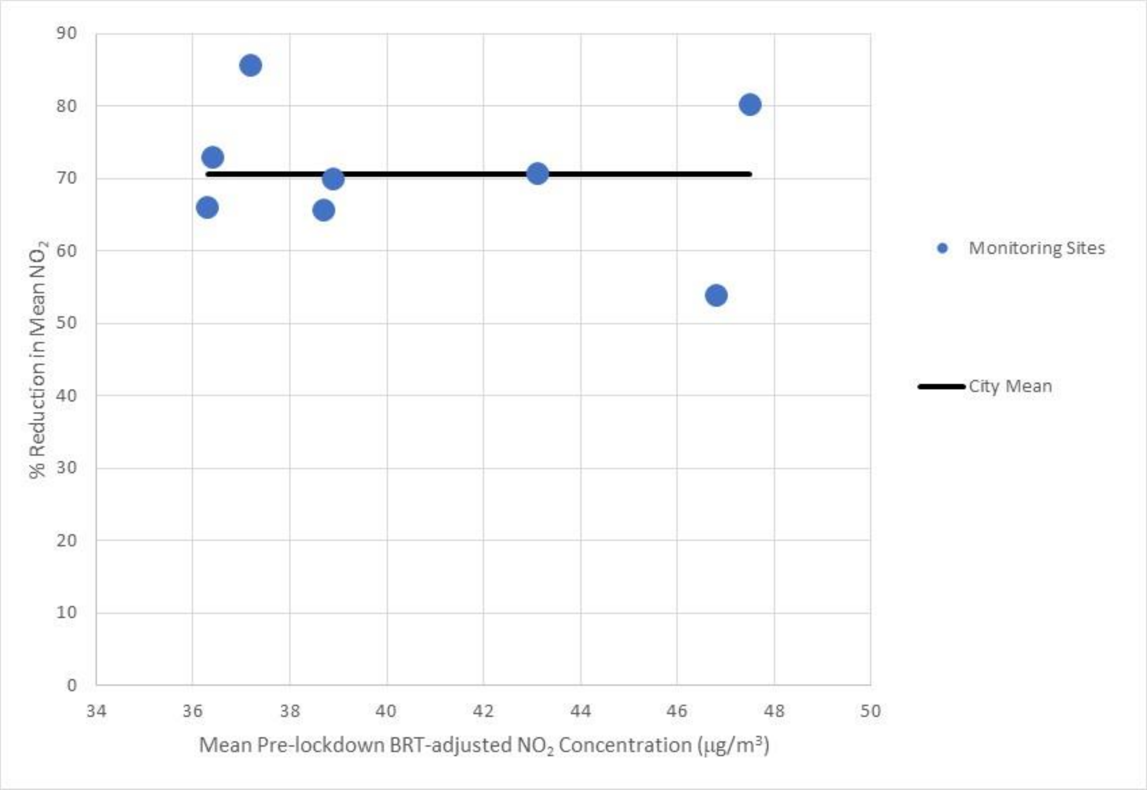

Figure 31: **Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Madrid**

Table 22: Baseline scenario outputs; Madrid

Pollutant	% reduction in traffic contribution	Mean pre-lockdown concentration (µ/m³)	Mean lockdown concentration (µ/m³)	Mean change (µ/m³)	Roadside sites included
NO_2	71	40.6	26.0	-14.6	8
NOx	65	76.9	50.2	-26.7	

Scenario Analysis

5.43 Table 23 shows that the emission reductions during the most stringent phase of lockdown are not attainable, other than for scenario C2 and, even then, the level of conversion required approaches 100%. This reflects the large reduction in emissions during the lockdown period, making the target harder to replicate through fleet and transport changes. Unfortunately, using the data available, the analysis was not able to differentiate between the central and outer areas of the city and does not show the effect of the low emission zone on either lockdown emission reductions or their attainment through scenario analysis.

Table 23: Outputs for scenario analyses, showing the conversion of ICE passenger car km to EV to meet target NOx emission reduction; Madrid

Scenario	Reduction in passenger car km	NOx emission reduction[16]
Scenario A, ICE cars to EV	>100%	55%
Scenario B, ICE cars, vans and trucks to EV	>100%	58%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	>100%	65%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	94%	
Scenario D, Long term ICE phase out	63%	

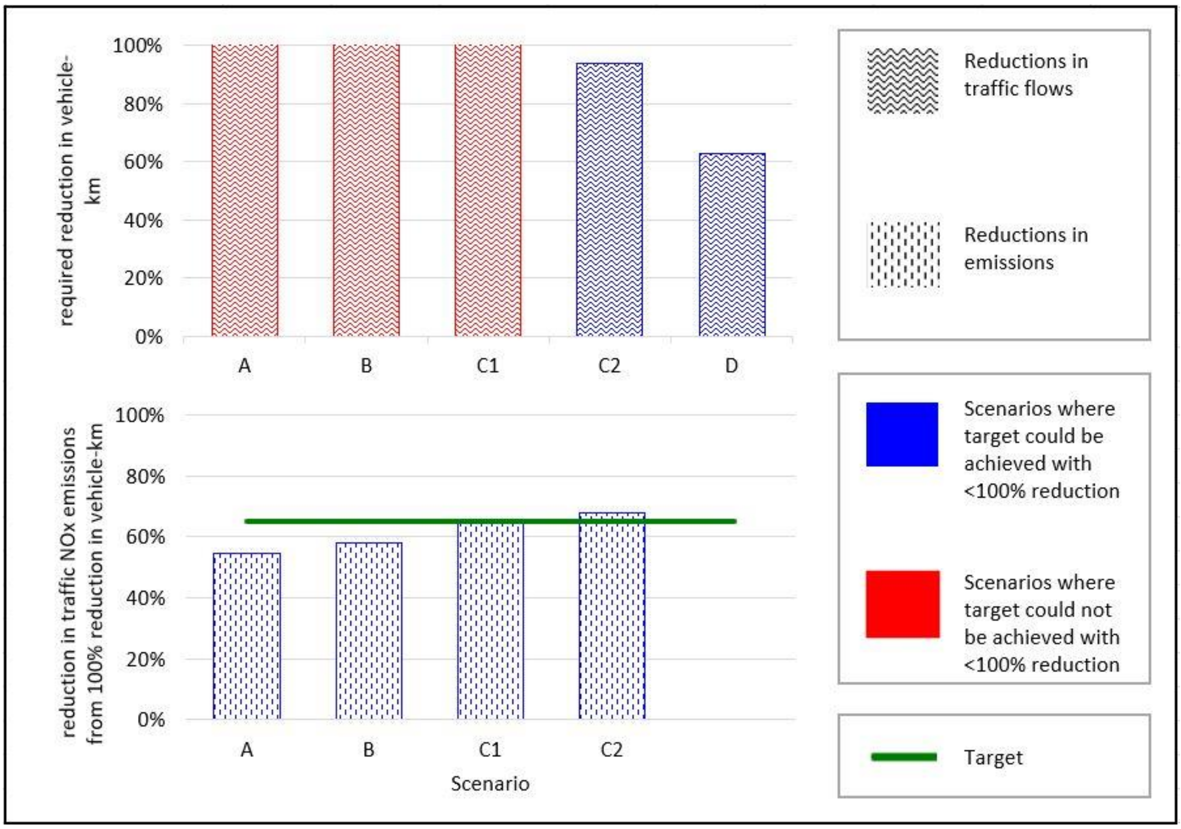

Figure 32: Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Madrid

5.44 Reductions in $PM_{2.5}$ emissions from full implementation of the scenarios are relatively modest and probably reflect a cleaner ICE vehicle fleet in current conditions. Given current levels of uptake, not

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[16] Reduction in the traffic component of NOx emissions, if 100% of ICE cars are converted to EV, where greater that 100% would be required to meet the baseline scenario reductions.

just of EV and hybrid vehicles but higher Euro 6 standard ICE cars, tailpipe emissions from road transport may not be the largest source of $PM_{2.5}$ emissions in many European cities by 2030 and beyond. Consequently, emission reductions achieved through conversion from ICE to EV may be relatively modest.

Table 24: Outputs for Scenario Analyses, showing traffic $PM_{2.5}$ emission reductions for achieving each scenario; Madrid

Scenario	Reduction in $PM_{2.5}$ emissions, traffic component [17]
Scenario A, ICE cars to EV	39%
Scenario B, ICE cars, vans and trucks to EV	41%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	44%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	44%
Scenario D, Long term ICE phase out	42%

5.45 Table 25 shows the emissions impact of converting each vehicle class, by turn, to EV, and shows the relative importance of each class in terms of emissions. The relative burden of reducing transport NOx emissions inevitably falls on passenger cars as they make up over half of the total emissions. However, for $PM_{2.5}$, LGVs become more important.

Table 25: Scenario E, class by class 100% switch to EV, emission reductions from the transport component; Madrid

Pollutant		Reduction in transport emissions
NOx	Cars	55%
	LGV	13%
	HGV	21%
	Bus & coach	10%
	Motorcycles	1%
$PM_{2.5}$	Cars	39%
	LGV	10%
	HGV	6%
	Bus & coach	3%
	Motorcycles	3%

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[17] Reduction in traffic component $PM_{2.5}$ emissions following 100% conversion is undertaken, where >100% is indicated in the previous table.

Paris

5.46 Paris is the capital of France and the largest city with a population of 2.1 million; the Paris region, the Ile de France, has a population of 12.3 million (in 2020). Paris has operated a low emission zone since 2015, covering the whole of the city, with less stringent controls covering the Greater Paris area. As with Berlin, the lowest concentrations of NO_2 are now routinely measured in the centre.

Baseline scenario

5.47 Appendix A3 shows the raw and BRT-adjusted NO_2 and NOx concentrations measured between January and May 2020 in Paris. Many of the sites show an episode of elevated NOx concentrations in late January which may be related to the regional episode observed in the London. As with other cites, these short-term temporal variations can mostly be removed using the BRT adjustment. Figure 29 shows the daily mean NO_2 concentrations averaged across all roadside and urban background monitoring sites. A step change in concentrations which coincides approximately with the start of the lockdown is clear in the roadside data. A smaller step change is also seen in the urban background data, inferring that these sites are still affected by traffic emissions.

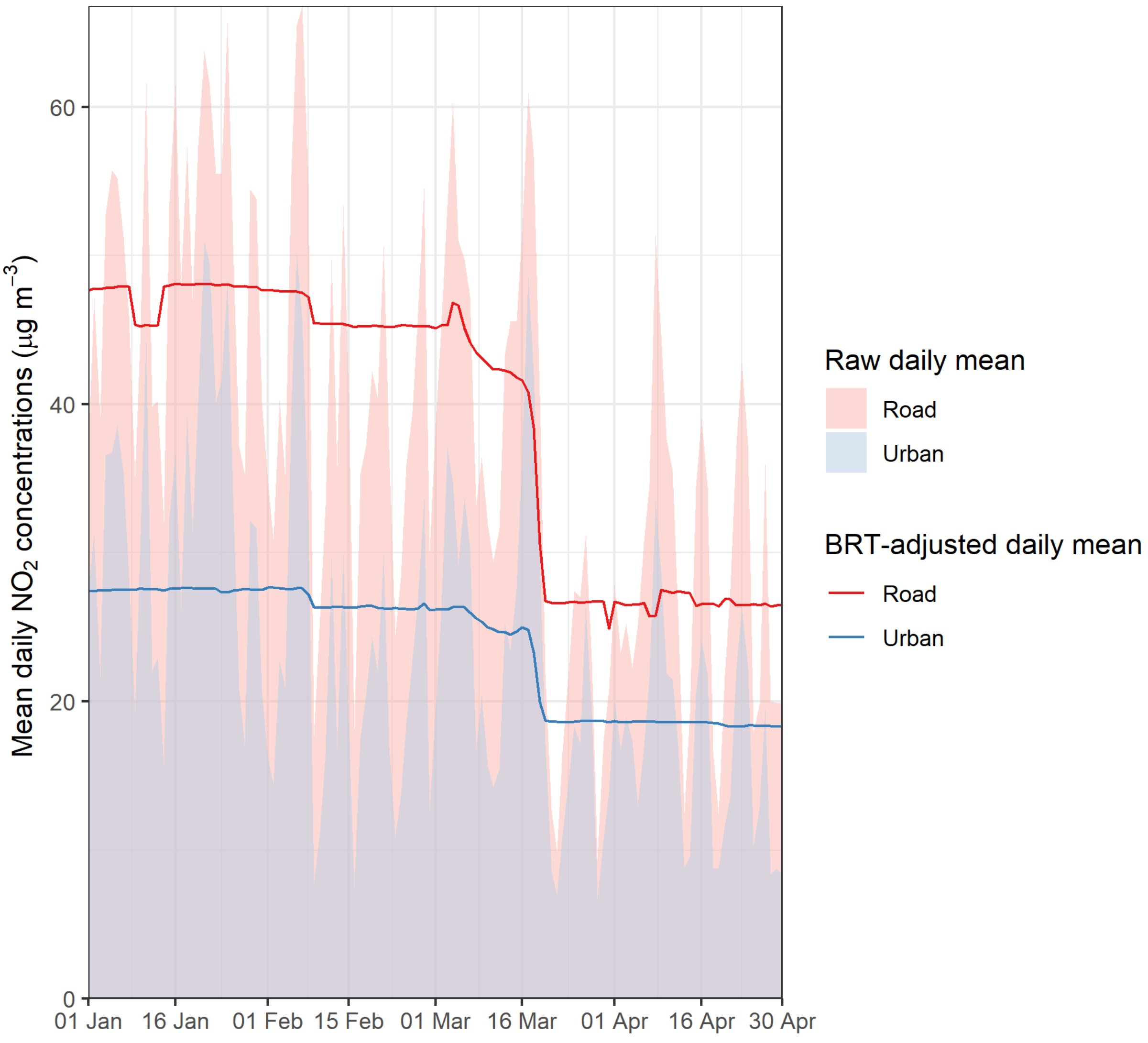

Figure 33: Mean Daily NO_2 Averaged across All Roadside and Urban Background Monitoring Sites in Paris

5.48 As for the other cities, the period averages before and during the most stringent phase of lockdown have been calculated across all three BRT model runs. Representative non-traffic background values have then been subtracted to show the change in the road increment. The relative changes in the road contributions to NO_2 concentrations during the Covid-19 lockdown at the nine roadside sites are shown in Figure 34. As observed for Berlin, London, and Madrid, there appears to be a clear spatial pattern, with larger reductions observed in the centre of the city, and smaller reductions in outer areas. This corresponds with an expectation that traffic activity levels may have fallen more steeply in commercial districts than in residential areas. Reflecting this spatial pattern, and as with Berlin and London, the sites have been grouped according to whether or not they are within this central area. The boundary is largely arbitrary, but the central area has been taken as that within, but not directly adjacent to, the Boulevard Périphérique (the inner ring road).

5.49 Figure 35 shows the site-specific reductions in NO_2 concentrations according to this grouping. There is a clear pattern with those sites within the Boulevard Périphérique showing consistent reductions of 84% or more, while sites in the outer areas showed reductions of 51-55%. Reductions for each zone have been calculated by averaging the results for each site as shown in Figure 35. It is a feature of the Paris data that most of the monitoring sites are within a relatively small area in the centre of the city. This means that the city-wide improvements which have been calculated are disproportionately weighted toward conditions in the centre despite.

5.50 As with the other cities, the overall patterns for NOx are very similar to those shown for NO_2 and are not presented separately. Averages have been calculated for NOx in the same way as shown for NO_2.

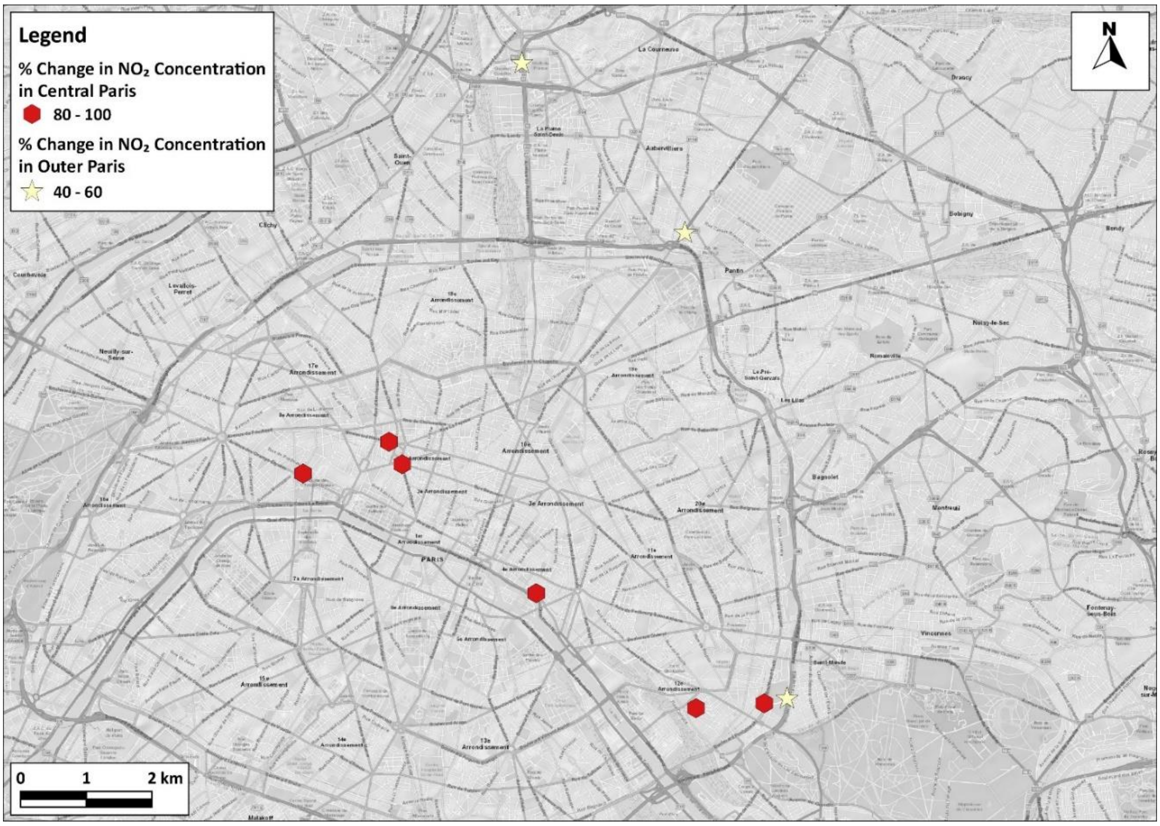

Figure 34: **Map of Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Paris**

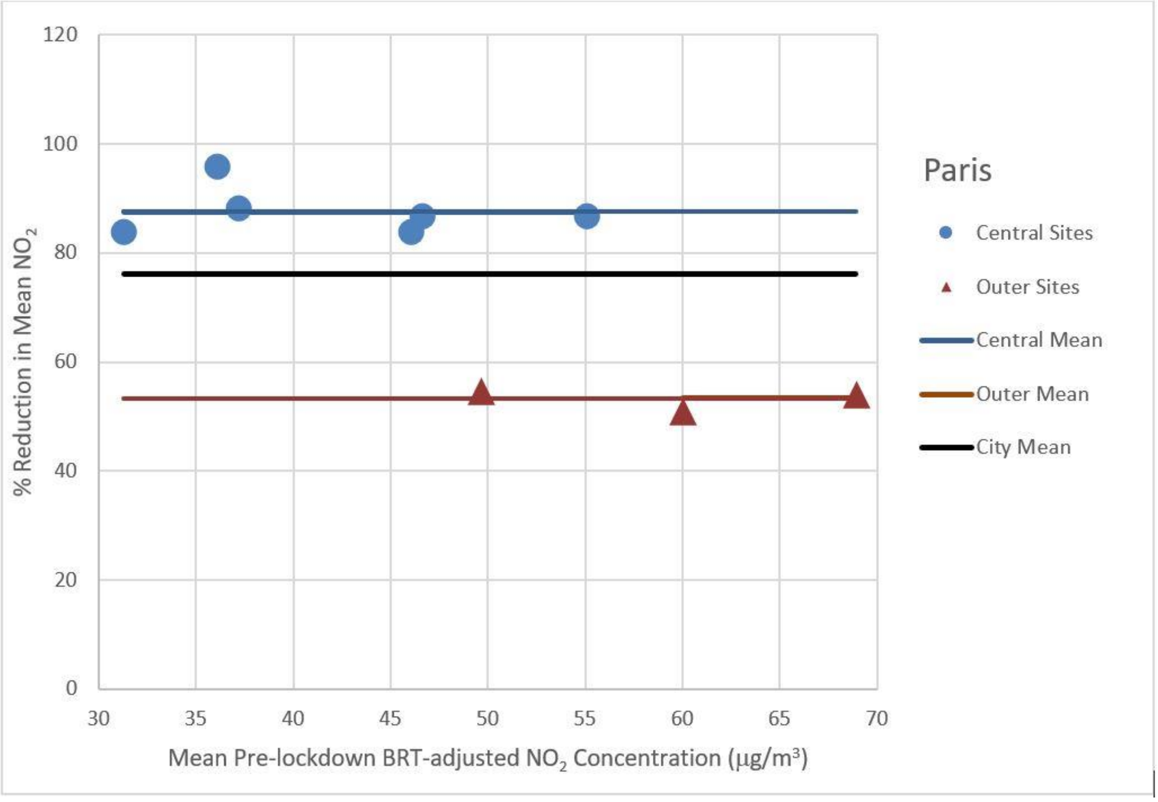

Figure 35: Relative Reductions in Mean Traffic-related NO_2 During the Most Stringent Phase of Lockdown in Paris

Table 26: Baseline scenario outputs; Paris

	Zone	% reduction in traffic contribution	Mean pre-lockdown concentration (μ/m³)	Mean lockdown concentration (μ/m³)	Mean change (μ/m³)	Roadside sites included
NO_2	Inner	88	42.1	21.8	-20.3	6
	Outer	53	59.5	38.2	-21.3	3
	City-wide	76	47.9	27.3	-20.6	9
NOx	Inner	76	89.2	39.8	-49.4	6
	Outer	52	173.6	99.6	-74.0	3
	City-wide	68	117.3	59.7	-57.6	9

Scenario Analysis

5.51 Table 27 shows that the emission reductions seen during the most stringent phase of lockdown are not attainable, other than for the outer zone for Scenarios C1 and C2 and only with a very high level of conversion. This reflects the large reduction in emissions during the lockdown period, making the target harder to replicate through fleet and transport changes. While the monitoring data have allowed the analysis to differentiate between the central and outer areas of the city, the emissions data used is based on a uniform fleet mix across the whole city. Therefore, any changes in the fleet in the centre of the city, as a result of transport policies, are not reflected in the analysis outputs.

Table 27: Outputs for scenario analyses, showing the conversion of ICE passenger car km to EV to meet target NOx emission reduction; Paris

Scenario	Zone	Reduction in passenger car km	NOx emission reduction[18]
Scenario A, ICE cars to EV	Inner	>100%	47%
	Outer	>100%	47%
	City-wide	>100%	47%
Scenario B, ICE cars, vans and trucks to EV	Inner	>100%	36%
	Outer	>100%	36%
	City-wide	>100%	36%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	Inner	>100%	57%
	Outer	88%	
	City-wide	>100%	57%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Inner	>100%	62%
	Outer	78%	
	City-wide	>100%	62%
Scenario D, Long term ICE phase out	Inner	75%	
	Outer	50%	
	City-wide	67%	

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[18] Reduction in the traffic component of NOx emissions, if 100% of ICE cars are converted to EV, where greater that 100% would be required to meet the baseline scenario reductions.

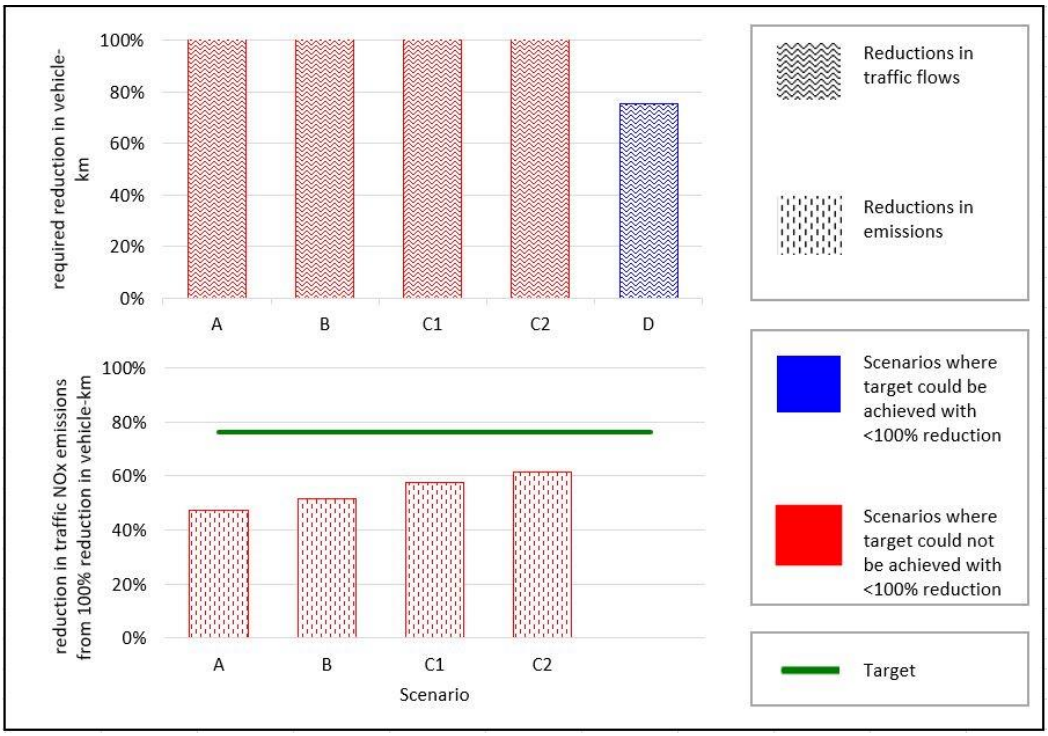

Figure 36: Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Central Paris

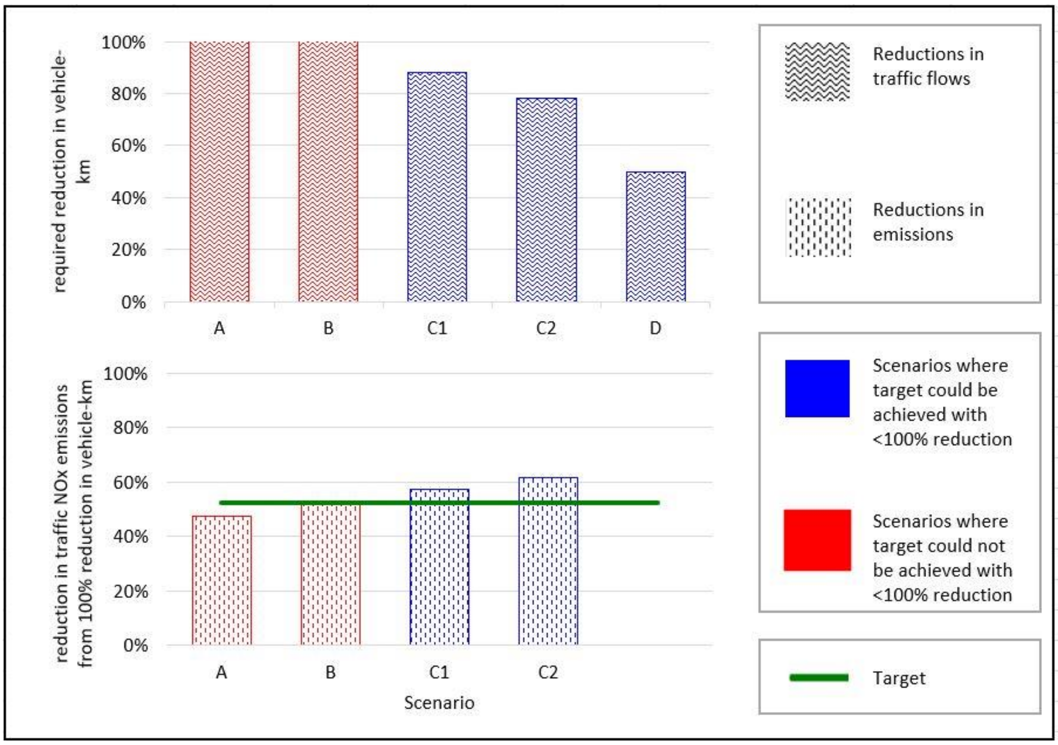

Figure 37: Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Outer Paris

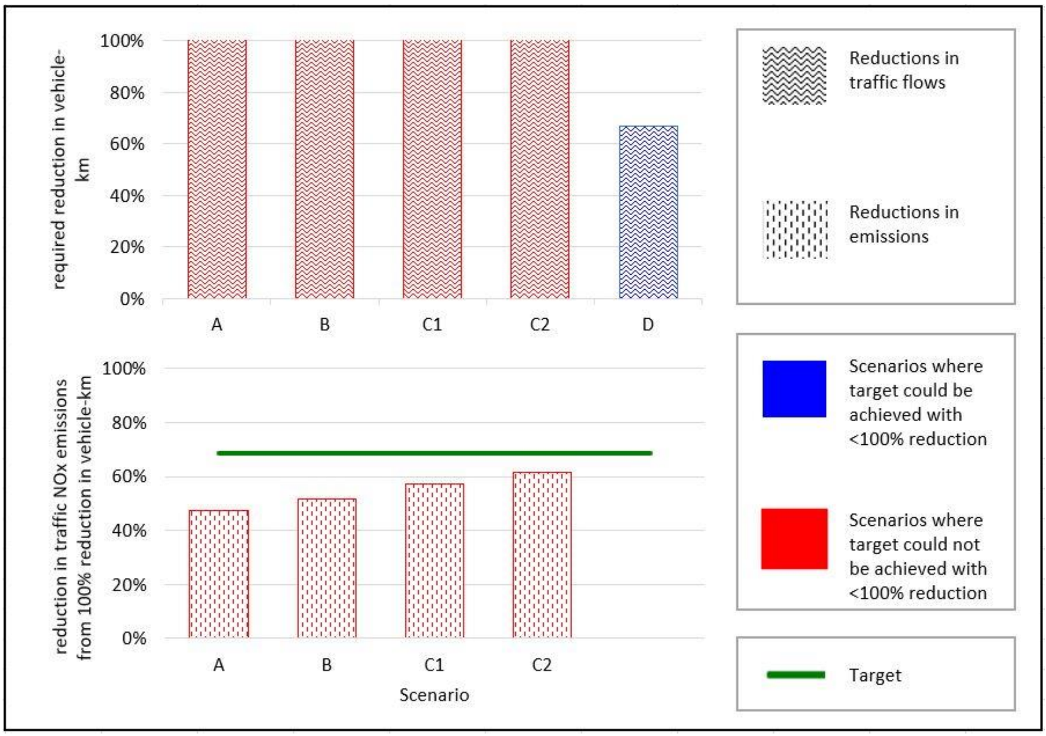

Figure 38: **Outputs for scenario analyses, showing the conversion of ICE to EV to meet target NOx emission reduction – also showing effect of 100% modal shift against the target; Paris City-wide**

5.52 Reductions in $PM_{2.5}$ emissions from full implementation of the scenarios are relatively modest and probably reflect a cleaner ICE vehicle fleet in current conditions. Given current levels of uptake, not just of EV and hybrid vehicles but higher Euro 6 standard ICE cars, tailpipe emissions from road transport may not be the largest source of $PM_{2.5}$ emissions in many European cities by 2030 and beyond. It is therefore unsuprising that emission reductions achieved through conversion from ICE to EV are relatively modest.

5.53 Table 29 shows the emissions impact of converting each vehicle class, by turn, to EV, and shows the relative importance of each class in terms of emissions. The relative burden of reducing transport NOx emissions inevitably falls on passenger cars as they make up over half of the total emissions.

Table 28: Outputs for Scenario Analyses, showing traffic $PM_{2.5}$ emission reductions for achieving each scenario; Paris

Scenario	Zone	Reduction in $PM_{2.5}$ emissions, traffic component [19]
Scenario A, ICE cars to EV	Inner	34%
	Outer	34%
	City-wide	34%
Scenario B, ICE cars, vans and trucks to EV	Inner	36%
	Outer	36%
	City-wide	36%
Scenario C1, ICE cars to EV and non-transport or non-emission transport	Inner	39%
	Outer	36%
	City-wide	39%
Scenario C2, ICE cars, vans and trucks to EV and non-transport or non-emission transport	Inner	41%
	Outer	35%
	City-wide	41%
Scenario D, Long term ICE phase out	Inner	45%
	Outer	34%
	City-wide	43%

Table 29: Scenario E, class by class 100% switch to EV, emission reductions from the transport component; Paris

Pollutant		Reduction in transport emissions[20]
NOx	Cars	47%
	LGV	22%
	HGV	20%
	Bus & coach	10%
	Motorcycles	0.4%
$PM_{2.5}$	Cars	34%
	LGV	15%
	HGV	5%
	Bus & coach	2%
	Motorcycles	1%

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[19] Reduction in traffic component $PM_{2.5}$ emissions following 100% conversion is undertaken, where >100% is indicated in the previous table.

[20] Only city-wide figures shown here as the COPERT traffic model used does not differentiate between different city zones.

6 Key Messages

6.1 The analysis presented in this report demonstrates that the responses to the Covid-19 pandemic have delivered a significant improvement to air quality in cities during the most stringent phase of lockdown. This is particularly notable in terms of measured concentrations of NO_2. Such a result is not surprising, given that the major source of NO_2 in urban areas in European cities is road traffic emissions of NOx, and that the key response to the pandemic was to restrict movement and thus reduce traffic flows.

6.2 Quantifying the air quality benefit of "lockdown" is more complex, and weather is a key confounding factor, as is the spatial variation of pollutant concentrations within any given area. The use of a de-weathered approach based on measured concentrations used in this study has produced results which are comparable to those cited elsewhere for specific cities. Nevertheless, any study attempting to show the response across a large urban area using only several data points will be subject to uncertainty. Therefore, the results presented in this report, both for the baseline reductions and the analysis scenarios, should be taken as providing the scale of change, rather than precise calculations.

6.3 With that in mind, the six cities can be divided into approximately three groups:

- Group 1, light lockdown: Budapest
- Group 2: moderate lockdown: Berlin, Brussels, London
- Group 3: severe lockdown: Madrid, Paris.

6.4 The terms light, medium and severe are not intended as a commentary on the lockdown strategies or the experience of people living and working in those cities. They are merely descriptors for the impacts that the Covid-19 response measures had on the levels of traffic, while they were in place.

6.5 As expected, the Group 3 cities saw a greater reduction in traffic NOx emissions during the most stringent phase of lockdown (65-81% on a city-wide basis) as compared to Group 2 and Group 1 (28-49% and 14% respectively). This also means that the reductions are harder to replicate in Group 3 through changes to mobility policies and the vehicle fleet, requiring more radical measures and a greater emphasis on the conversion of HGV and LGV traffic to zero emission equivalents. This is reflected in the results of the analysis scenarios, where more than 100% passenger car conversion is required to meet the reduction target in the Group 3 cities.

Group 1 (Budapest)

6.6 The analysis indicates that the air quality improvements in Budapest could be achieved solely through converting passenger car journeys from petrol and diesel to electric. However, while a 42% conversion is theoretically feasible, a more balanced approach would also deliver the improvement by including the conversion of vans, HGVs and buses to electric, and encouraging a shift to walking, cycling and teleworking. Scenario C1, which does not include HGVs and LGVs, but does include buses, requires only a 6% shift in passenger car journeys to electric. Including HGVs and LGVs, in Scenario C2, suggests that a shift in the car fleet is not required to meet the target, suggesting that Budapest could go further and deliver even greater improvements than those seen during lockdown, in the next 10 to 15 years.

Group 2 (Berlin, Brussels, London)

6.7 This Group experienced reductions in the traffic component of NOx emissions of between 25 and 49%, although central London experienced a reduction of 66%. This makes the required reductions more challenging to meet through fleet changes and modal shift, but in all but one case they were achievable using the analysis scenarios presented.

6.8 One feature to note is that there is a significant difference in achievability of the target between the central and outer zones in both Berlin and London. Local emissions data was available for both cities, and both have actively sought to encourage cleaner vehicles in the central areas through the Low Emission Zone in Berlin and the Ultra-low Emission Zone in London. Such efforts have been made in other European cities but, unfortunately, this is not fully reflected in the Emisia COPERT data used.

6.9 Using London as an example, the relatively clean car fleet in the central area means that a higher proportion of the fleet must change to achieve the same level of reduction. Together with the large reduction in traffic NOx emissions observed during the most stringent phase of lockdown, this results in the largest necessary mobility change identified in this study, peaking at 245% for Scenario A in the central area. A similar mismatch between the requirements in the central and outer areas is seen in Berlin, although the lower level of emission reduction during lockdown means that the target fleet shifts are all below 100%. This may also be the case in other cities, but local emissions and activity data would be required to fully analyse it.

6.10 In all of the Group 2 cities, a shift of over 50% of passenger car journeys from ICE to electric is required to meet the target, rising to around 90% for London on a city-wide basis. This is not likely to be feasible in the short term, nor is the shift required with only a 10% move to electric in HGV and LGV journeys (Scenario B). Scenario C1 is potentially more achievable, in areas such as outer Berlin, together with Scenario C2, although shifting all buses to EV, along with 10% of HGVs and LGVs, will remain significant challenge (albeit a stated aspiration for many cities).

6.11 Scenario F reveals the importance of different vehicle classes for different cities. LGVs are clearly an important emission source in London, and going beyond the 10% cap in Scenarios B, C1 and C2 will make a proportionately greater impact than in other cities, while HGVs should be a key target for Brussels. However, this latter point should be tested using local emission data.

Group 3 (Madrid, Paris)

6.12 With the largest calculated reductions in traffic NOx emissions (52-76%), the Group 3 cities have the greatest challenge in terms of delivering that target through fleet and mobility changes. Those targets are not met in Scenarios A, B and C1, and only marginally so for Madrid in Scenario C2, requiring a shift of 94% of car journeys to electric. Achieving the target in outer Paris is, theoretically, possible for Scenarios C1 and C2, although at 78% for C2, this would still represent a substantial challenge in the short term.

6.13 The required reduction in outer Paris is similar to that for outer London; however, the required shifts to electric in passenger car journeys are significantly higher in outer Paris than in outer London. One key difference between the two cities is that the reduction in absolute concentrations of NO_2 during lockdown was far higher in Paris than in London. Thus, in absolute terms, the reduction in emissions required to replicate it is accordingly greater.

6.14 It is likely that the level of emissions reductions seen during the most stringent phase of lockdown in Madrid and Paris are only replicable in the longer term, with between 60 and 70% of *all* vehicle journeys converted to electric. The lack of zero emission options for HGVs and the high level of fleet turnover required means that this is likely to be an ambition which can only be realised over a longer timescale.

7 Acknowledgements

7.1 The authors of this report would like to thank and acknowledge the following people (and their colleagues) for their assistance in the identification and supply of data:

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Kéri András	City of Budapest, Office of the Deputy Mayor for Climate and Developments
Rosalind O'Driscoll	Greater London Authority
Sergio Fernández Balaguer	EMT Madrid
Silvia Monori	Mobility Environment and Territory Agency, Milan
Karine Léger	Airparif, Paris

8 Glossary

AQC	Air Quality Consultants
BRT	Boosted Regression Tree
EU	European Union
EV	Electric Vehicle; for the purposes of this study, this can also include hybrid vehicles operating in fully electric mode
HDV	Heavy Duty Vehicles (> 3.5 tonnes)
HGV	Heavy Goods Vehicle
ICE	Internal Combustion Engine
LDV	Light Duty Vehicles (<3.5 tonnes)
LGV	Light Goods Vehicle
$\mu g/m^3$	Microgrammes per cubic metre
NO	Nitric oxide
NO_2	Nitrogen dioxide
NOx	Nitrogen oxides (taken to be NO_2 + NO)
NRMM	Non-road Mobile Machinery
PM_{10}	Small airborne particles, more specifically particulate matter less than 10 micrometres in aerodynamic diameter
$PM_{2.5}$	Small airborne particles less than 2.5 micrometres in aerodynamic diameter
WHO	World Health Organisation

9 Appendices

A1 Analysis of $PM_{2.5}$ and PM_{10} measurements

A1.1 The calculation of the effects of the city lockdowns has focused on NOx and NO_2 measurements. This is because, while it seems self-evident that emissions of $PM_{2.5}$ and PM_{10} from road traffic must have reduced during the most stringent phase of lockdown, it is extremely difficult to show any effect directly from the ambient measurements. The principal difficulty is that local traffic emissions typically contribute a much smaller proportion of roadside PM concentrations than of roadside NOx concentrations. While NOx concentrations measured at the kerbside can be many times those at background, the kerbside increment of PM tends to be just a fraction of the total. Different methods of measuring PM concentrations are routinely used, and it is difficult to make precise comparisons between different instrument types. Thus, when comparing measurements made at the roadside with those made at background, it can be difficult to ascribe differences purely to environmental factors (i.e. real concentration increments) rather than measurement artefacts. Another key difficulty is that the non-road increment to PM concentrations is highly temporally variable, and often not driven by local-scale meteorology. The BRT approach used in this study does not typically capture changes to long-range transport well. This is exacerbated by the timing of the lockdowns in Europe, which coincided with a period in which ammonia emissions (which are a key driver of secondary $PM_{2.5}$ formation) were elevated.

A1.2 On balance, it is considered unreasonable to expect the same approach used for NOx to also show the effect of lockdown on traffic emissions of PM_{10} or $PM_{2.5}$. It is nevertheless helpful to examine the measurements from each city. Figure A1.1 to Figure A1. show the observed and BRT-adjusted time series of $PM_{2.5}$ concentrations in each of the cities (PM_{10} data are shown for Budapest because $PM_{2.5}$ is not measured).

A1.3 The BRT-adjusted time series for Berlin in Figure A1.1 are almost all affected by a large episodic peak of concentrations in late March and by a small step change downward in late January. The average BRT-adjusted concentrations measured during lockdown are consistently higher than those measured before lockdown at both the road and the urban background sites. There is no obvious pattern of these increases being either higher or lower at either site type. For the reasons explained in Paragraph A1.1, this should not be taken to suggest that lockdown in Berlin did not affect traffic emissions of $PM_{2.5}$, but it has not been possible to disentangle this response from the other factors affecting concentrations over this period.

A1.4 Figure A1.2 shows that there are no roadside $PM_{2.5}$ monitors in Brussels and episodes and data gaps during the time series make it impossible to see any effect of the lockdown.

A1.5 Figure A1.3 shows the time series of PM_{10} concentrations in Budapest. The averages of BRT-adjusted concentrations at each site are higher during lockdown than earlier in the year, but the variability in these data make it impossible to discern the cause. Concentrations measured at the background sites are also higher than those measured at roadside site hu5.

A1.6 All of the London data in Figure A1.4 and Figure A1.5 show similar peak episodes, first in late January and then in late March and April. Most, but not all, of these peaks have been removed through the BRT adjustment. Because of the number of sites in London, it is helpful to summarise the changes as shown in Figure A1.. This shows that there is relatively little differentiation between the concentrations measured at roadside and urban background sites, either in terms of the total pre-lockdown concentrations, or the scale or direction of changes during lockdown.

A1.7 Figure A1. shows the time series for Madrid. In this case, average concentrations measured during lockdown appear to be systematically different from those measured earlier in the year (the relative changes in period-mean total roadside concentrations all lie between -15% and -21%). However, the same reasons which mean that little weight can be placed on this analysis for the other cities also mean that these reductions should not be ascribed directly to changes in transport emissions.

A1.8 Figure A1. shows the equivalent data for Paris. Again, all of the period-mean BRT-adjusted concentrations are higher during lockdown than earlier in the year; again, most likely reflecting regional transport patterns and potentially episodes of higher ammonia emissions.

A1.9 The UK Government's Air Quality Expert Group has produced an initial review of the effects of lockdown on UK air quality (AQEG, 2020). This concurs with the findings presented in this study (for all cities except Madrid), and that meteorological conditions caused higher $PM_{2.5}$ levels during lockdown than the averages experienced over other periods. An analysis was, however, presented, which combined UK ambient measurements, machine learning methods similar to those presented in this study, and forecasts using the GEOS Composition Forecasting System. This indicated that despite coinciding with a (most likely) unrelated episode of higher concentrations, levels of $PM_{2.5}$ measured in the south of the UK were approximately 2 - 5 $\mu g/m^3$ lower than might reasonably have been expected under a business-as-usual emissions scenario.

A1.10 An analysis such as that carried out using the GEOS Composition Forecasting System requires in-depth modelling which is beyond the scope of this study, but shows that the observations of higher concentrations during lockdown do not mean that transport emissions increased. The lockdowns reduced the amount of traffic on roads. In this respect, a relative reduction in emissions of traffic-NOx is likely to have been associated with a very similar relative reduction in emissions of traffic-PM. It is considered most appropriate to base the analysis on the measurements of NOx and NO_2 since this provides the most robust method for quantifying the effect of the lockdown. This is the approach that has been taken in this study. It should, however, be recognised that many of the managed interventions considered involve a switch to electric vehicles. In this respect the same intervention which achieves the lockdown improvements for NOx is unlikely to achieve the lockdown improvements for $PM_{2.5}$. This is because electric vehicles emit non-exhaust $PM_{2.5}$.

Where electric vehicles are part of the scenario, larger interventions would most likely be required to achieve the reductions in traffic-PM which occurred during lockdown, but it has not been possible to quantify this.

Berlin

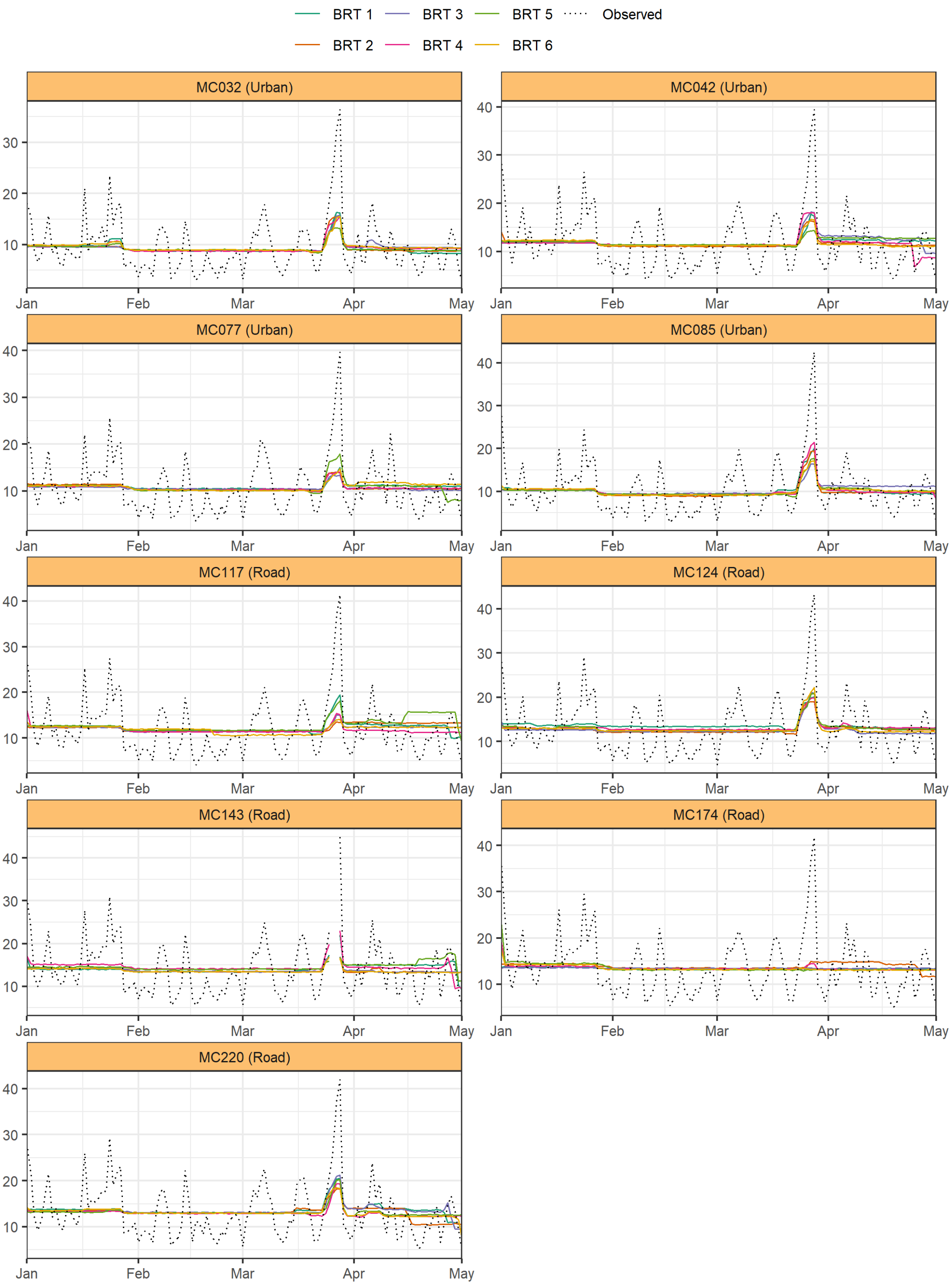

Figure A1.1: Daily mean $PM_{2.5}$ (µg/m^3) concentrations in Berlin

Brussels

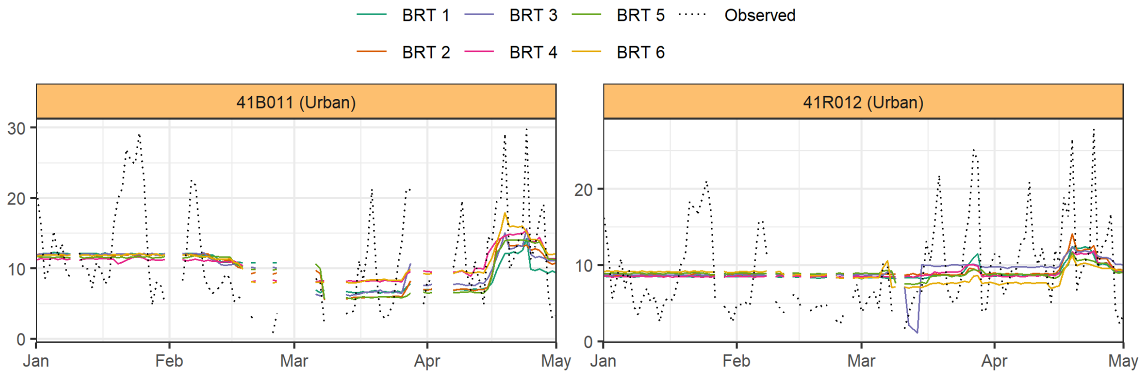

Figure A1.2: Daily mean $PM_{2.5}$ (µg/m^3) concentrations in Brussels

Budapest

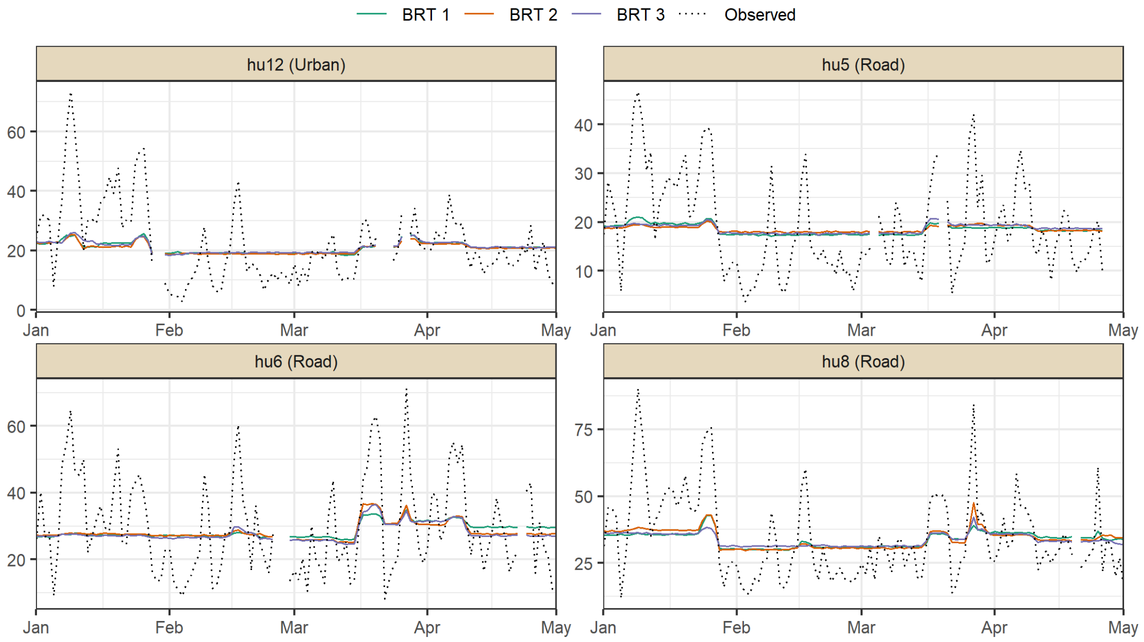

Figure A1.3: Daily mean PM_{10} (µg/m^3) concentrations in Budapest

London

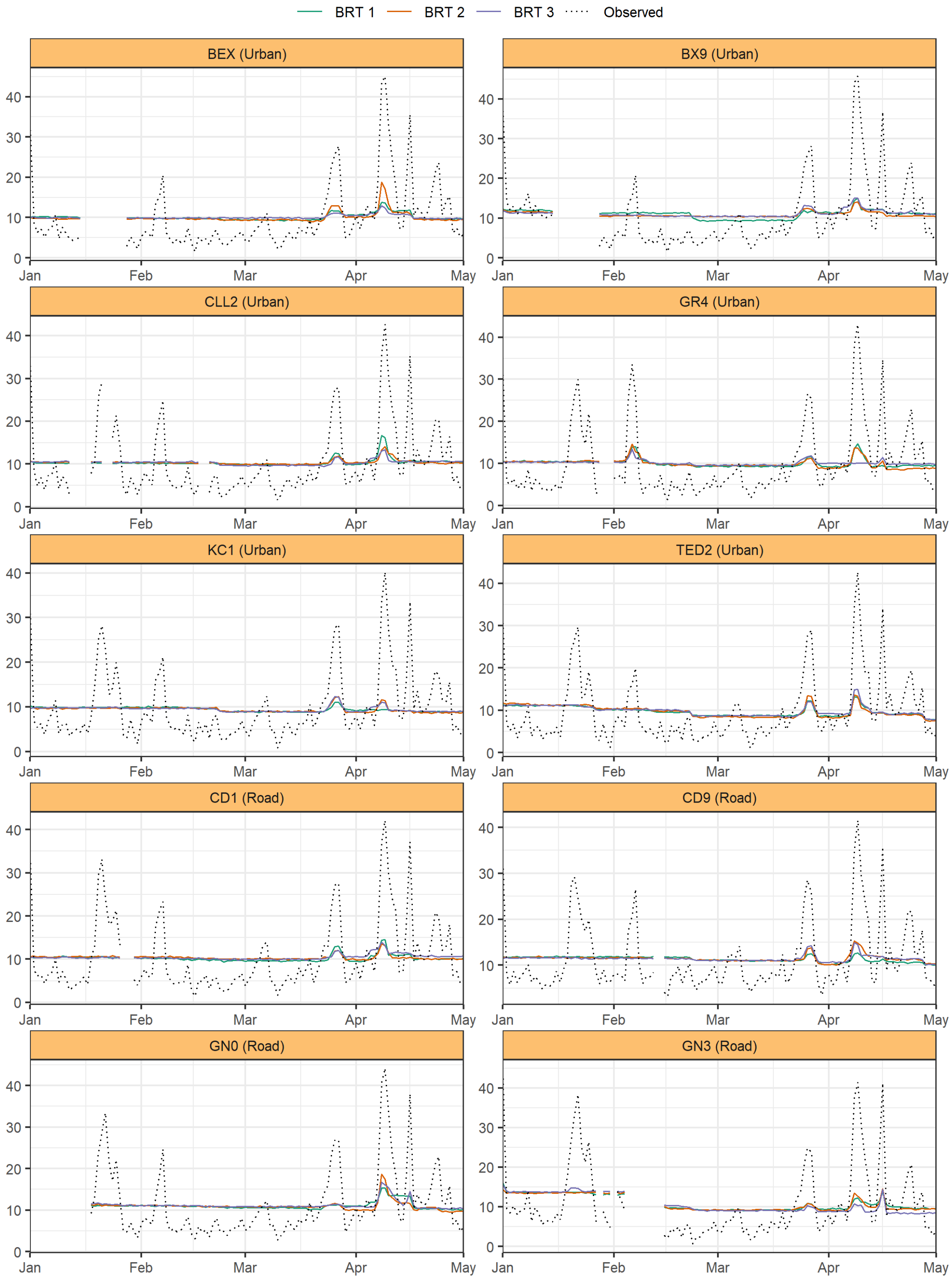

Figure A1.4: **Daily mean $PM_{2.5}$ ($\mu g/m^3$) concentrations in London**

London

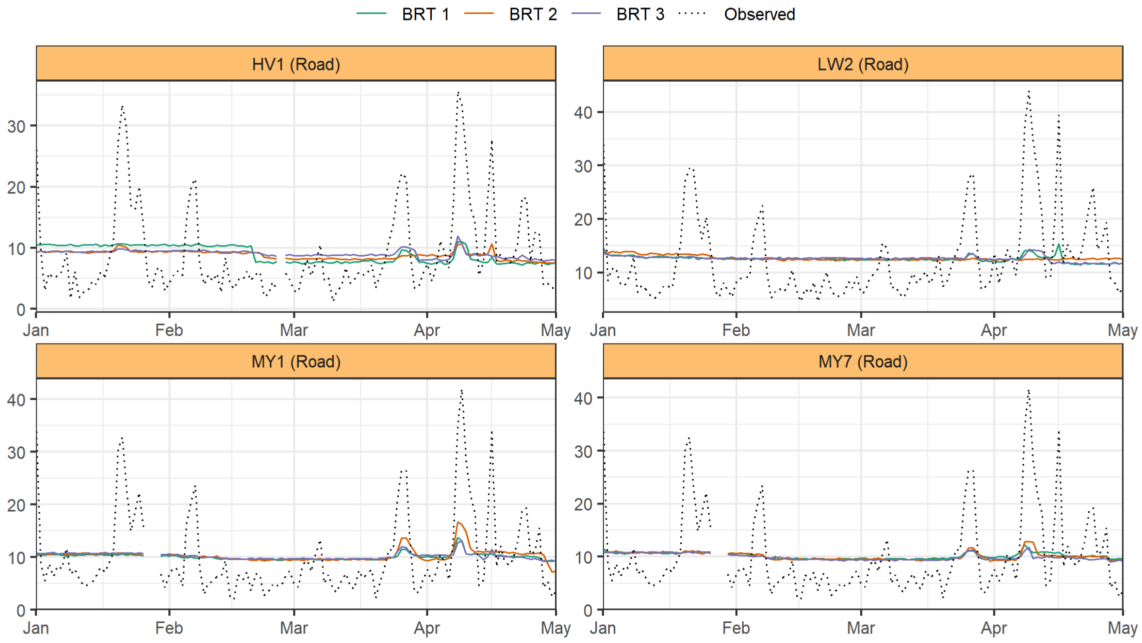

Figure A1.5: Daily mean $PM_{2.5}$ (µg/m^3) concentrations in London

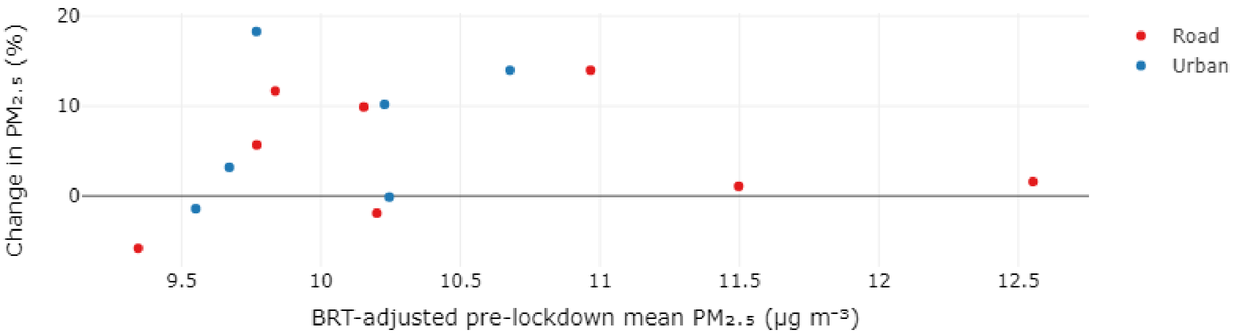

Figure A1.6: Change in BRT-adjusted mean $PM_{2.5}$ (µg/m^3) during the most stringent phase of lockdown in London

Madrid

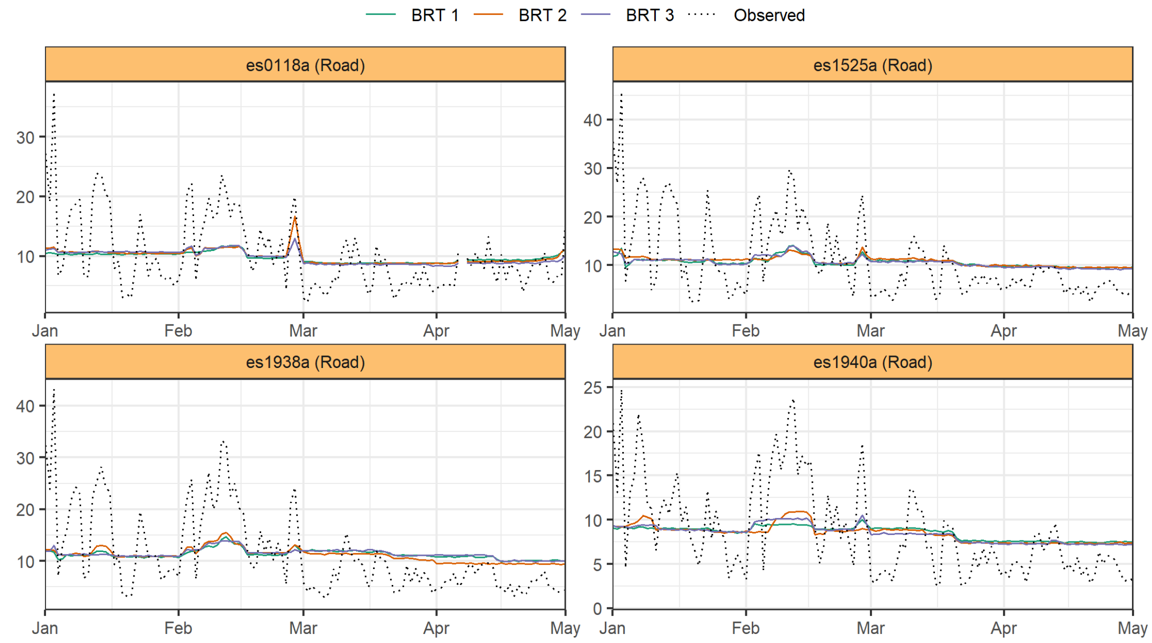

Figure A1.7: Daily mean $PM_{2.5}$ ($\mu g/m^3$) concentrations in Madrid

Paris

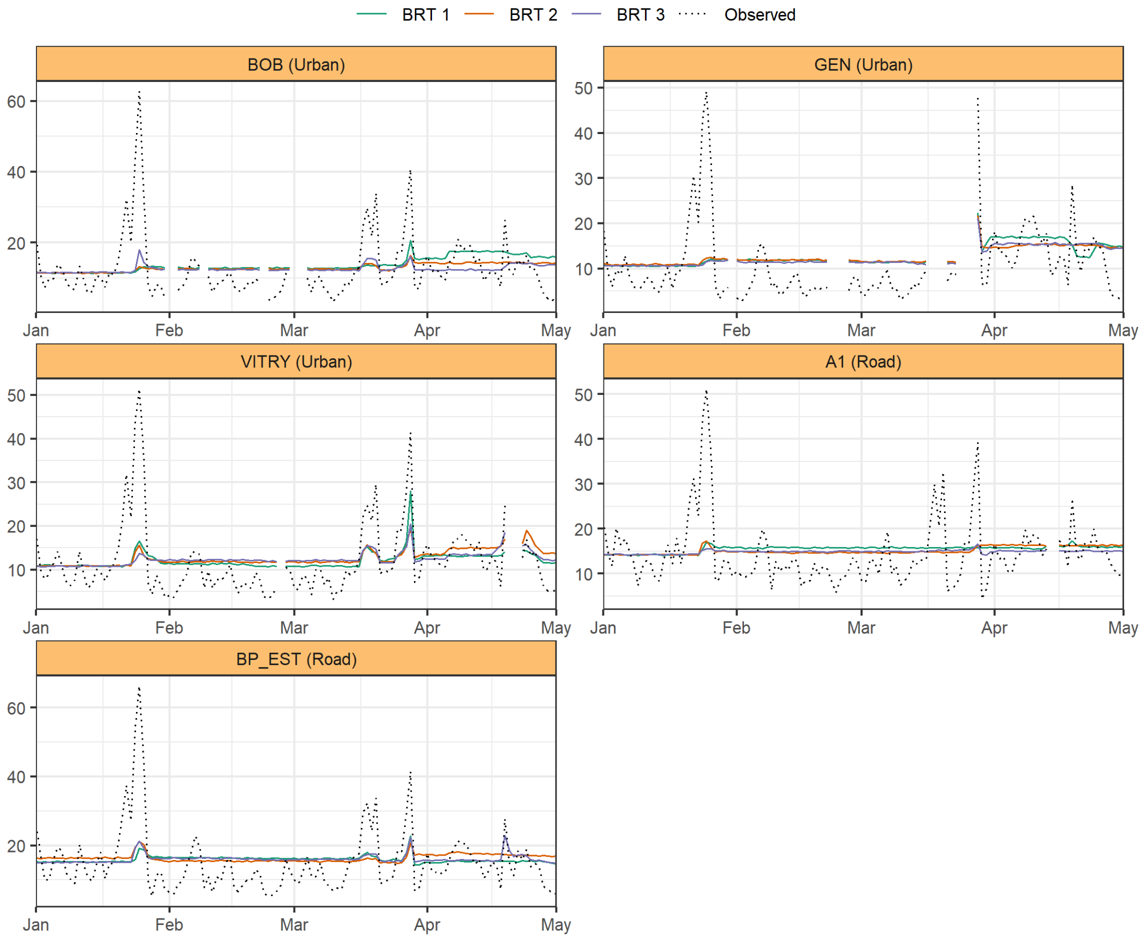

Figure A1.8: Daily mean $PM_{2.5}$ (µg/m^3) concentrations in Paris

A2 Details of Air Quality Monitoring Sites

Table A2.1: Air Quality Monitoring Sites Included in this Analysis

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
Berlin Sites					
MC010	Amrumer Str.	Urban Background	13.349	52.543	
MC018	Belziger Str.	Urban Background	13.349	52.486	X
MC027	Schichauweg	Urban Background	13.368	52.398	X
MC032	Jagen	Urban Background	13.225	52.473	X
MC042	Nansenstr.	Urban Background	13.431	52.489	X
MC077	Wiltbergstr.	Urban Background	13.490	52.644	X
MC085	Müggleseedamm	Urban Background	13.647	52.448	X
MC115	Hardenbergplatz	Traffic	13.333	52.507	✓
MC117	Schildhornstr.	Traffic	13.318	52.464	✓
MC124	Mariendorfer Damm	Traffic	13.388	52.438	✓
MC143	Silbersteinstr.	Traffic	13.442	52.468	✓
MC145	Jägerstieg	Urban Background	13.296	52.653	X
MC171	Brückenstr.	Urban Background	13.419	52.514	X
MC174	Frankfurter Allee	Traffic	13.470	52.514	✓
MC220	Karl-Marx Str.	Traffic	13.434	52.482	✓
MC282	Rheingoldstr.	Urban Background	13.530	52.485	X
MW088	Leipziger Straße	Traffic	13.388	52.51	✓
Brussels Sites					
41B001	Bruxelles (Arts-Loi)	Traffic	4.368	50.846	
41B004	Bruxelles (Sainte-Catherine)	Background	4.349	50.851	X
41B006	Bruxelles (Parlement EU)	Background	4.374	50.839	
41B008	Bruxelles (Rue Belliard)	Traffic	4.376	50.841	
41B011	Berchem-Sainte-Agathe	Background	4.288	50.858	X
41MEU1	Neder-Over-Heembeek	Background	4.393	50.895	X
41R001	Avant-port (Haren)	Background	4.334	50.85	
41R002	Molenbeek-Saint-Jean	Traffic	4.385	50.825	✓
41R012	Ixelles	Background	4.359	50.797	X
41WOL1	Uccle	Traffic	4.426	50.857	✓

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
Budapest Sites					
hu1	Budatétény	Urban	19.01	47.406	
hu3	Gilice tér	Background	19.181	47.43	
hu5	Kosztolányi D. tér	Traffic	19.041	47.475	
hu6	Teleki tér	Traffic	19.088	47.492	✓
hu7	Erzsébet tér	Traffic	19.052	47.499	
hu8	Széna tér	Traffic	19.028	47.509	✓
hu9	Honvéd	Background	19.07	47.523	
hu10	Korakás park	Background	19.144	47.544	
hu11	Káposztásmegyer	Background	19.115	47.582	
hu12	Pesthidegkút	Background	18.961	47.562	**X**
London Sites					
BEX	London Bexley	Suburban Background	0.185	51.466	**X**
BG2	Barking and Dagenham	Suburban	0.133	51.529	**X**
BN1	Tally Ho	Kerbside	-0.177	51.615	✓
BN2	London Barnet – Chalgrove School	Urban Background	-0.206	51.592	**X**
BQ7	Bexley – Belvedere West	Urban Background	0.137	51.495	**X**
BX1	Bexley – Slade Green	Suburban	0.185	51.466	**X**
BX2	Bexley – Belvedere	Suburban	0.159	51.491	**X**
BX9	Bexley – Slade Green	Suburban	0.185	51.466	
CD1	Camden – Swiss Cottage	Kerbside	-0.175	51.544	✓
CD9	Camden – Euston Road	Roadside	-0.129	51.528	✓
CLL2	London Bloomsbury	Urban Background	-0.126	51.522	
CR5	Croydon – Norbury	Kerbside	-0.123	51.411	✓
CR7	Croydon – Purley Way A23	Roadside	-0.118	51.362	✓
CT3	City of London – Sir John Cass School	Urban Background	-0.078	51.514	**X**
CT4	City of London – Beech Street	Roadside	-0.096	51.52	✓
CT6	City of London – Walbrook Wharf	Roadside	-0.092	51.51	✓
EA6	Ealing – Hanger Lane Gyratory	Roadside	-0.292	51.531	✓

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
EI1	Ealing – Western Avenue	Roadside	-0.266	51.524	✓
EN1	Enfield – Bush Hill Park	Suburban	-0.066	51.645	
EN4	Enfield – Derby Road	Roadside	-0.051	51.615	✓
EN5	Enfield – Bowes Primary School	Roadside	-0.125	51.614	✓
EN7	Enfield – Prince of Wales School	Urban Background	-0.022	51.669	**X**
GB6	Greenwich – Falconwood	Roadside	0.086	51.456	✓
GN0	Greenwich – A206 Burrage Grove	Roadside	0.074	51.491	✓
GN3	Greenwich – Plumstead High Street	Roadside	0.095	51.487	✓
GN4	Greenwich – Fiveways Sidcup Road A20	Roadside	0.064	51.435	✓
GR4	Greenwich – Eltham	Suburban	0.071	51.453	**X**
GR7	Greenwich – Blackheath	Roadside	-0.012	51.473	✓
GR8	Greenwich – Woolwich Flyover	Roadside	0.018	51.487	✓
GR9	Greenwich – Westhorne Avenue	Roadside	0.041	51.456	✓
HF4	Shepherd’s Bush	Roadside	-0.225	51.505	✓
HG1	Haringey Roadside	Urban Traffic	-0.068	51.599	✓
HG4	London Haringey Priory Park South	Urban Background	-0.125	51.584	**X**
HI1	Hillingdon 1 – South Ruislip	Roadside	-0.403	51.552	✓
HI3	London Hillingdon – Oxford Avenue	Roadside	-0.424	51.481	✓
HIL	London Hillingdon	Roadside	-0.461	51.496	✓
HIL1	London Hillingdon – Harmondsworth	Urban Background	-0.481	51.488	**X**
HIL5	London Hillingdon – Hayes	Roadside	-0.412	51.498	✓
HK6	Hackney – Old Street	Roadside	-0.085	51.526	✓
HORS	London Westminster	Urban Background	-0.132	51.495	**X**
HR1	Harrow – Stanmore	Urban Background	-0.299	51.617	

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
HR2	Harrow – Pinner Road	Roadside	-0.363	51.588	✓
HS2	Hounslow – Cranford	Suburban	-0.412	51.483	**X**
HS4	Hounslow – Chiswick	Roadside	-0.257	51.493	✓
HS5	Hounslow – Brentford	Roadside	-0.31	51.489	✓
HS6	Hounslow – Heston	Roadside	-0.365	51.479	✓
HS7	Hounslow – Hatton Cross	Urban Background	-0.428	51.463	
HS8	Hounslow – Gunnersbury	Roadside	-0.284	51.501	✓
HS9	Hounslow – Feltham	Roadside	-0.409	51.447	✓
HV1	Havering – Rainham	Roadside	0.205	51.521	✓
HV3	Havering – Romford	Roadside	0.179	51.573	✓
IS2	Islington – Holloway Road	Roadside	-0.116	51.555	✓
IS6	Islington – Arsenal	Urban Background	-0.107	51.558	**X**
KC1	London N. Kensington	Urban Background	-0.213	51.521	**X**
KC2	Cromwell Road	Roadside	-0.179	51.496	✓
KC3	Knightsbridge	Roadside	-0.164	51.499	✓
KC4	Chelsea	Roadside	-0.168	51.487	✓
KC5	Earls Court Road	Kerbside	-0.191	51.49	✓
KT4	Kingston Upon Thames – Tolworth Broadway	Roadside	-0.281	51.379	✓
LB4	Lambeth – Brixton Road	Kerbside	-0.115	51.464	✓
LB6	Lambeth – Streatham Green	Urban Background	-0.132	51.428	
LW2	Lewisham – New Cross	Roadside	-0.04	51.475	✓
LW4	Lewisham – Loampit Vale	Roadside	-0.016	51.465	✓
MY1	London Marylebone Road	Urban Traffic	-0.155	51.523	✓
MY7	London Marylebone Road	Kerbside	-0.155	51.523	
NB1	Westminster – Strand (Northbank BID)	Roadside	-0.117	51.512	✓

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
NEW2	Cam Road	Roadside	-0.002	51.538	✓
NEW3	Wren Close	Urban Background	0.015	51.515	X
RB4	Redbridge – Gardner Close	Roadside	0.031	51.577	✓
RB7	Redbridge – Ley Street	Urban Background	0.083	51.569	
RI1	Richmond Upon Thames – Castelnau	Roadside	-0.237	51.48	✓
RI2	Richmond Upon Thames – Barnes Wetlands	Suburban	-0.23	51.476	X
SIPS	Sipson	Urban Background	-0.456	51.484	X
SK5	Southwark – A2 Old Kent Road	Urban Traffic	-0.06	51.48	✓
SK6	Southwark – Elephant and Castle	Urban Background	-0.102	51.493	
ST4	Sutton – Wallington	Kerbside	-0.15	51.359	✓
ST6	Sutton – Worcester Park	Kerbside	-0.24	51.378	✓
TED2	London Teddington – Bushy Park	Urban Background	-0.346	51.425	
TH004	Tower Hamlets – Blackwall	roadside	-0.008	51.515	✓
TH2	Tower Hamlets Roadside	Urban Traffic	-0.042	51.523	✓
WA7	Wandsworth – Putney High Street	Kerbside	-0.216	51.463	✓
WA8	Wandsworth – Putney High Street façade	Roadside	-0.216	51.464	✓
WA9	Wandsworth – Putney	Urban Background	-0.216	51.465	
WAA	Wandsworth – Battersea	Roadside	-0.142	51.479	✓
WAB	Wandsworth – Tooting High Street	Roadside	-0.167	51.429	✓
WL1	Waltham - Forest Dawlish Road	Urban Background	-0.005	51.562	X
WL4	Waltham – Crooked Billet	Kerbside	-0.016	51.602	✓
WL5	Waltham – Forest Leyton	Roadside	-0.014	51.556	✓
WM6	Westminster – Oxford Street	Kerbside	-0.153	51.514	✓

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
Madrid Sites					
es0115a	Pza. de España	Traffic	-3.712	40.424	
es0118a	Escuelas Aguirre	Traffic	-3.682	40.422	✓
es0120a	Avda. Ramón y Cajal	Traffic	-3.677	40.451	✓
es0124a	Arturo Soria	Background	-3.639	40.44	
es0125a	Villaverde	Background	-3.713	40.347	
es0126a	Farolillo	Background	-3.732	40.395	
es1193a	Casa de Campo	Background	-3.747	40.419	
es1645a	Barajas Pueblo	Background	-3.58	40.477	
es1422a	Pza. del Carmen	Background	-3.703	40.419	
es1426a	Moratalaz	Traffic	-3.645	40.408	✓
es1525a	Cuatro Caminos	Traffic	-3.707	40.446	✓
es1521a	Barrio del Pilar	Traffic	-3.712	40.478	✓
es1532a	Vallecas	Background	-3.652	40.388	
es1937a	Mendez Alvaro	Background	-3.687	40.398	
es1938a	Castellana	Traffic	-3.69	40.44	✓
es1939a	Parque del Retiro	Background	-3.682	40.414	**X**
es1940a	Plaza Castilla	Traffic	-3.689	40.466	✓
es1941a	Ensanche de Vallecas	Background	-3.612	40.373	
es1942a	Urb. Embajada	Background	-3.581	40.462	
es1943a	Pza. Elíptica	Traffic	-3.719	40.385	✓
es1944a	Sanchinarro	Background	-3.661	40.494	
es1945a	El Pardo	Background	-3.775	40.518	
es1946a	Juan Carlos I	Background	-3.616	40.461	
es1947a	Tres Olivos	Background	-3.69	40.501	
Paris Sites					
A1	Auto A1 – Saint-Denis	Traffic	2.357	48.925	✓
ARG	Argenteuil	Background	2.224	48.951	**X**
AUB	Aubervilliers	Background	2.385	48.904	**X**
BASCH	Place Victor Basch	Traffic	2.328	48.828	
BOB	Bobigny	Background	2.453	48.903	
BONAP	Rue Bonaparte	Traffic	2.335	48.856	
BP_EST	Bld peripherique Est	Traffic	2.413	48.839	✓
CELES	Quai des Celestins	Traffic	2.361	48.853	✓

Site Code	Site Name / Location	Type	Latitude	Longitude	Modelled [a]
CHAMP	Champigny-sur-Marne	Background	2.518	48.816	**X**
DEF	La Defense	Background	2.241	48.891	**X**
ELYS	Av Champs Elysees	Traffic	2.312	48.869	✓
GEN	Gennevilliers	Background	2.295	48.930	**X**
HAUS	Boulevard Haussmann	Traffic	2.330	48.873	✓
HAUS	Boulevard Haussmann	Traffic	2.330	48.873	
NEUIL	Neuilly-sur-Seine	Background	2.278	48.881	**X**
OPERA	Place de l'Opéra	Traffic	2.333	48.870	✓
PA01H	Paris 1er Les Halles	Background	2.346	48.864	
PA07	Paris 7eme	Background	2.293	48.857	**X**
PA12	Paris 12eme	Background	2.394	48.837	
PA13	Paris 13eme	Background	2.360	48.829	**X**
PA15L	Paris Stade Lenglen	Background	2.270	48.830	**X**
PA18	Paris 18eme	Background	2.347	48.892	
RN2	Rn2-Pantin	Traffic	2.391	48.902	✓
SOULT	Boulevard Soult	Traffic	2.408	48.838	✓
STDEN	Saint-Denis	Background	2.362	48.937	**X**
TREMB	Tremblay-en-France	Background	2.575	48.956	**X**
VILLEM	Villemomble	Background	2.507	48.882	**X**
VITRY	Vitry-sur-Seine	Background	2.376	48.776	**X**

[a] **X** = included to define background conditions

✓ = included to define roadside conditions

Blank = excluded from the analysis owing to data capture or other quality control concerns.

A3 Site-specific NOx and NO$_2$ Time Series

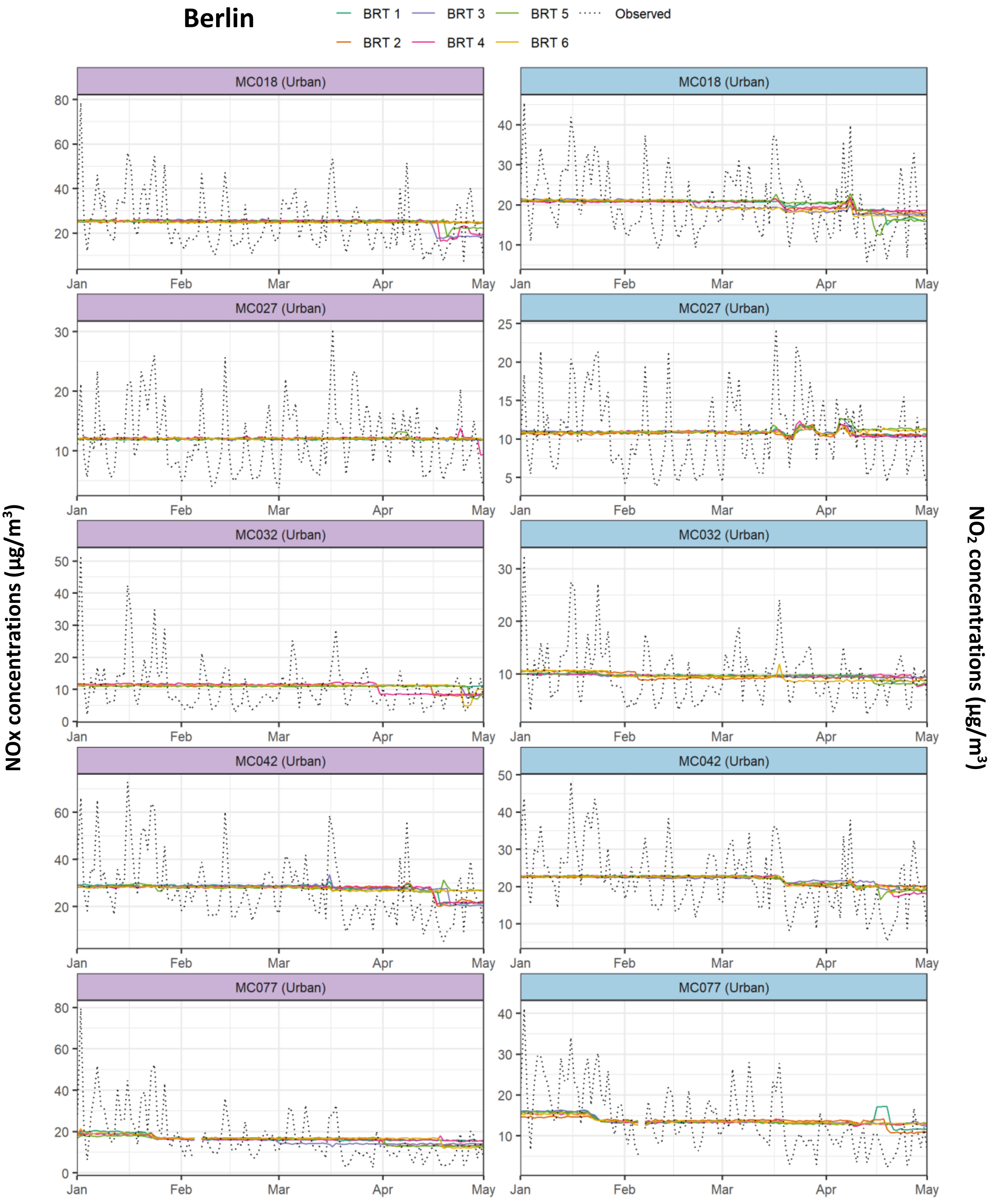

Figure A3.1: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in Berlin

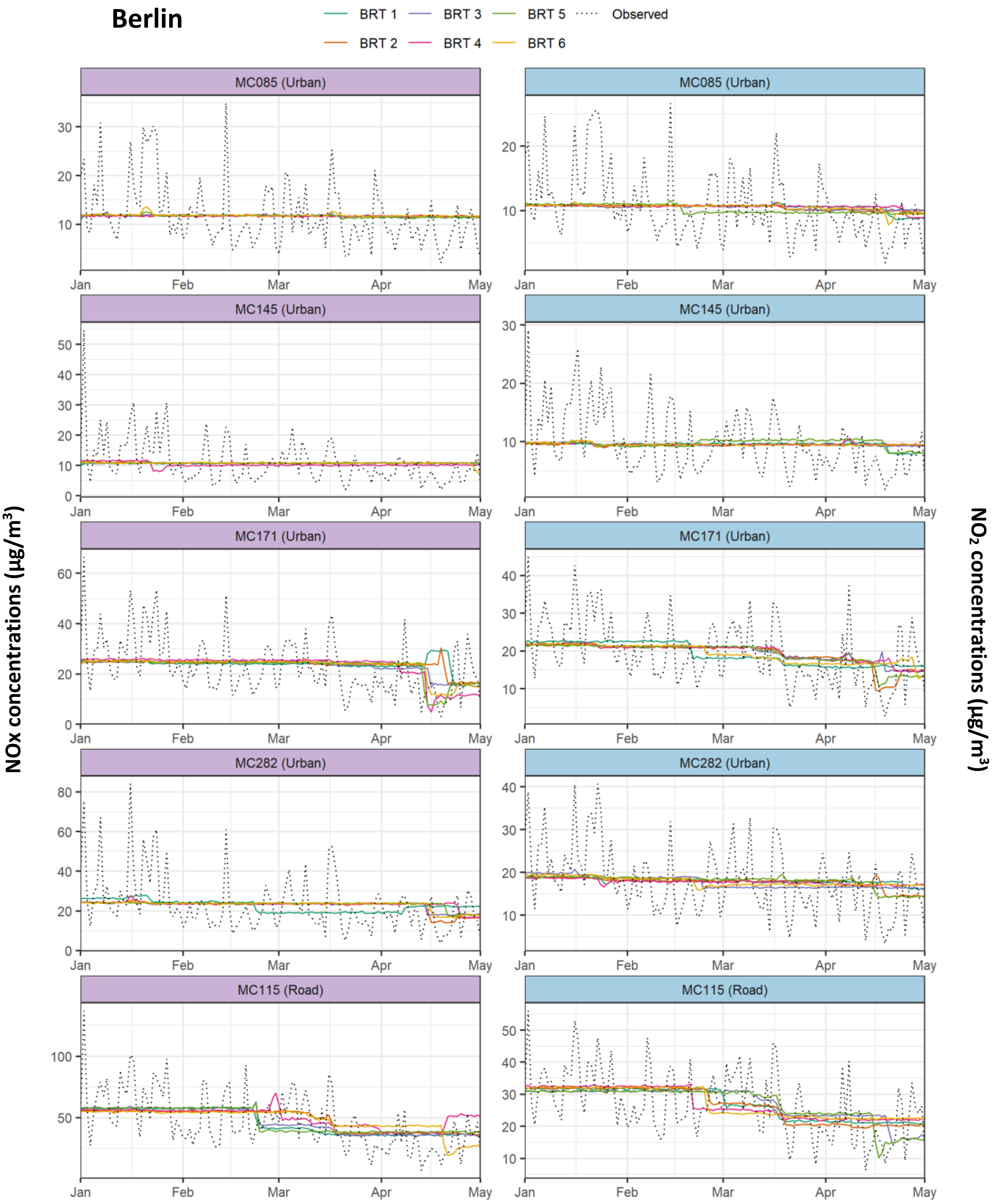

Figure A3.2: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in Berlin

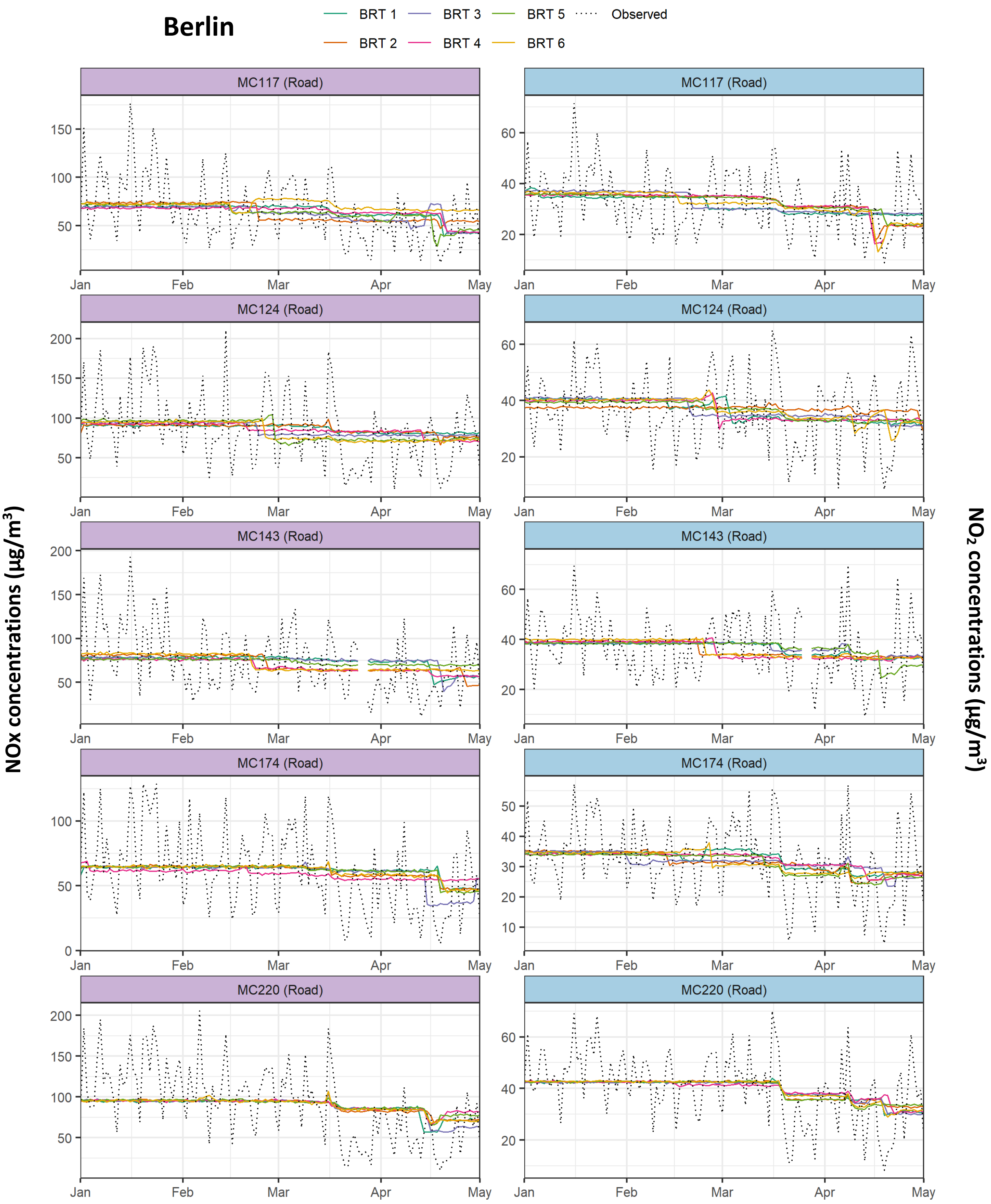

Figure A3.3: Daily mean NOx and NO_2 (µg/m^3) concentrations in Berlin

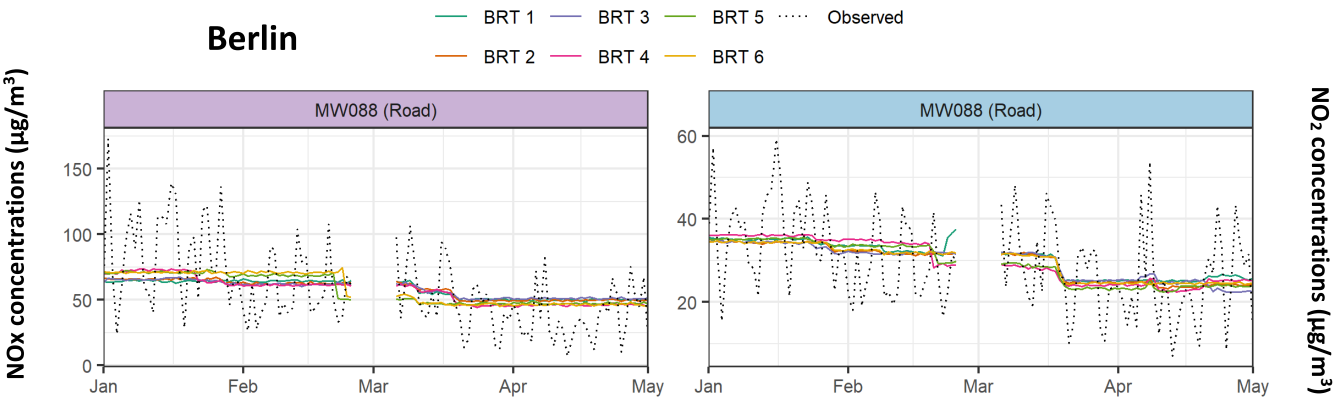

Figure A3.4: Daily mean NOx and NO_2 ($\mu g/m^3$) concentrations in Berlin

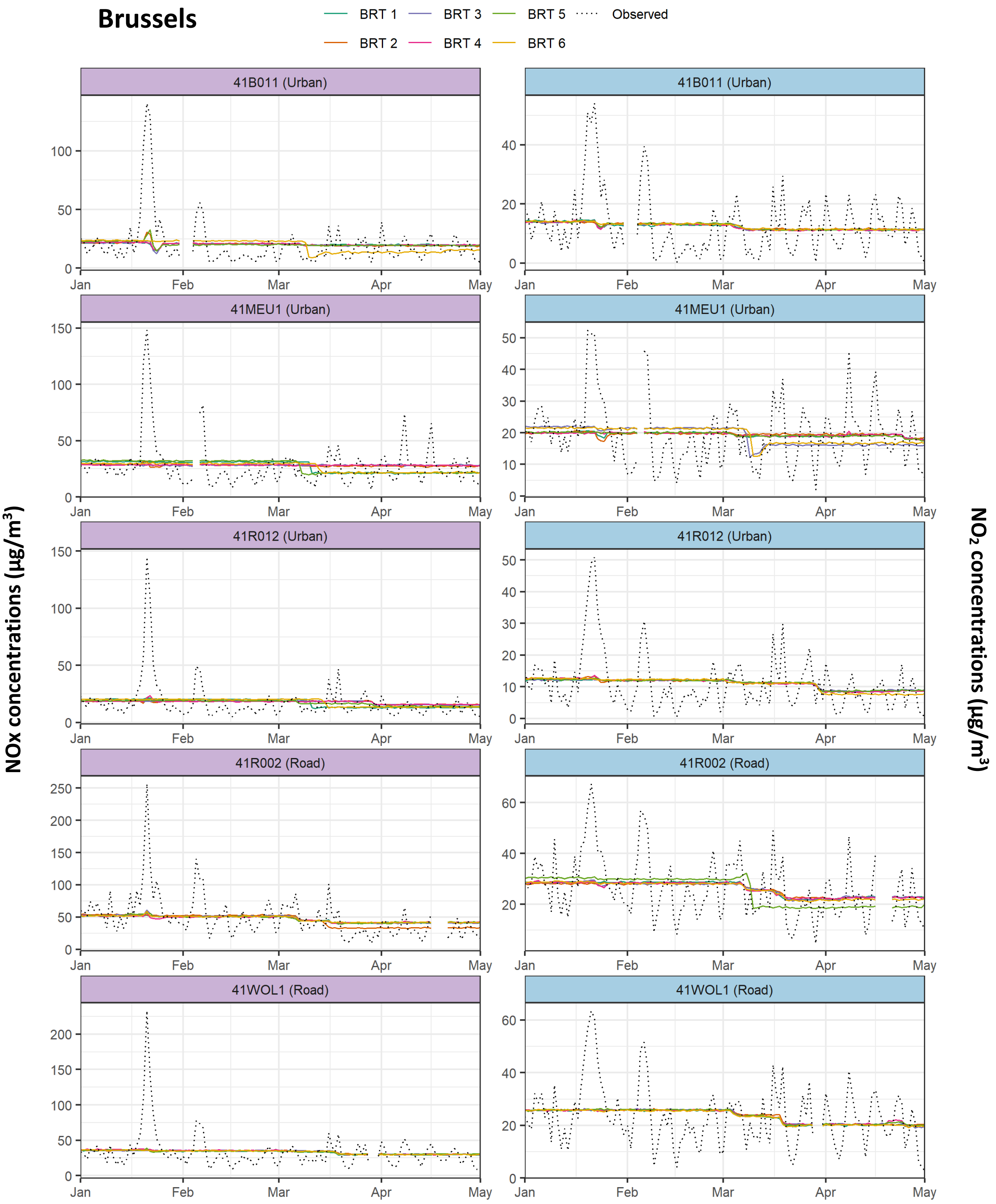

Figure A3.5: Daily mean NOx and NO_2 (µg/m^3) concentrations in Brussels

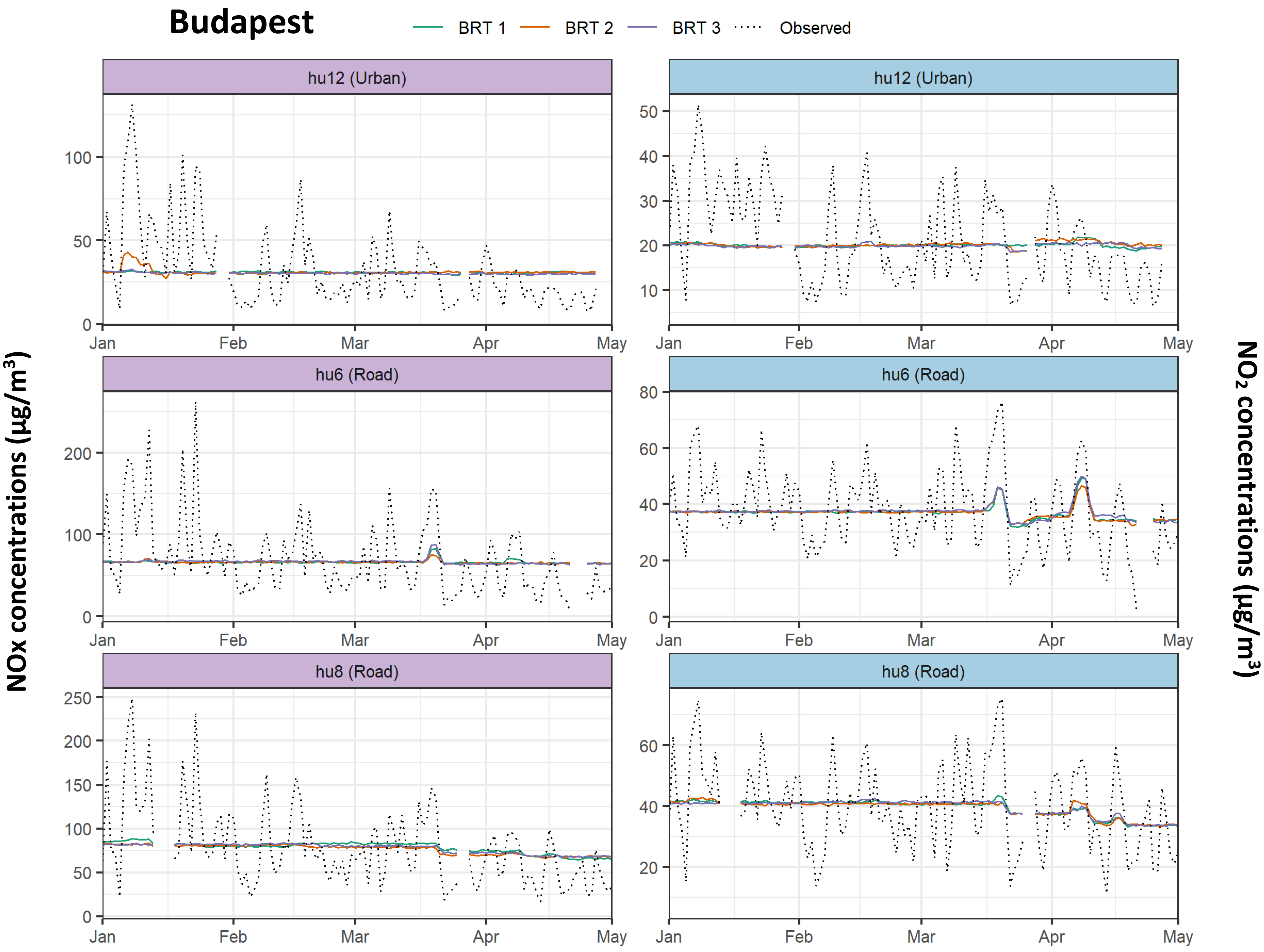

Figure A3.6: Daily mean NOx and NO_2 (µg/m^3) concentrations in Budapest

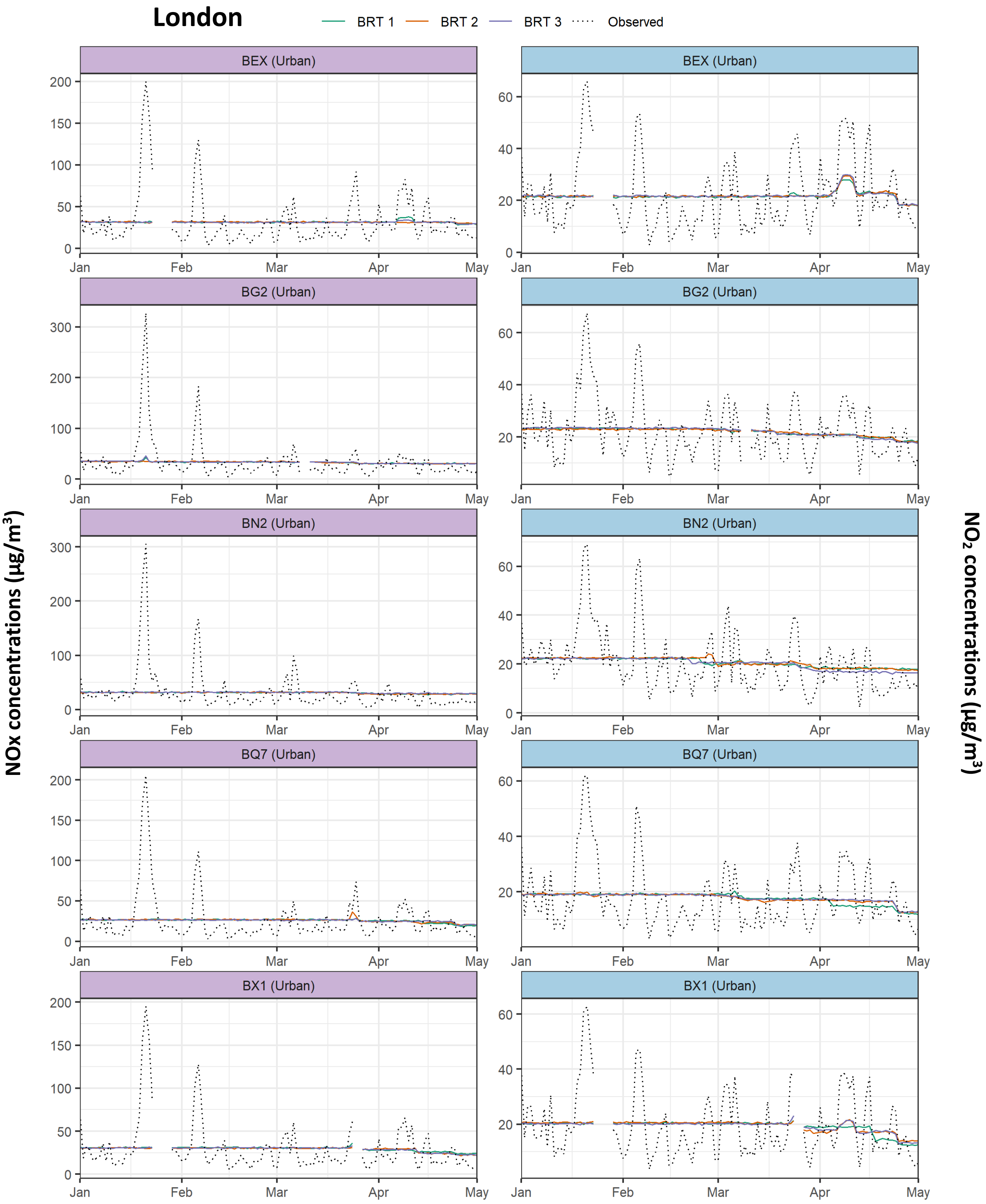

Figure A3.7: Daily mean NOx and NO_2 (µg/m³) concentrations in London

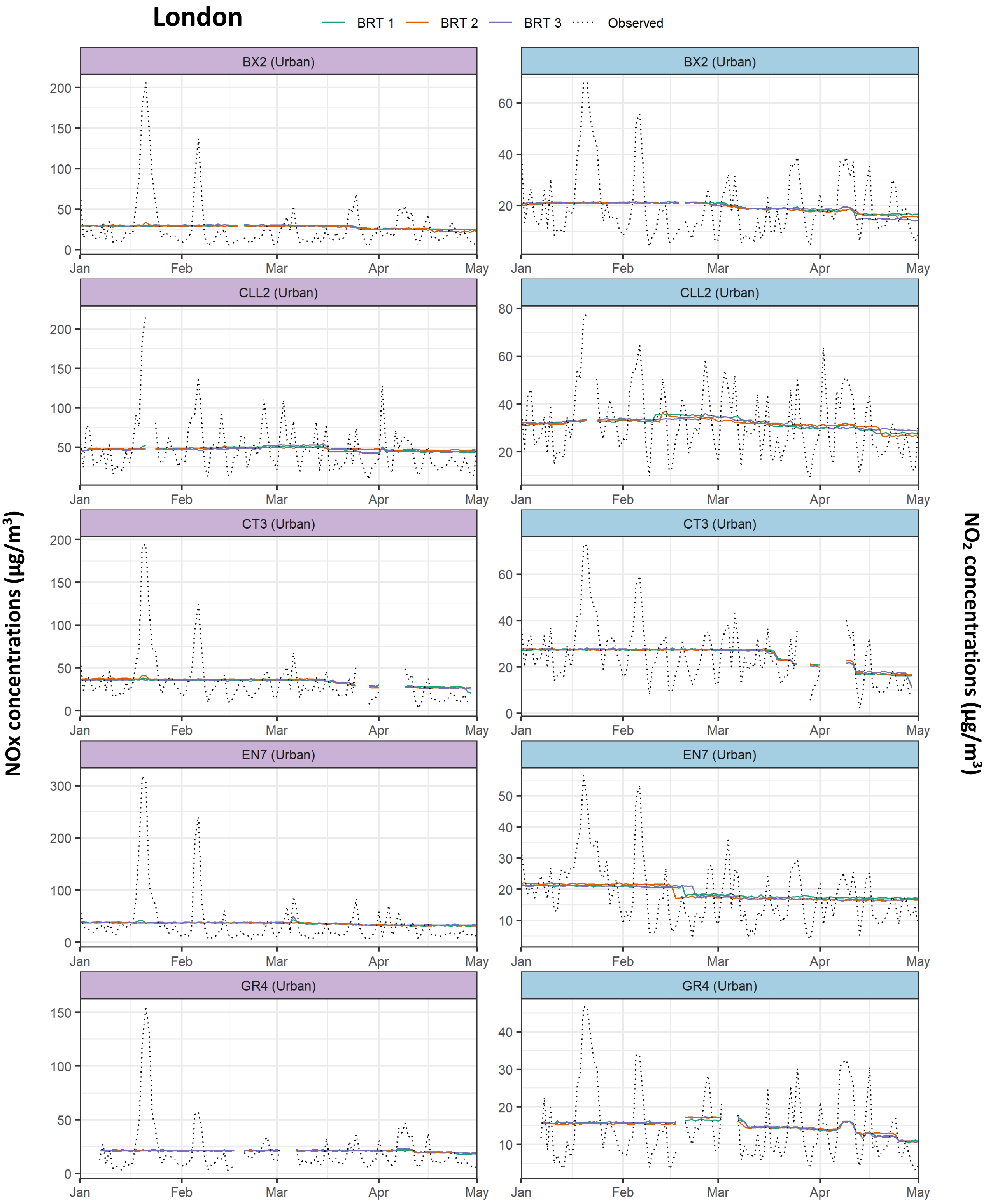

Figure A3.8: Daily mean NOx and NO_2 (µg/m^3) concentrations in London

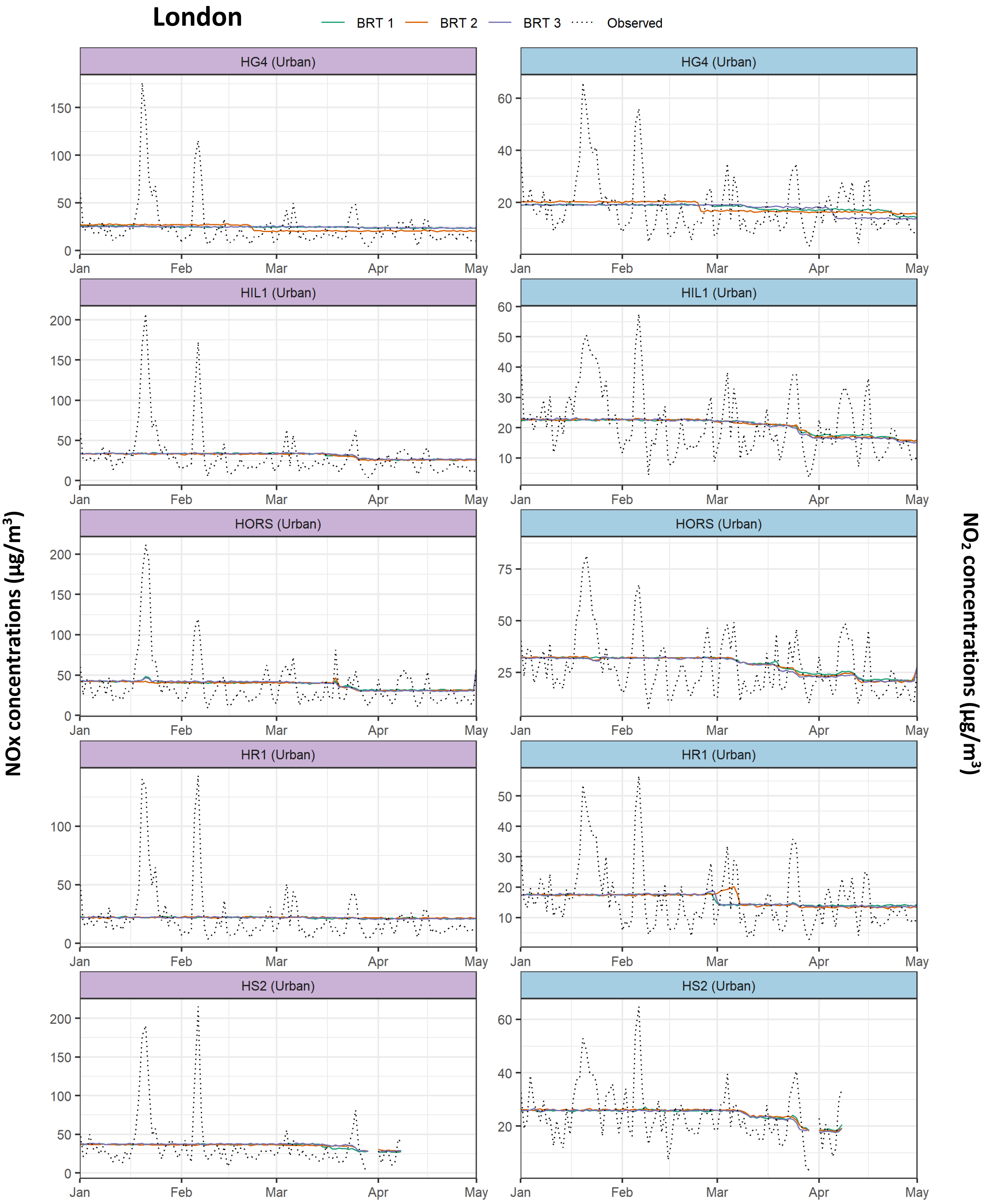

Figure A3.9: Daily mean NOx and NO_2 ($\mu g/m^3$) concentrations in London

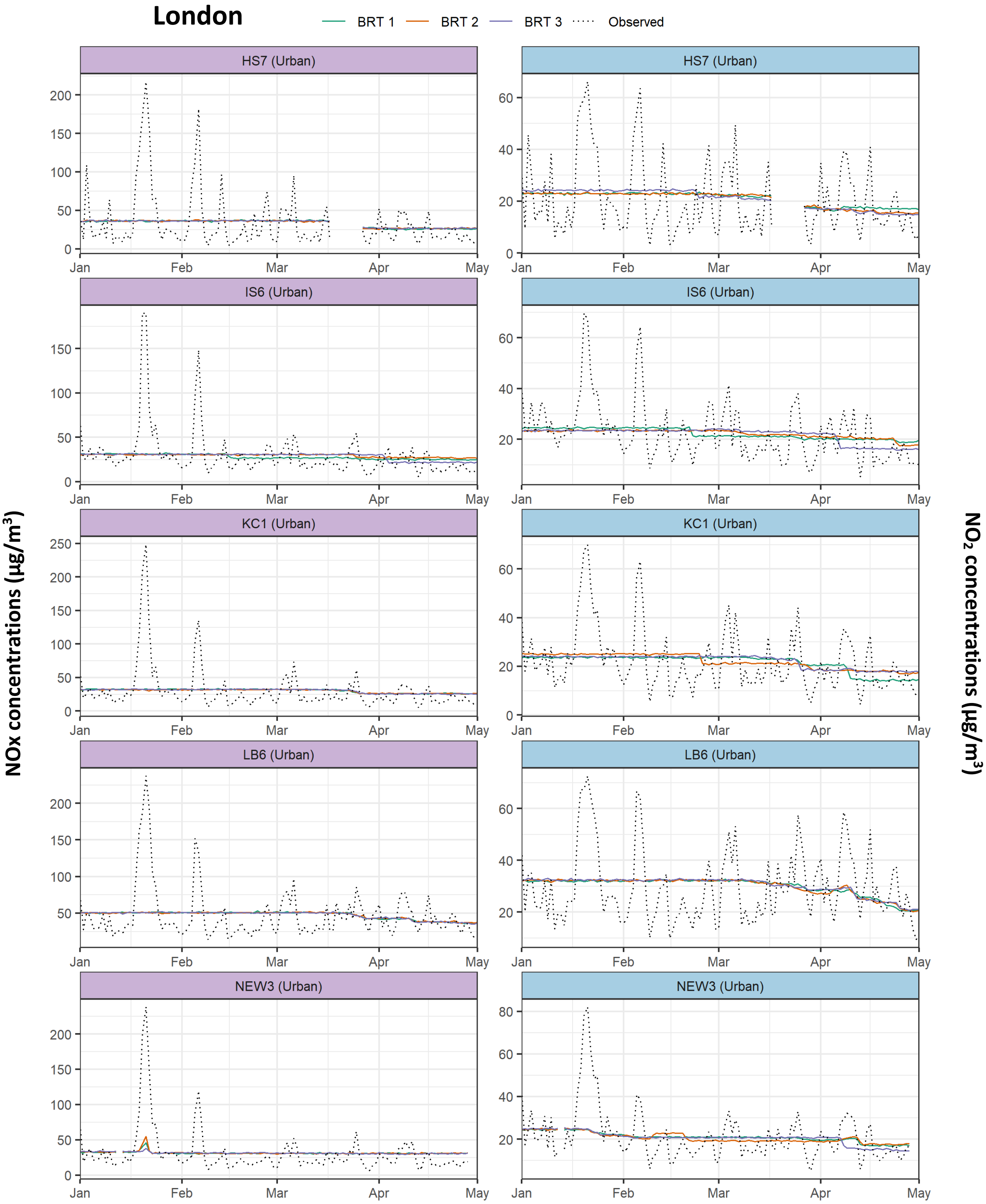

Figure A3.10: Daily mean NOx and NO_2 (µg/m³) concentrations in London

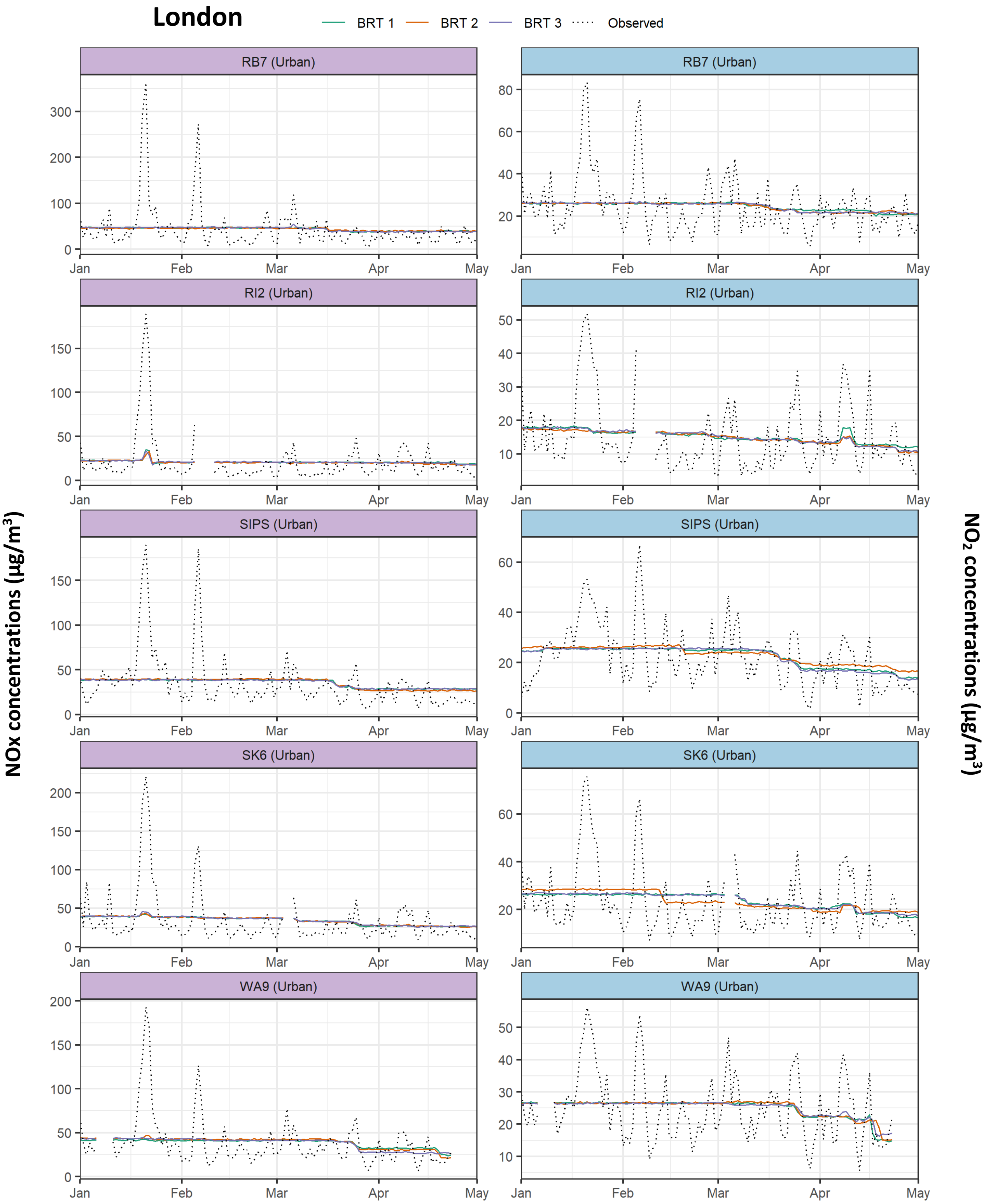

Figure A3.11: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in London

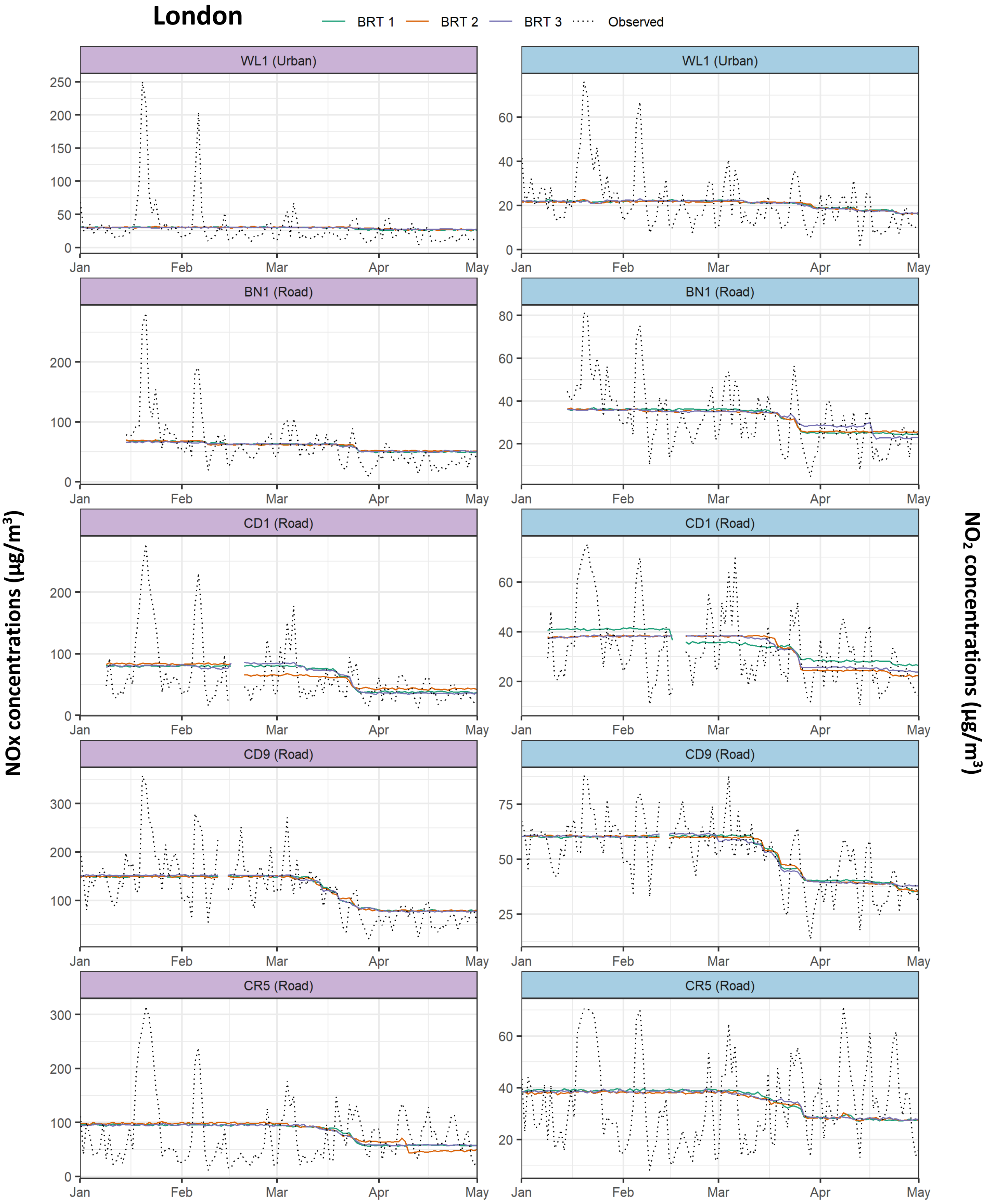

Figure A3.12: Daily mean NOx and NO_2 (µg/m^3) concentrations in London

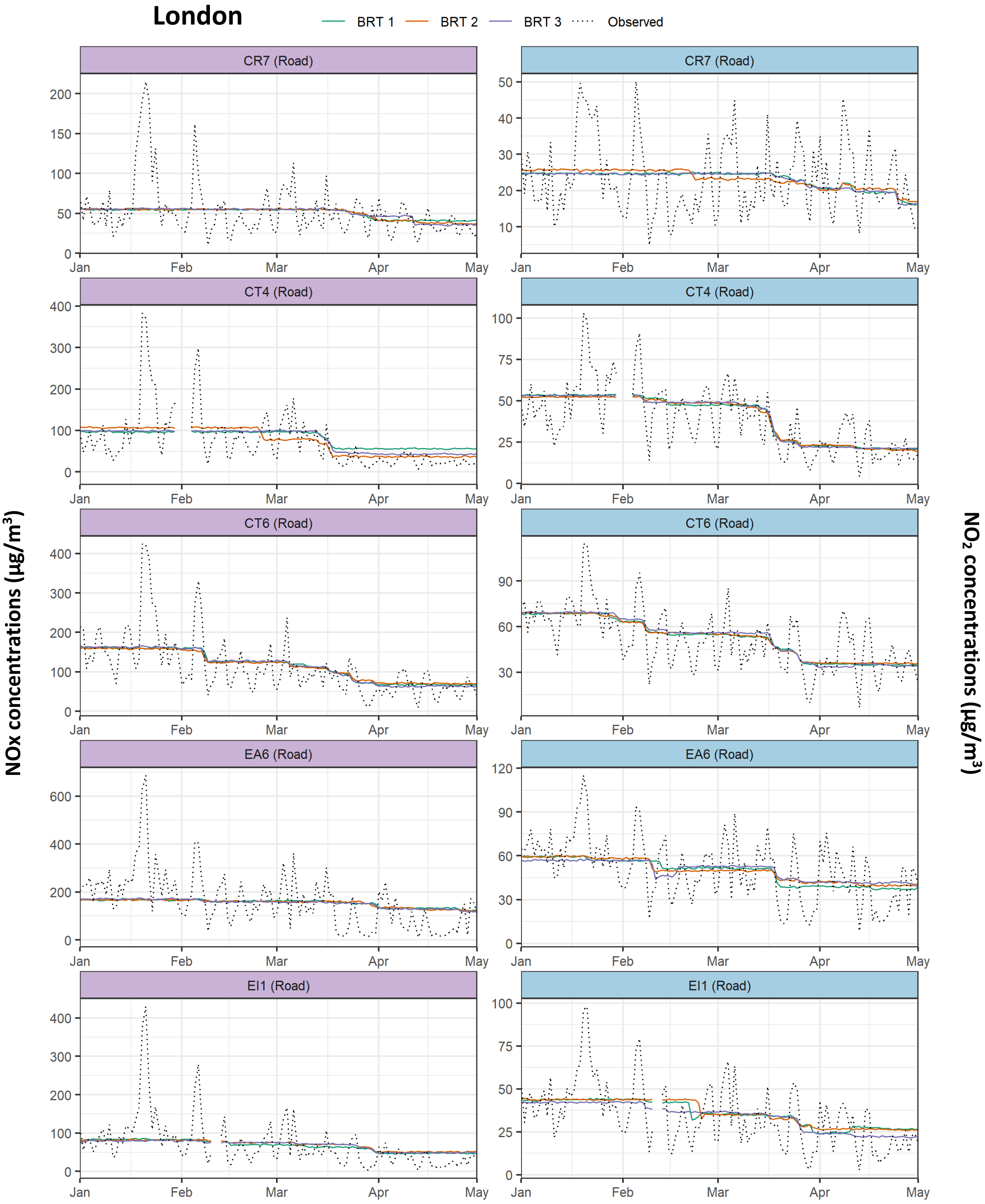

Figure A3.13: Daily mean NOx and NO_2 ($\mu g/m^3$) concentrations in London

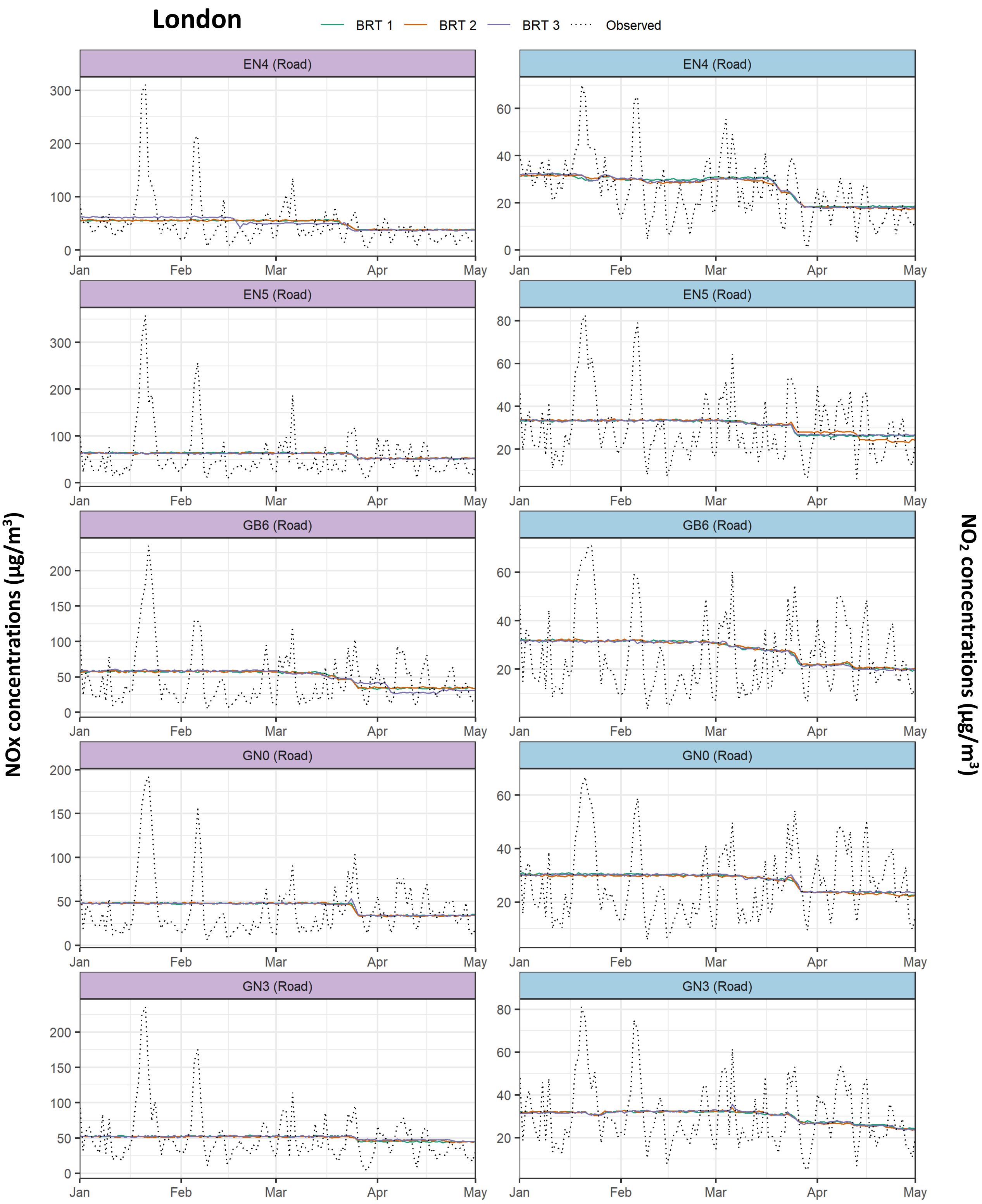

Figure A3.14: Daily mean NOx and NO_2 (µg/m^3) concentrations in London

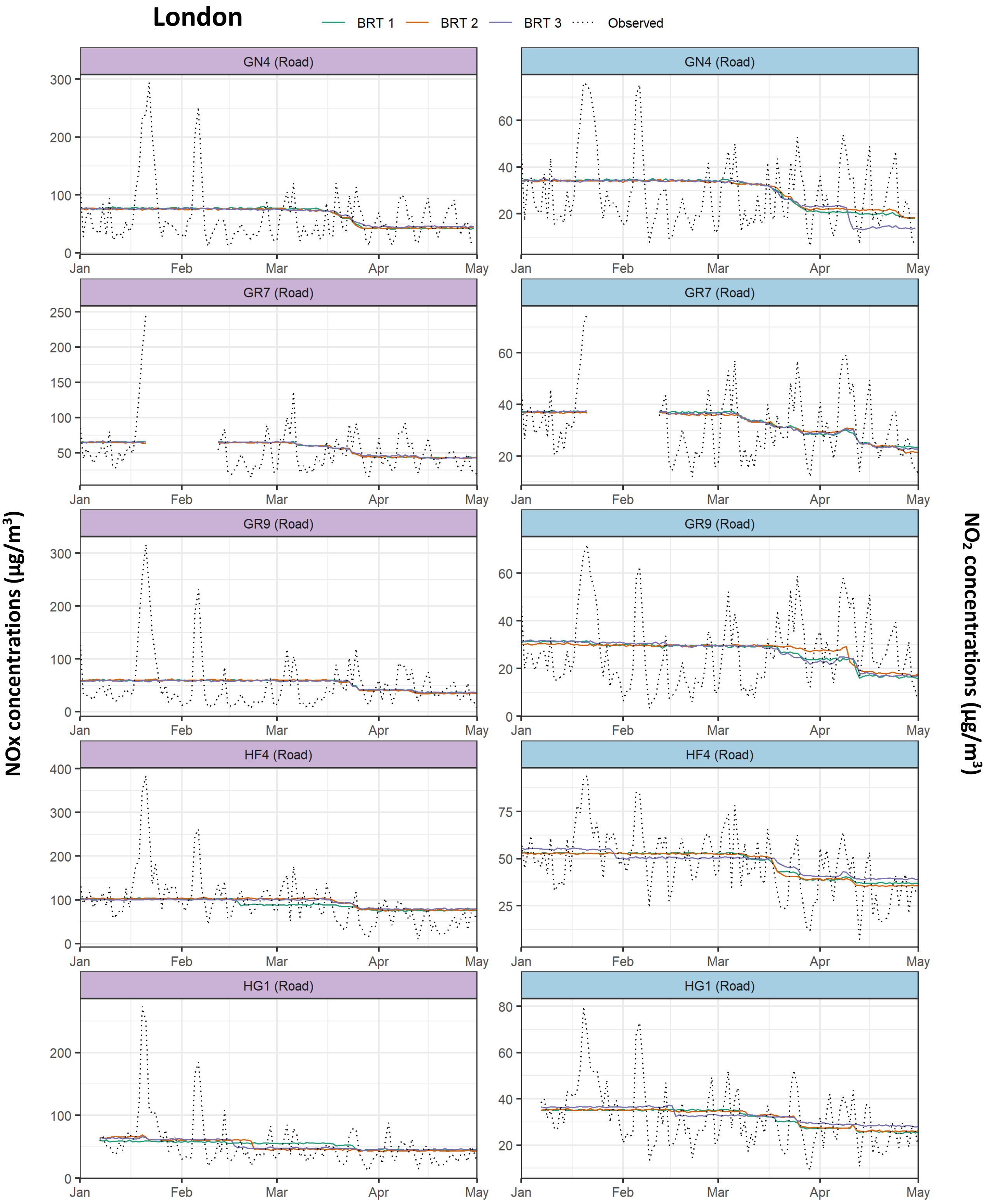

Figure A3.15: Daily mean NOx and NO_2 ($\mu g/m^3$) concentrations in London

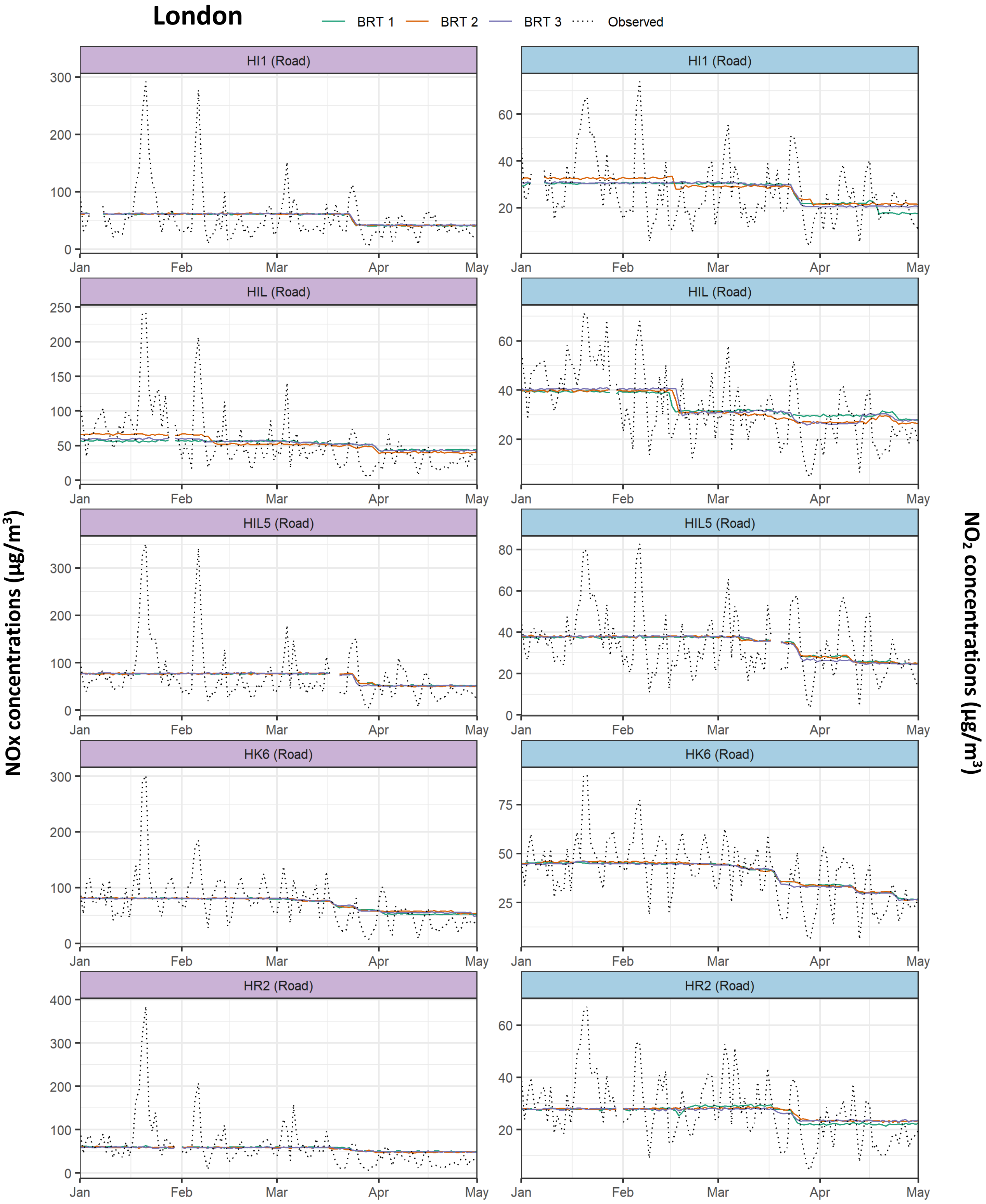

Figure A3.16: Daily mean NOx and NO_2 ($\mu g/m^3$) concentrations in London

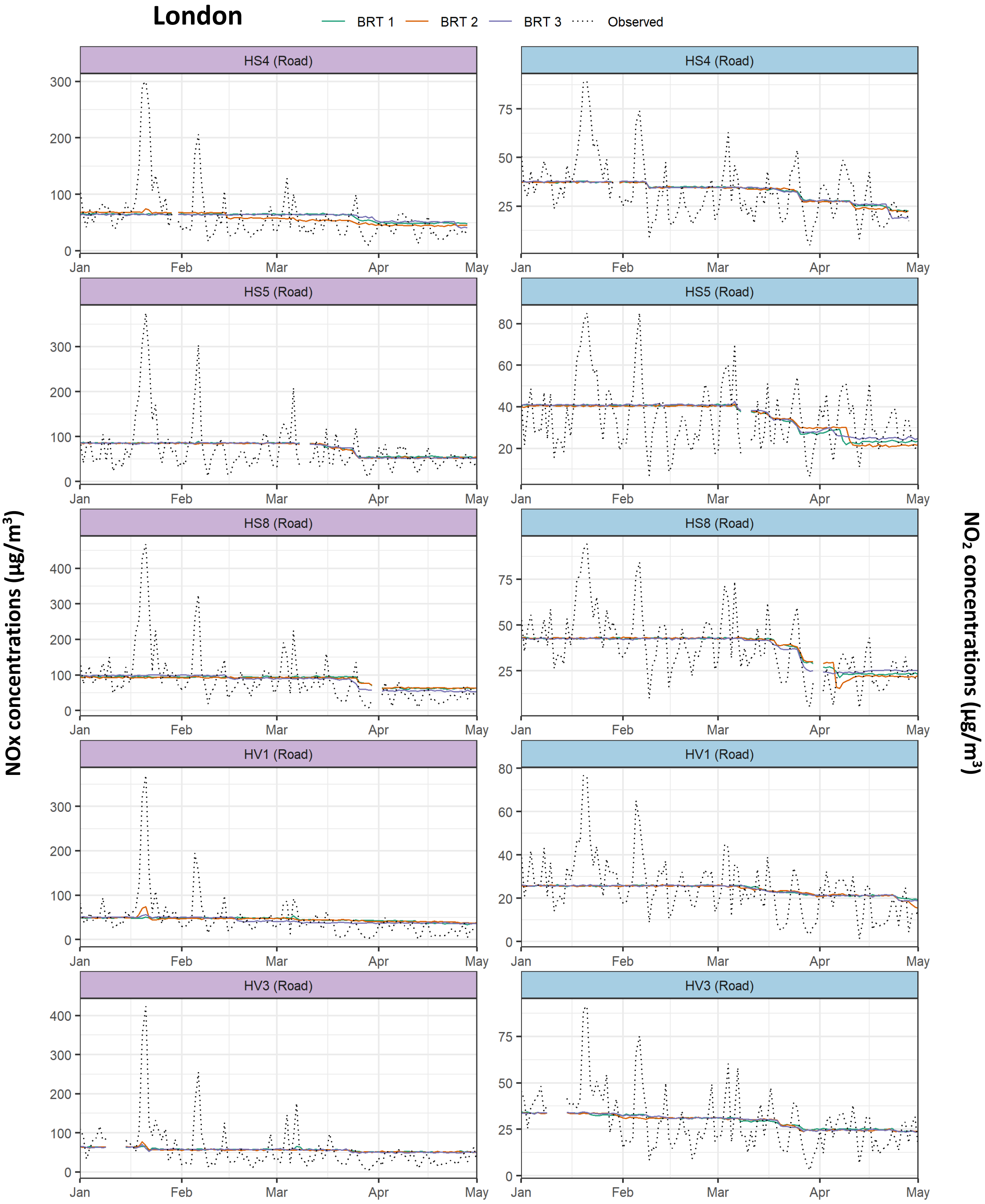

Figure A3.17: Daily mean NOx and NO_2 (µg/m^3) concentrations in London

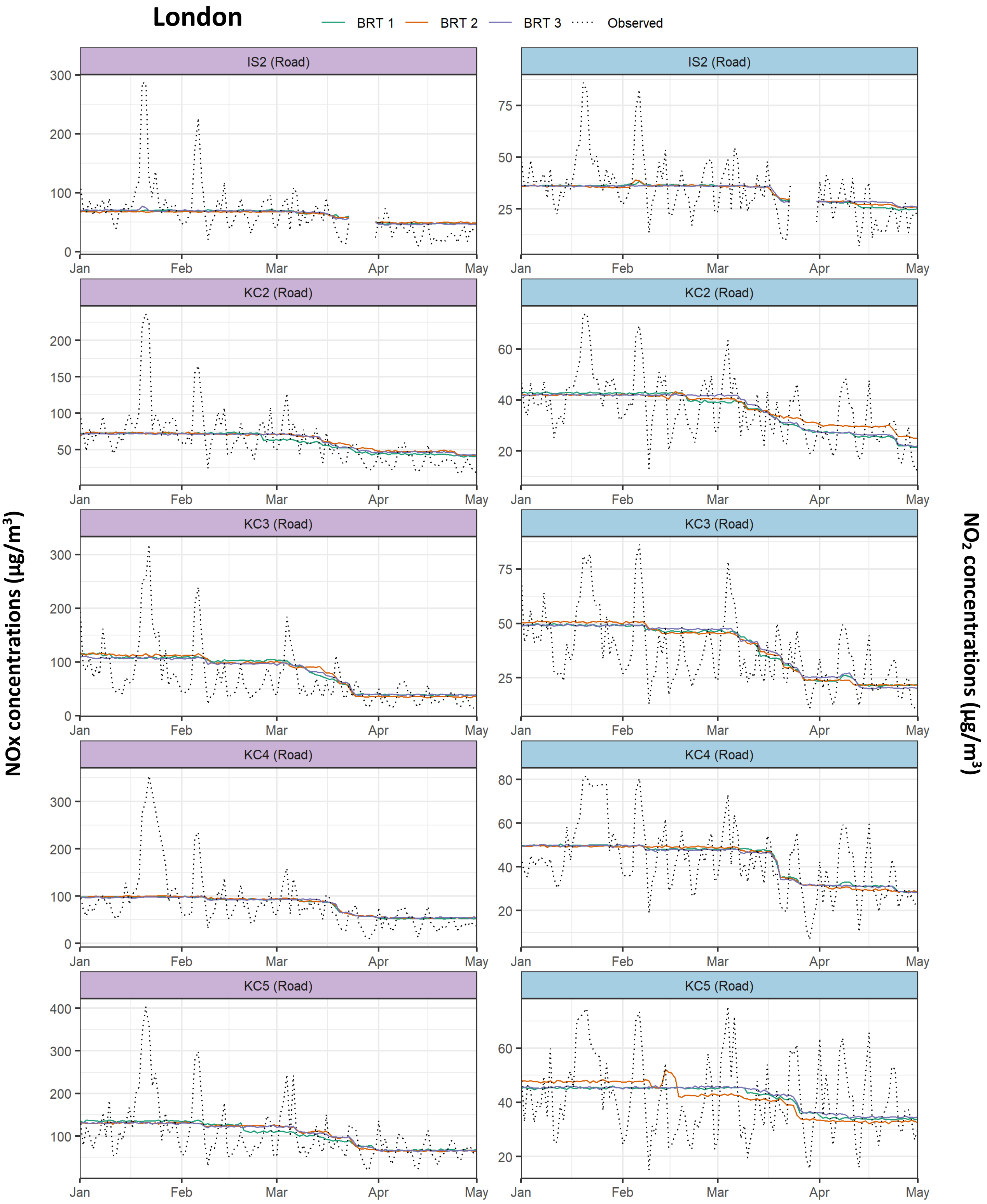

Figure A3.18: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in London

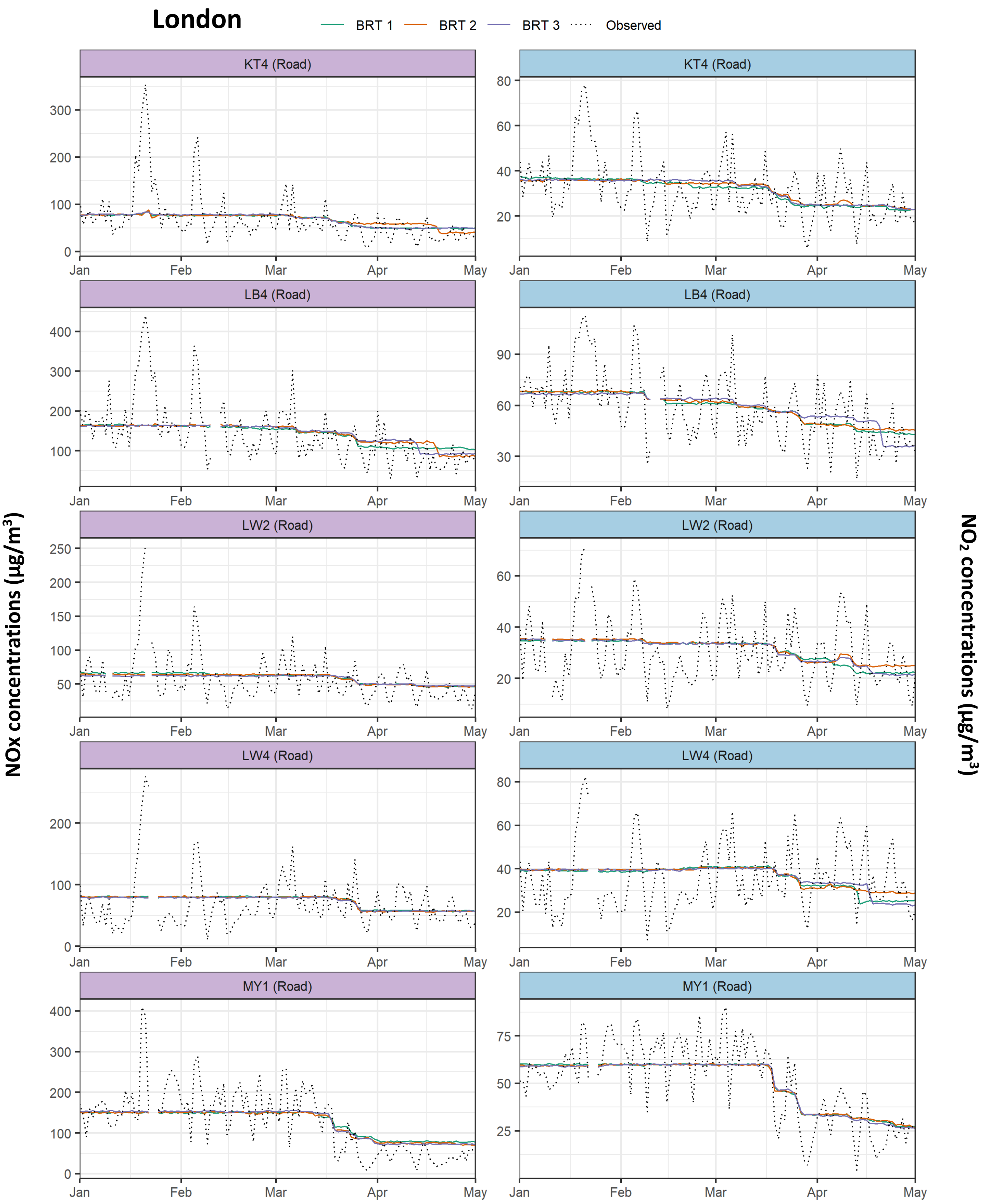

Figure A3.19: Daily mean NOx and NO_2 (µg/m³) concentrations in London

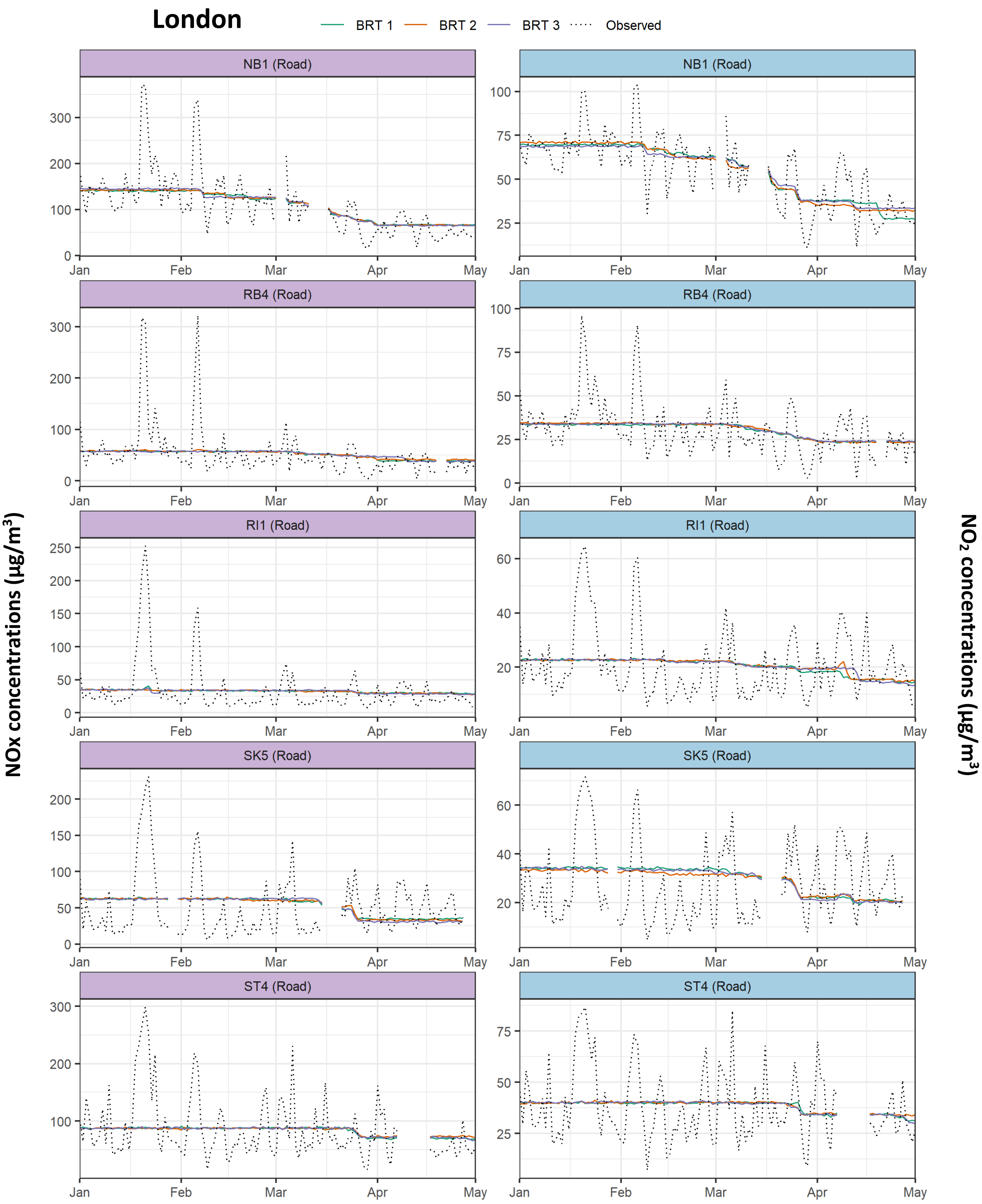

Figure A3.20: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in London

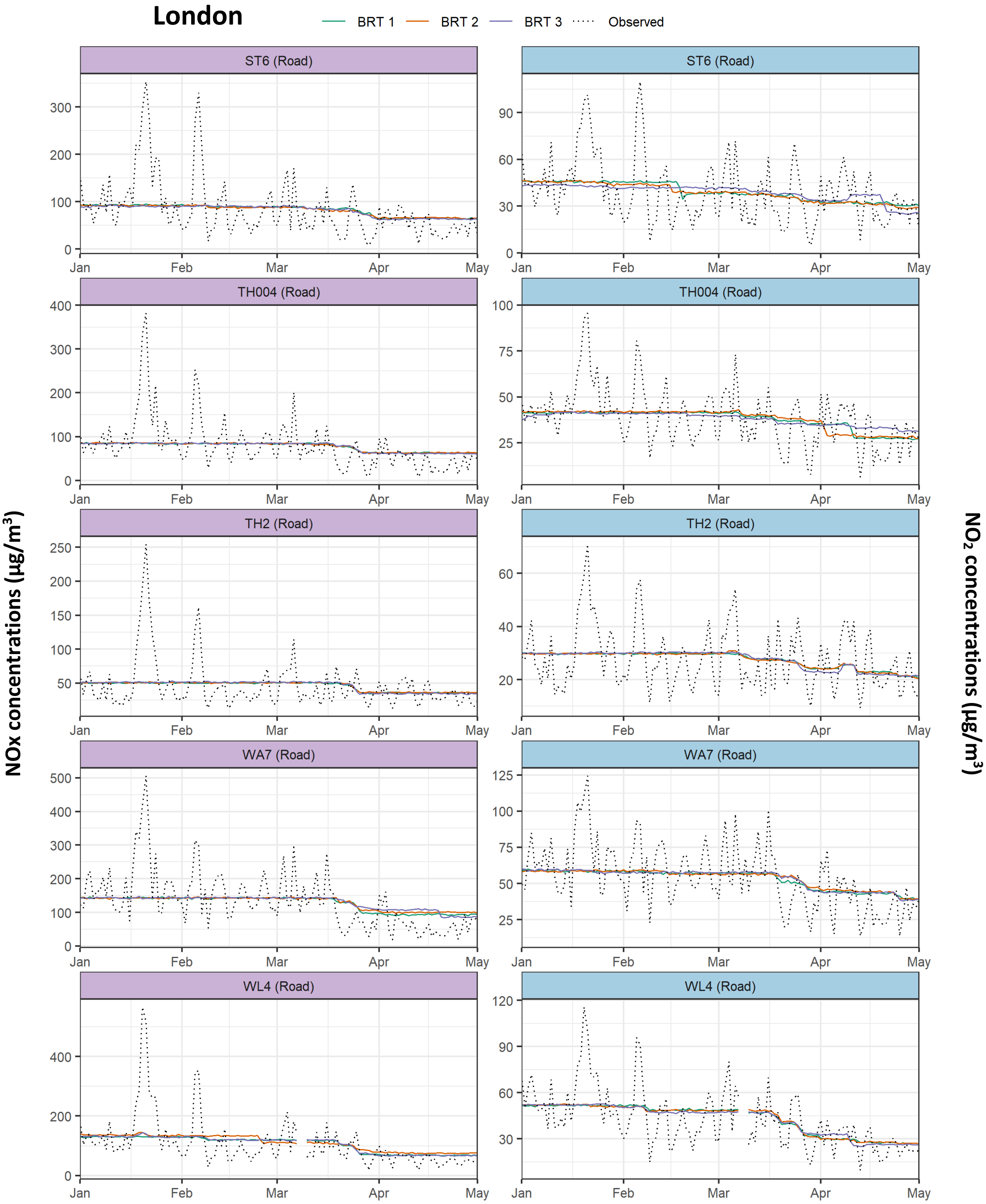

Figure A3.21: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in London

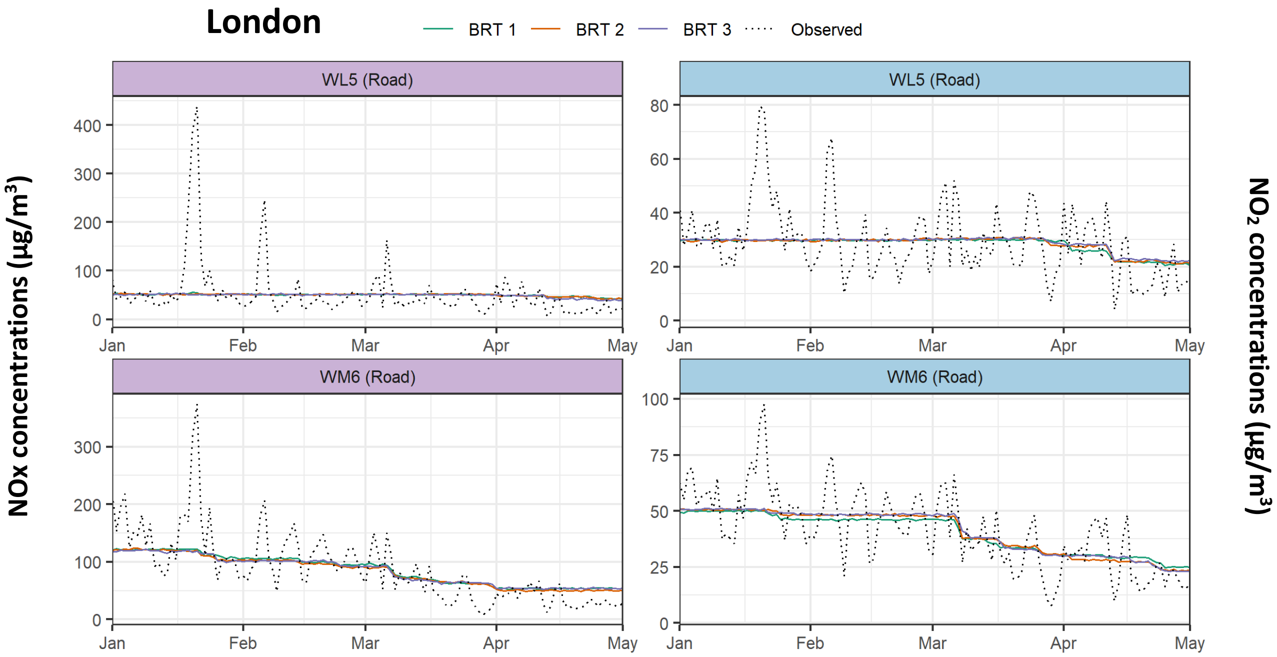

Figure A3.22: Daily mean NOx and NO_2 ($\mu g/m^3$) concentrations in London

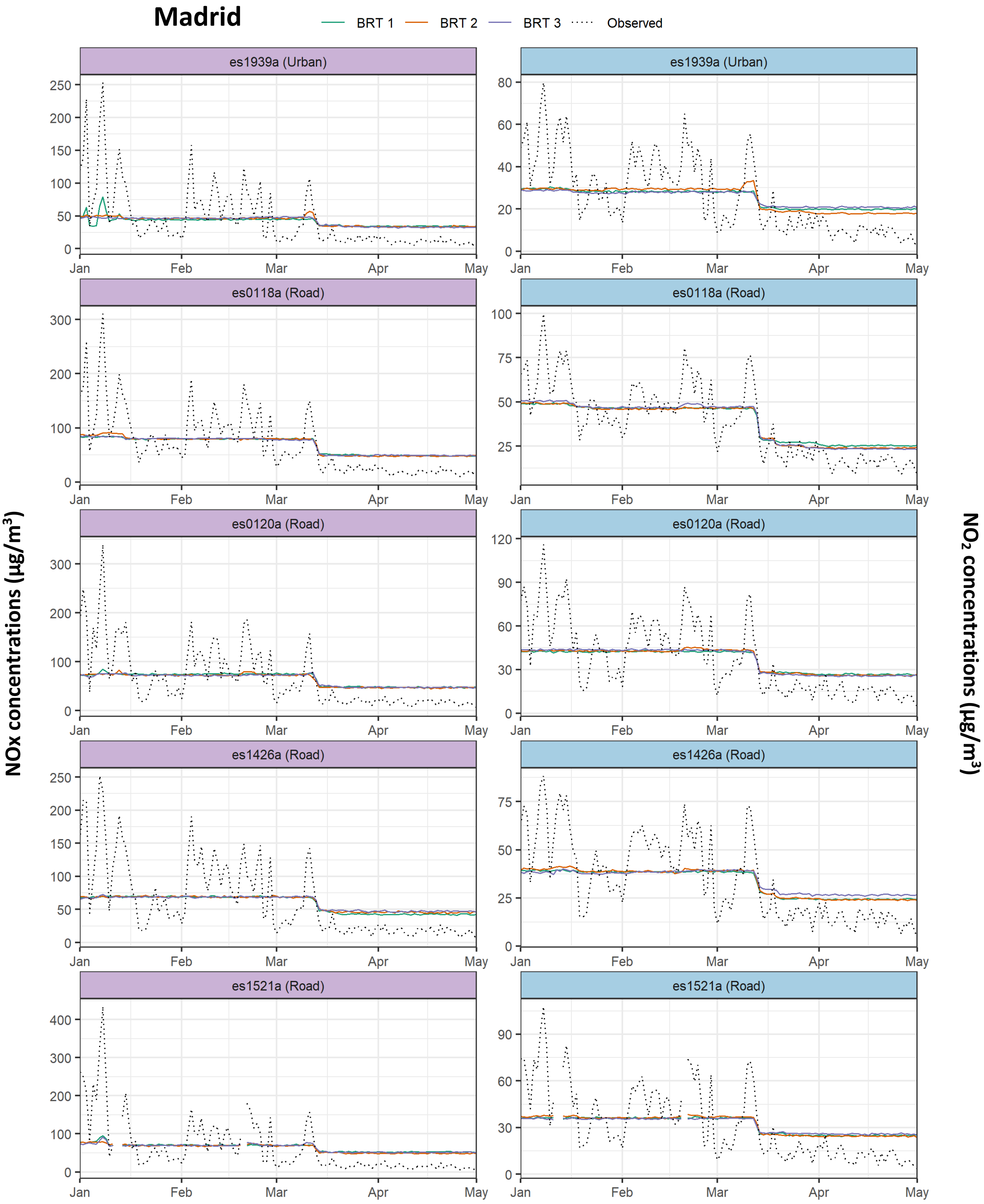

Figure A3.23: Daily mean NOx and NO_2 (μg/m^3) concentrations in Madrid

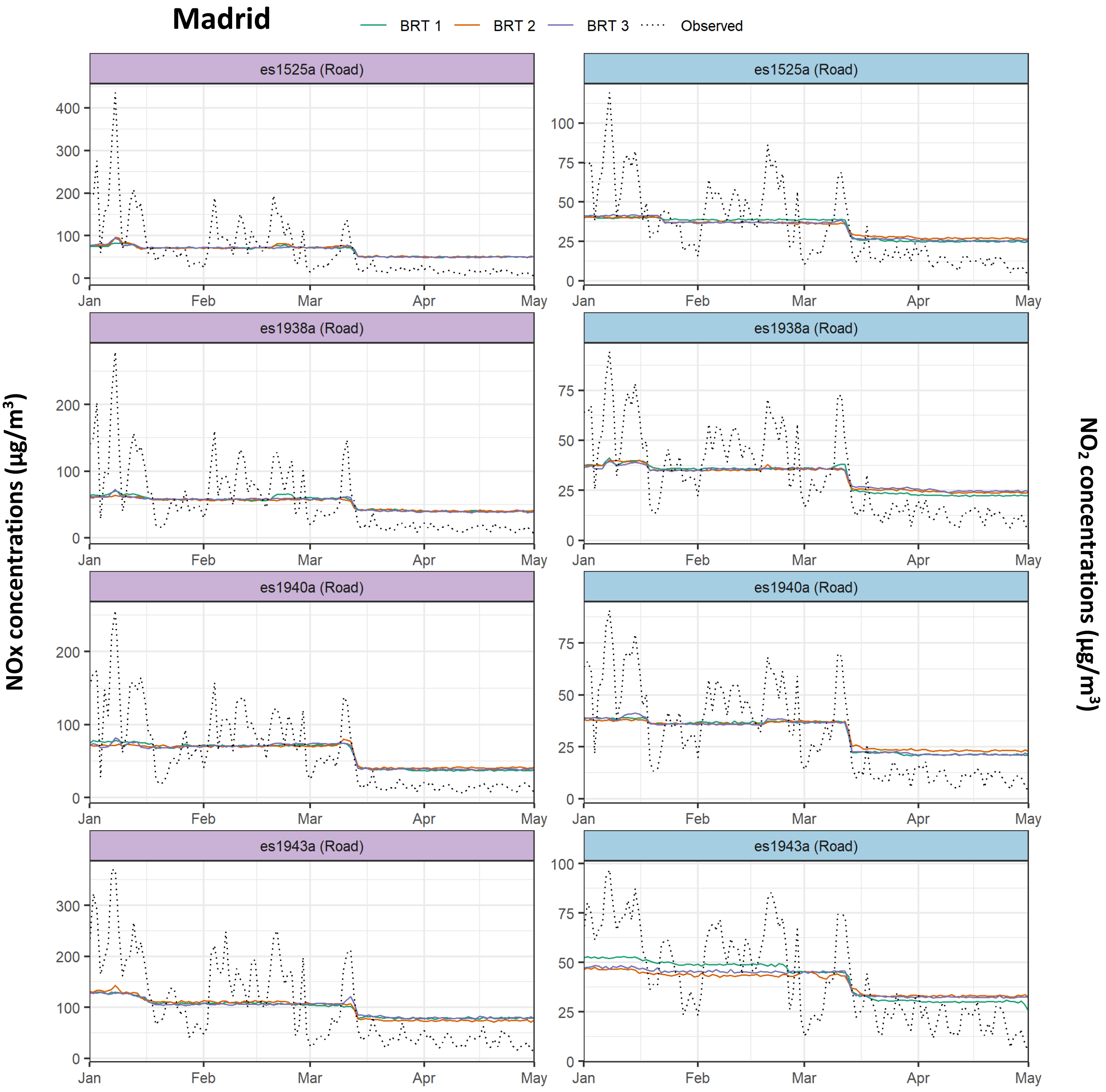

Figure A3.24: Daily mean NOx and NO$_2$ (µg/m^3) concentrations in Madrid

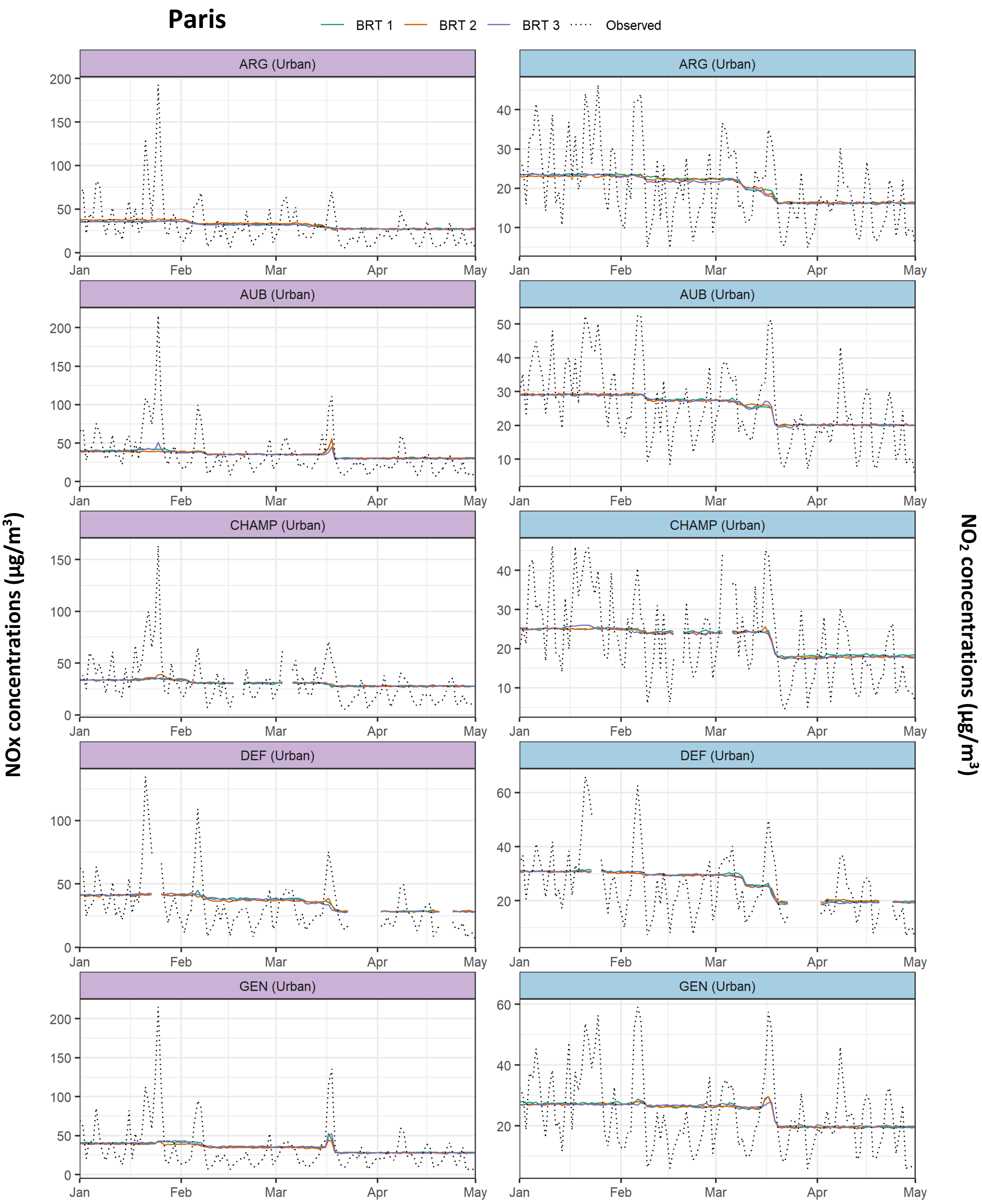

Figure A3.25: Daily mean NOx and NO_2 (µg/m^3) concentrations in Paris

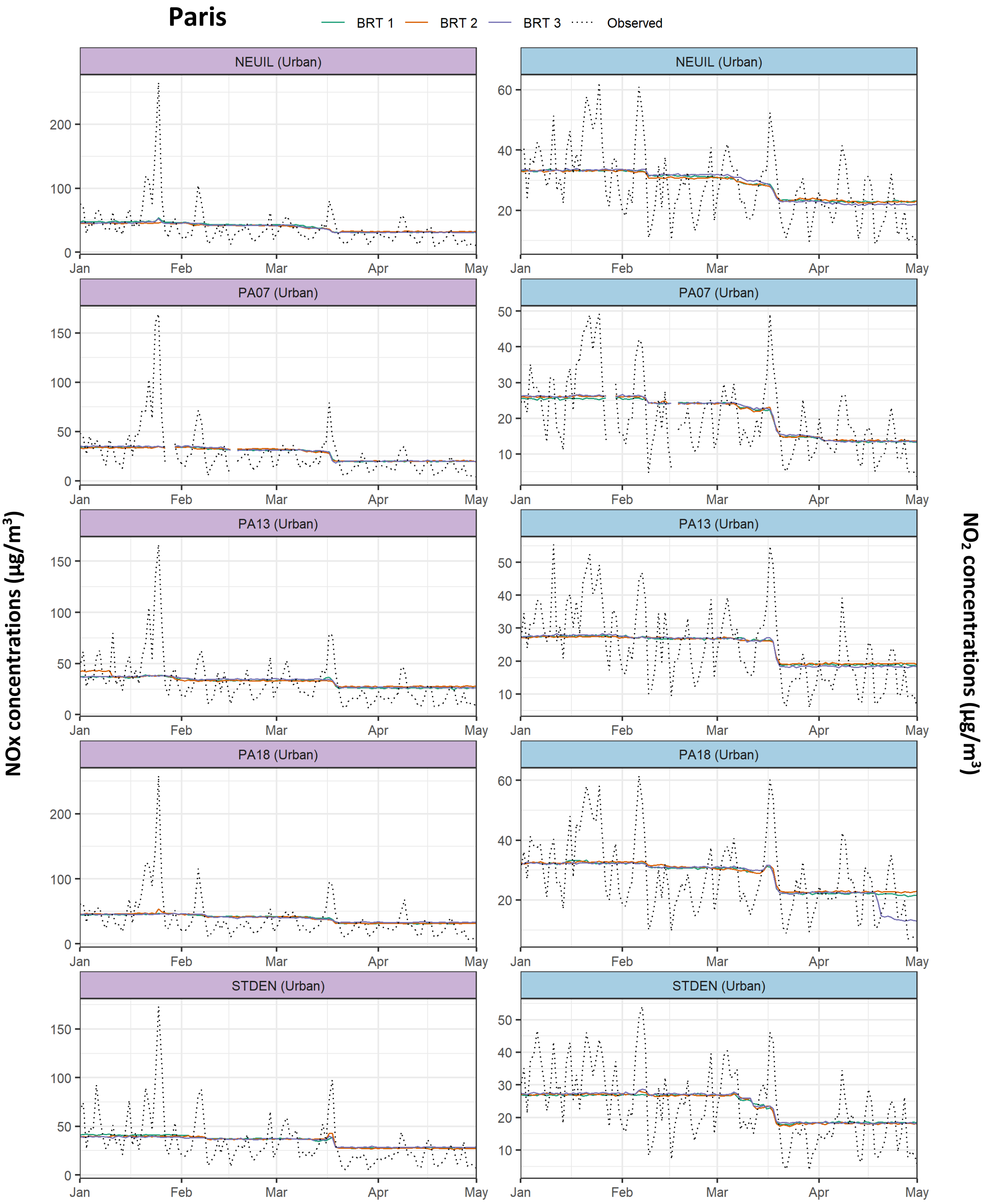

Figure A3.26: Daily mean NOx and NO_2 (µg/m³) concentrations in Paris

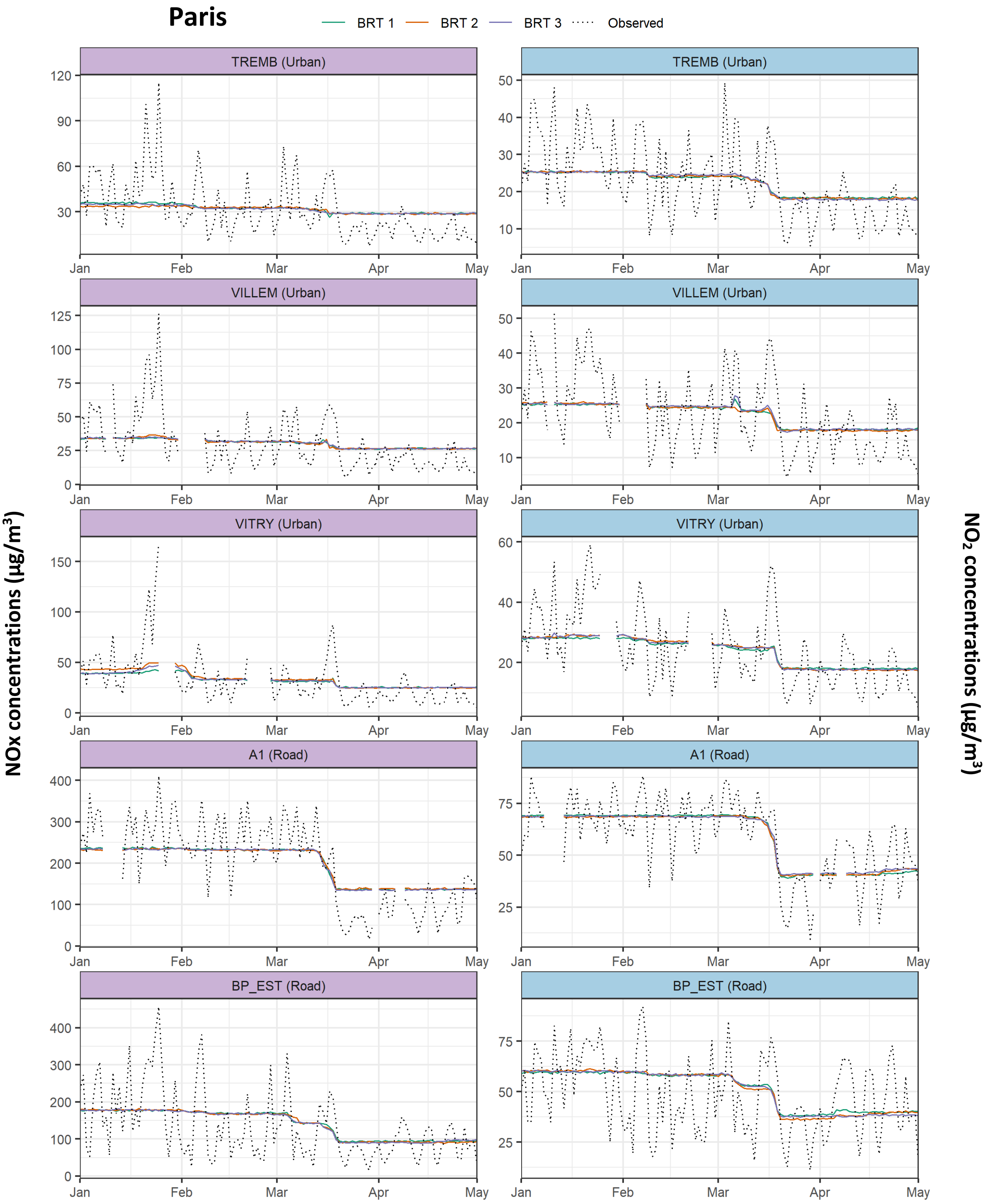

Figure A3.27: Daily mean NOx and NO_2 (µg/m^3) concentrations in Paris

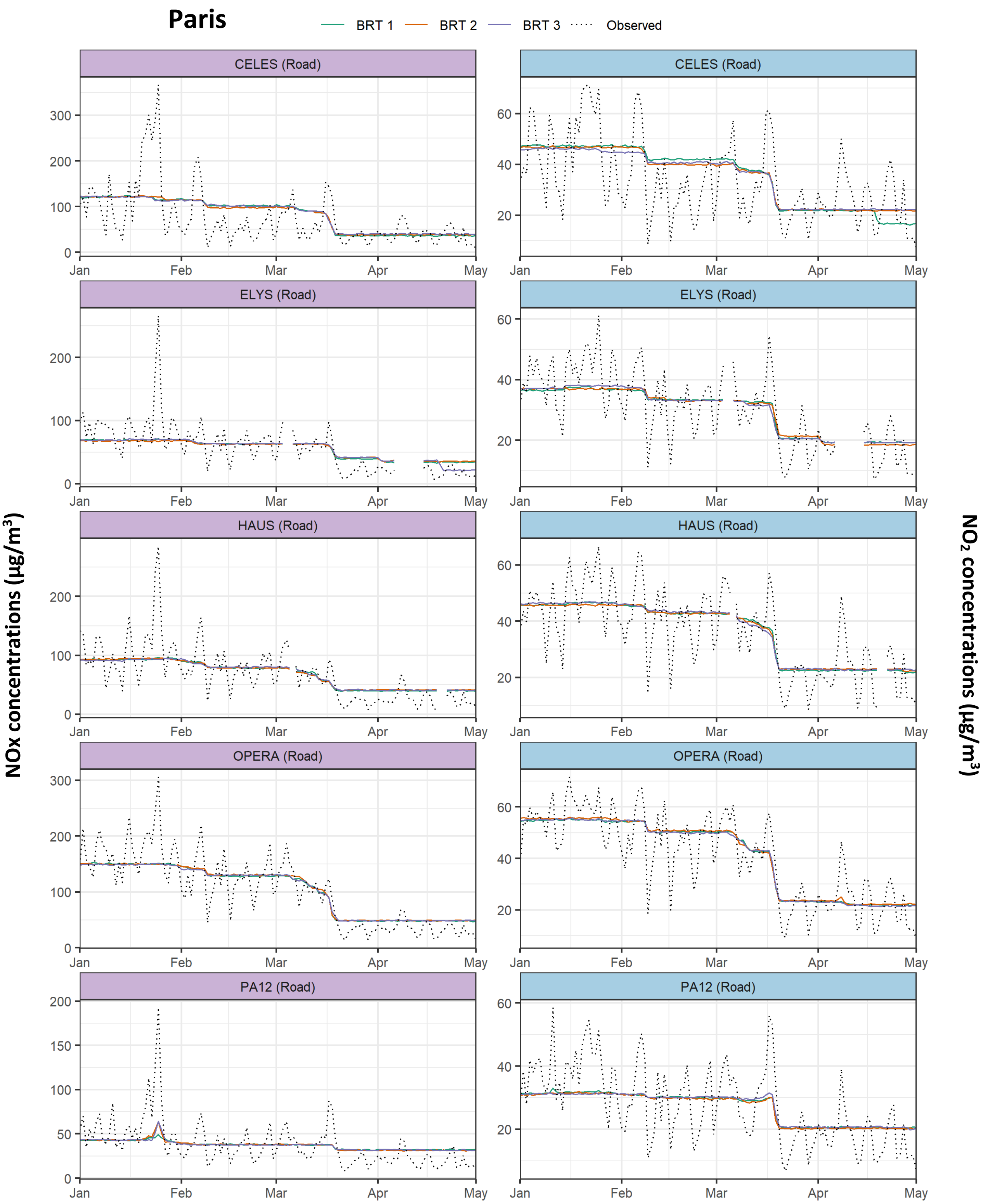

Figure A3.28: Daily mean NOx and NO_2 (µg/m³) concentrations in Paris

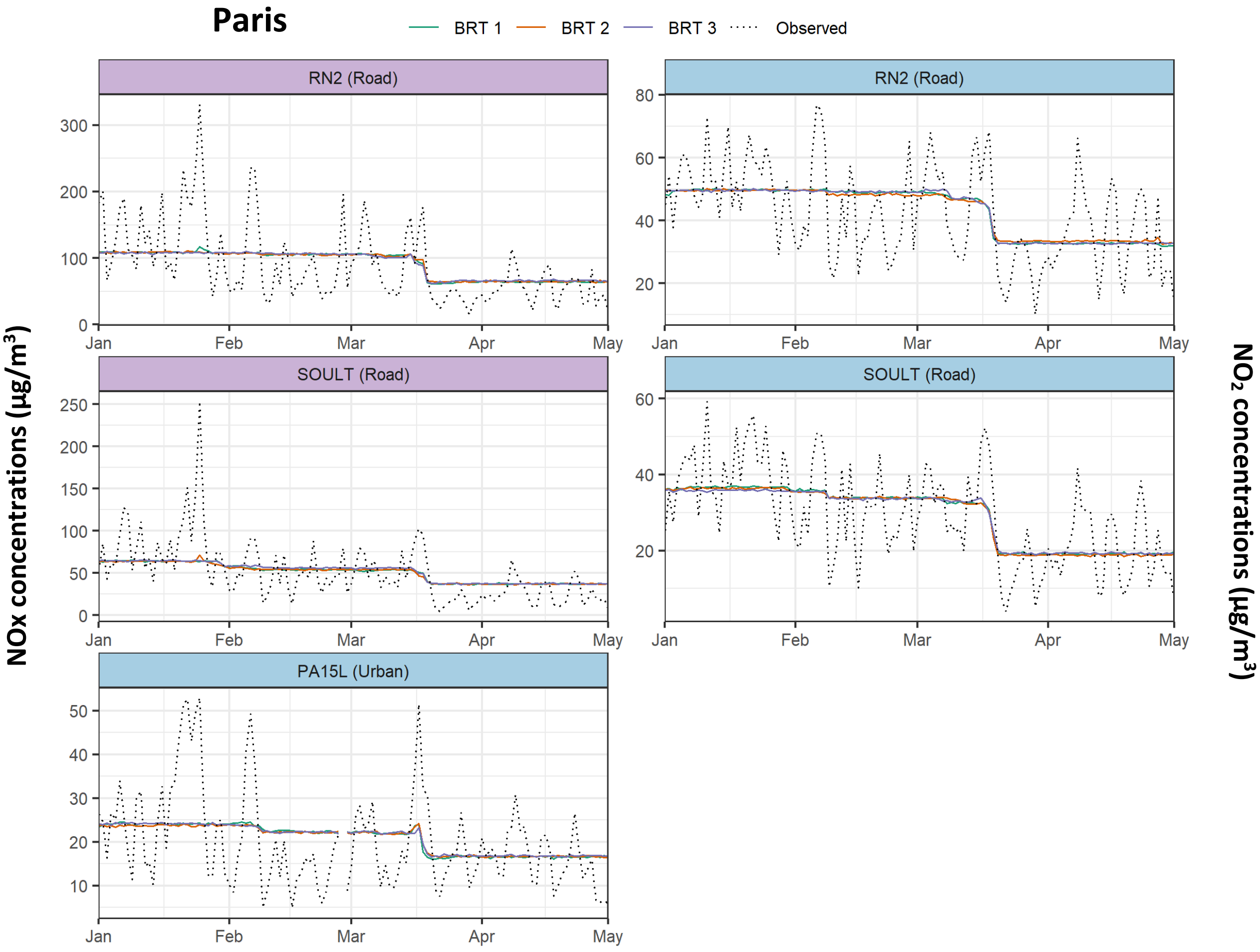

Figure A3.29: Daily mean NOx and NO_2 (µg/m^3) concentrations in Paris

A4 Professional Experience

Stephen Moorcroft, BSc (Hons) MSc DIC CEnv MIEnvSc MIAQM

Mr Moorcroft is a Director of Air Quality Consultants, and has worked for the company since 2004. He has over 35 years' postgraduate experience in environmental sciences. Prior to joining Air Quality Consultants, he was the Managing Director of Casella Stanger, with responsibility for a business employing over 100 staff and a turnover of £12 million. He also acted as the Business Director for Air Quality services, with direct responsibility for a number of major Government projects. He has considerable project management experience associated with Environmental Assessments in relation to a variety of development projects, including power stations, incinerators, road developments and airports, with particular experience related to air quality assessment, monitoring and analysis. He has contributed to the development of air quality management in the UK, and has been closely involved with the LAQM process since its inception. He has given expert evidence to numerous public inquiries, and is frequently invited to present to conferences and seminars. He is a Member of the Institute of Air Quality Management.

Dr Ben Marner, BSc (Hons) PhD CSci MIEnvSc MIAQM

Dr Marner is a Technical Director with AQC and has over 20 years' experience in the field of air quality. He has been responsible for air quality and greenhouse gas assessments of road schemes, rail schemes, airports, power stations, waste incinerators, commercial developments and residential developments in the UK and abroad. He has been an expert witness at several public inquiries, where he has presented evidence on health-related air quality impacts, the impacts of air quality on sensitive ecosystems, and greenhouse gas impacts. He has extensive experience of using detailed dispersion models, as well as contributing to the development of modelling best practices. Dr Marner has arranged and overseen air quality monitoring surveys, as well as contributing to Defra guidance on harmonising monitoring methods. He has been responsible for air quality review and assessments on behalf of numerous local authorities. He has also developed methods to predict nitrogen deposition fluxes on behalf of the Environment Agency, provided support and advice to the UK Government's air quality review and assessment helpdesk, Transport Scotland, Transport for London, and numerous local authorities. He is a Member of the Institute of Air Quality Management and a Chartered Scientist. Dr Marner is a member of Defra's Network of Evidence Experts and is a member of Defra's Air Quality Expert Group.

Tim Williamson, BSc (Hons) MSc MIEnvSci MIAQM

Mr Williamson has 25 years' experience in environmental policy support, development and analysis, mainly in air quality but also covering climate change and resource efficiency. He has broad experience of the field, having held positions in the public and private sectors, and for an environmental NGO, Environmental Protection UK. Tim has worked at the national level, leading multi-disciplinary evidence teams on air quality and, latterly, resource efficiency in Defra for 11 years. He has also worked both for and with local authorities, covering Local Air Quality Management and carbon reduction programmes. Tim has a strong track record in international work, having been involved in EU policy development and on projects supporting both the European Commission and European Environment Agency, and Governments in several parts of the world. He is a Member of the Institute of Air Quality Management and is a Chartered Scientist.

Ricky Gellatly, BSc (Hons) CSci MIEnvSc MIAQM

Mr Gellatly is a Principal Consultant with AQC with over eight years' relevant experience. He has undertaken air quality assessments for a wide range of projects, assessing many different pollution sources using both qualitative and quantitative methodologies, with most assessments having included dispersion modelling (using a variety of models). He has assessed road schemes, airports, energy from waste facilities, anaerobic digesters, poultry farms, urban extensions, rail freight interchanges, energy centres, waste handling sites, sewage works and shopping and sports centres, amongst others. He also has experience in ambient air quality monitoring, the analysis and interpretation of air quality monitoring data, the monitoring and assessment of nuisance odours and the monitoring and assessment of construction dust. He is a Member of the Institute of Air Quality Management and is a Chartered Scientist.

George Chousos, BSc MSc AMIEnvSc AMIAQM

Mr Chousos is an Assistant Consultant with AQC, having joined in May 2019. Prior to joining AQC, he completed an MSc in Air Pollution Management and Control at the University of Birmingham, specialising in air pollution control technologies and management, and data processing using R. He also holds a degree in Environmental Geoscience from the University of Cardiff, where he undertook a year in industry working in the field of photo-catalytic technology. He is now gaining experience in the field of air quality monitoring and assessment.