An analysis of the impact of COVID-19 restrictions on air quality in the Wellington region

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Executive summary

The COVID-19 pandemic resulted in restrictions on people's usual activities and travel patterns. This report investigates changes in air quality from March to September 2020 during lockdown and subsequent alert levels compared to a business-as-usual scenario. Air quality measurements from Wellington city (alongside the urban motorway), suburban Upper Hutt and the rural town of Masterton were used. The purpose of the analysis was to examine the usefulness of local air quality monitoring data to detect any temporary or long-term changes in traffic and home heating emissions arising from behavioural change under the different Alert Level restrictions. Furthering our understanding of the relationship between human behaviour, emissions and pollutant concentrations can inform any future management strategies designed to improve air quality and community health. In this study, statistical modelling using machine learning was used to control for the confounding effect of meteorology on air pollutant levels. This method allowed air pollutant levels measured during the Alert Level periods to be compared to levels that would have occurred under business-as-usual (BAU) travel and home heating patterns.

The study highlighted that differences in air quality from BAU varied by pollutant and by monitoring site due to local differences in emission source contributions from traffic, home heating and natural sources. During Level 4 lockdown directly emitted traffic exhaust pollutants (oxides of nitrogen and black carbon) were reduced by 71% at the Wellington city roadside monitoring site commensurate with the approximately 80% reduction in state highway volumes. Reductions in nitrogen dioxide (a harmful regulated pollutant) were much lower than the other traffic-related air pollutants and rebounded to higher than expected levels in Level 1 despite no significant increase in state highway traffic compared to the previous year.

Particulate matter concentrations at the roadside were less affected during Level 4 showing a 17% decrease attributed to lower levels of road dust and traffic exhaust emissions as marine aerosol (sea salt) is estimated to make up at least 30% of particulate matter and this contribution is unaffected by changes in travel patterns.

Masterton had a 30% increase in $PM_{2.5}$ during Alert Level 2 compared to previous years which was attributed to a combination of more stay at home behaviour and therefore higher use of solid fuel heating combined with a greater frequency of low wind speed periods compared to previous years. Upper Hutt also appeared to have an increased wood burning contribution due to much higher than expected overnight carbon monoxide levels from L3 to L1, although there was not a noticeable increase in winter PM10 levels despite lower than average wind speed.

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1. Introduction

1.1 Purpose and context

In 2020, movement restrictions and disruptions arising from NZ's response to the COVID-19 pandemic curtailed people's usual activities from March through to June and then again in August. These restrictions provided a unique opportunity to evaluate how air quality was affected by abrupt changes in traffic-related emissions and any other emission sources. Quantifying how levels of air pollutants associated with different emission sources responded over the COVID-19 travel and social restrictions deepens our understanding of the strengths and limitations of using air quality measurements to infer underlying emissions trends.

Specifically, this research investigated changes in traffic-related air pollution related to changes in traffic intensity during COVID-19 at a roadside in Wellington city. The research also looked at how traffic-related and home-heating related pollutants varied at an urban background site in Upper Hutt and in Masterton, a rural town with high use of solid fuels for home heating.

Statistical modelling using machine learning was used to control for the impact of meteorology on air pollutant levels. This method allowed air pollutant levels measured during the Alert Level periods to be compared to levels that would have occurred under business-as-usual (BAU) travel and home heating patterns by removing the confounding effects of meteorology.

1.2 Study objectives

- Investigate any changes in air quality measured at a city roadside, an urban residential and in a rural town during Alert Level 4 "lockdown" and subsequent Alert Levels.

- Apply a machine learning statistical technique to account for influence of meteorology on observed air pollution levels during the periods that Alert Levels were in force.

- Assess usefulness of local air quality monitoring data to detect any temporary or long-term changes in traffic and home heating emissions arising from changes in people's travel patterns.

2. Background

2.1 Traffic-related air pollutants

Traffic-related air pollution is a complex mixture of gases and particles with differing chemical, physical and toxicological properties. Exposure to traffic-related air pollutants are associated with exacerbation of asthma, respiratory symptoms, impaired lung function, and increased risk of cardiovascular disease and premature death (Health Effects Institute, 2010). Key indicators of traffic-related air pollution including, nitrogen dioxide, carbon monoxide, black carbon, particulate matter and ozone, are discussed below.

Greater Wellington regional council (GWRC) has a traffic-related air quality monitoring programme based on two permanent monitoring sites, a busy roadside in Wellington city and at an urban residential background site in Upper Hutt. These sites are used to track long term trends, investigate relationships between pollutants and to confirm ongoing compliance with national regulatory standards and guidelines (Mitchell, 2020). To extend the spatial coverage of these two monitoring stations, GWRC has progressively established a network of passive diffusion tubes to track trends in annual average nitrogen dioxide (NO_2) as an indicator of trends in regional traffic emissions for future GWRC Regional Land Transport Plan reporting (Mitchell, 2017).

2.1.1 Gaseous traffic-related pollutants

Nitrogen dioxide (NO_2) is a harmful air pollutant derived from fuel combustion. NO_2 levels measured at Wellington central and Upper Hutt monitoring sites meet national guidelines and standards (Mitchell, 2020).

Generally, only a small amount of NO_2 is directly emitted in exhaust emissions. Most NO_2 is formed as a secondary pollutant when nitric oxide (NO) produced during high temperature fuel combustion is rapidly oxidised by ozone (O_3) in outdoor air. The reaction between NO and O_3 continues until either the O_3 is depleted. Once formed NO_2 can then be broken back down to NO the presence of strong sunlight. Collectively NO and NO_2 are known as NOx. Although all motor vehicles emit NOx, heavy-duty (diesel) vehicles emit more NOx per vehicle than petrol-powered vehicles.

Ozone in the upper atmosphere protects us from harmful levels of UV radiation. Some of the naturally occurring ozone from the upper atmosphere is transported to the lower atmosphere where it can be known as the 'background' concentration. Background ozone measured away from roadside sites has a marked seasonal cycle with a winter maxima and summer minima as well as significant day-to-day differences due to synoptic scale meteorological variability[1].

High concentrations ozone can also be formed at ground level through a photochemical reaction involving emissions of NOx and volatile organic compounds (VOCs) usually from traffic sources. This can lead to elevated ozone

[1] https://niwa.co.nz/atmosphere/our-data/trace-gas-plots

downwind of a large area emissions source under meteorological conditions conducive to photochemical activity, including sufficient sunshine, high temperatures and low wind speeds (McKendry, 1996). Fortunately, ground-level ozone concentrations in NZ are generally low relative to the National Environmental Standard for Air Quality and when compared to large overseas cities (Ministry for the Environment and Stats NZ, 2018).

Ozone has been measured at the Wellington central station since late 2017 to provide information to help explain local NO_2 formation. Peak NOx emissions in the morning commuter rush-hour result in the quenching of locally available background ozone as NO is oxidised to NO_2. Therefore, the production of NO_2 from NO is ozone limited. During the weekend, when NOx and NO_2 levels are lower than during weekdays, proportionally more NO_2 is produced for a given level of NOx as there is less ozone quenching (Figure 2.1). Therefore levels of ozone measured in Wellington central roadside are higher in the weekend compared to weekdays. Ozone also shows seasonality with the highest levels in spring which is likely related to stratospheric intrusion, when naturally occurring ozone aloft is transported to ground level during periods of strong winds (Adeeb & Shooter, 2003).

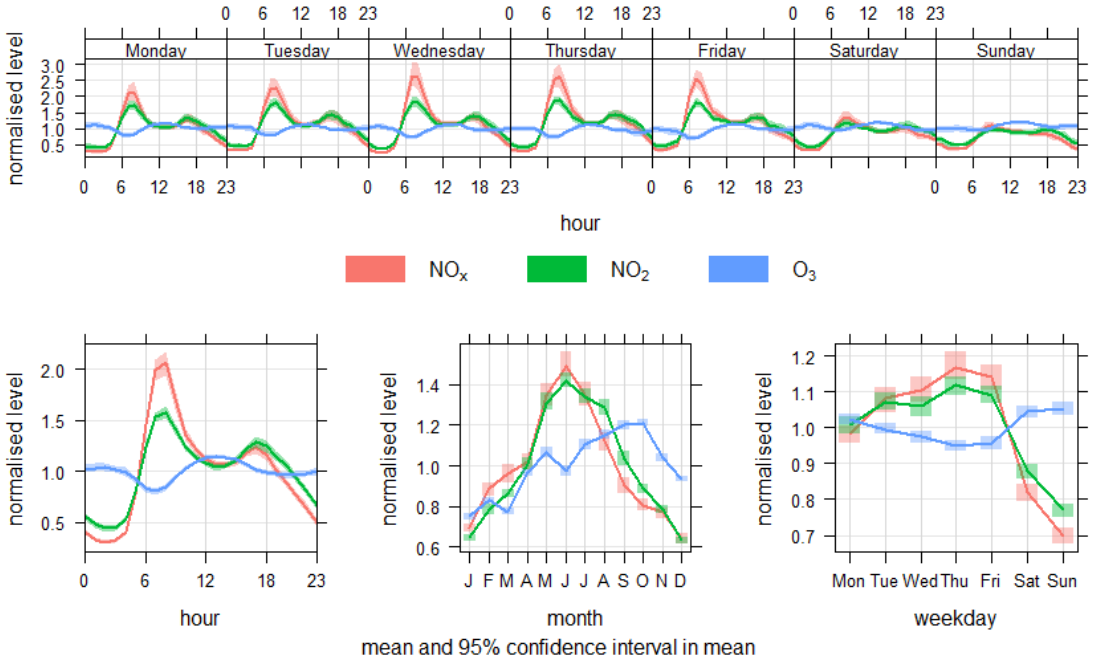

Figure 2.1: Wellington Central air monitoring station temporal variation of NOx, nitrogen dioxide (NO_2) and ozone (O_3) from 2017 to 2019

Petrol vehicles are the main source of carbon monoxide which is directly emitted in tail pipe exhaust. Although carbon monoxide is a harmful and regulated pollutant, roadside levels have lowered dramatically over the past decade in New Zealand due to improvement in engine technology resulting in lower emissions (Bluett et al., 2016). Carbon monoxide is still monitored at roadside because it is useful for tracking trends in traffic as it is correlated with traffic intensity and is a good indicator of emission trends in the petrol fleet.

Carbon monoxide is also emitted by wood burners, but can generally be distinguished from traffic sources as the peak emissions occur much later in the evening than the commuter peak travel time.

2.1.2 Black carbon and particulates from traffic

Fine particles known as $PM_{2.5}$ (aerodynamic diameter < 2.5 µm) are emitted by vehicle exhaust (primarily diesel) due to incomplete combustion. Most of these exhaust particles are ultra-fine in size (ie, less than 0.1 µm) and therefore do not make up a large fraction of measured $PM_{2.5}$ mass concentrations. $PM_{2.5}$ at roadside sites also contains particles not directly produced by tail pipe exhausts, such as, brake, tyre and road wear which create re-suspended road dust due to the turbulence of moving vehicles and from wind. These road dust particles make up a larger fraction of PM_{10}.

At the Wellington Central air monitoring site a recent short term pilot study in January and February 2020, in which GNS Science analysed elemental composition of air particulate filters, found that $PM_{2.5}$ was on average made up of 26% black carbon, 26% secondary sulphate, 28% sea salt, 19% soil and 1% smoke. The relatively high contribution of non-traffic sources to $PM_{2.5}$ at this location reduce its usefulness as a specific indicator for traffic-related particulates.

Black carbon or soot is a light-absorbing, carbon-containing constituent of $PM_{2.5}$ formed by the incomplete combustion of fossil fuels, biomass and biofuels (European Environment Agency, 2013). Main sources of black carbon include vehicles (particularly diesel powered), shipping, home-heating (wood or coal burning) and open burning (eg, wildfires or burning of agricultural waste). Diesel-powered vehicles, especially heavy commercial vehicles, are very high emitters of black carbon compared to petrol vehicles (Davy et al. 2011).

Black carbon and correlated co-emissions appear causally related with cardiovascular and lung cancer deaths (Grahame et al. 2014). Studies of short-term health effects show that the associations with BC are more robust than those with $PM_{2.5}$ suggesting that BC is a better indicator of harmful particulate from combustion sources (particularly traffic) than un-differentiated PM mass (Janssen et al. 2011). Although there are no health-based guidelines for black carbon, black carbon is thought to be a universal carrier for other harmful components of traffic-related air pollution (World Health Organization, 2012).

There are also climate concerns regarding black carbon emissions. Black carbon, because of its dark colour is very good at absorbing heat and in the atmosphere it acts as short-term climate warming agent (Ministry for the Environment, 2018). Therefore, reductions in black carbon can more quickly mitigate changes in the climate compared to carbon dioxide which persists for hundreds to thousands of years in the atmosphere (Ramanathan & Carmichael, 2008).

2.1.3 Traffic activity, emissions and air quality

Linking air quality trends at a monitoring site to locally-generated traffic emissions at any specific monitoring site is complex and challenging. Bluett et al. (2016) describe the main long term drivers of air pollutant trends as:

(a) Number of vehicle trips per day past the site

(b) Fleet profile (vehicle age distribution, vehicle type (eg, light duty, heavy duty), fuel type (eg, petrol, diesel)

(c) Driving conditions (congested or free-flowing)

(d) Real-world emissions performance of vehicles (effectiveness of emission control technology and fuel quality).

Other urban design features, such as building heights, road width proximity to intersections, traffic lights and number of bus stops along a street also strongly influence spatial variation in nitrogen dioxide in central Auckland (Miskell et al. 2015).

As the relationship between traffic and measured air pollution levels is strongly influenced by weather conditions, it can be difficult to detect changes in air pollution relating to changes in traffic intensity as opposed to meteorologically-driven changes (Anh et al., 1995).

Wind speed and direction, temperature, humidity and atmospheric stability are important factors affecting near-road air quality data. Wind speed and wind direction are particularly important as elevated concentrations can occur under low wind speeds and when the monitoring station is downwind of the roadside emission source. Temperature and relative humidity also affect the generation, transformation and transport of vehicle emissions (Baldauf et al., 2009). Meteorological conditions also affect atmospheric chemical reactions, for example, the formation of nitrogen dioxide from precursor emissions (Kroll et al., 2020).

2.2 Particulate matter from home heating, natural and other sources

Particulate matter concentrations, measured as PM_{10} and $PM_{2.5}$, are pollutants with significant adverse health impacts from long-term and short-term exposure (Ministry for the Environment, 2018). In terms of health effects, PM is important because there is no evidence of a safe exposure level or a threshold below which no adverse health effects occur (World Health Organization, 2013). Therefore, reducing particulate matter, even when air quality guidelines are met, results in health benefits for a population. In the Greater Wellington region, Masterton fails to meet the short-term standard (NES-AQ[2]) for PM_{10} and the World Health Organization guidelines for $PM_{2.5}$ (World Health Organization, 2006) for due to high level of emissions from solid fuels used for home heating.

[2] Resource Management (National Environmental Standards for Air Quality) Regulations 2004

Source apportionment studies of air particulate matter samples at various sites in the region identified domestic fires, motor vehicle emissions and secondary sulphate as the main contributors to measured levels of fine particulate matter (Davy, 2007). The main contributors to the coarse fraction ($PM_{2.5-10}$) were found to be re-suspended road dust containing brake, tyre and road surface wear, crustal matter associated with wind-blown dust from construction, unpaved areas and soils; and marine aerosol from long range transport from surrounding oceans (Davy, 2007). The relative contribution of each source to observed particulate concentrations depends on the monitoring site location and seasonality in emission sources (eg, winter wood burning) and the influence of meteorology (eg, marine aerosol concentrations increase with wind speed) (Davy & Trompetter, 2018). Figure 2.2 and Figure 2.3 shows the differences in annual particulate matter composition found at Upper Hutt and Masterton West air monitoring sites.

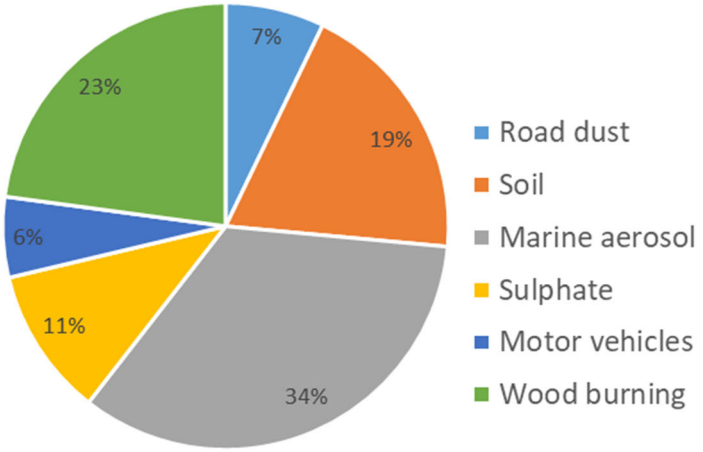

Figure 2.2: Annual source contribution to PM_{10} measured at Upper Hutt air monitoring station (2000-2002) (Davy, 2007)

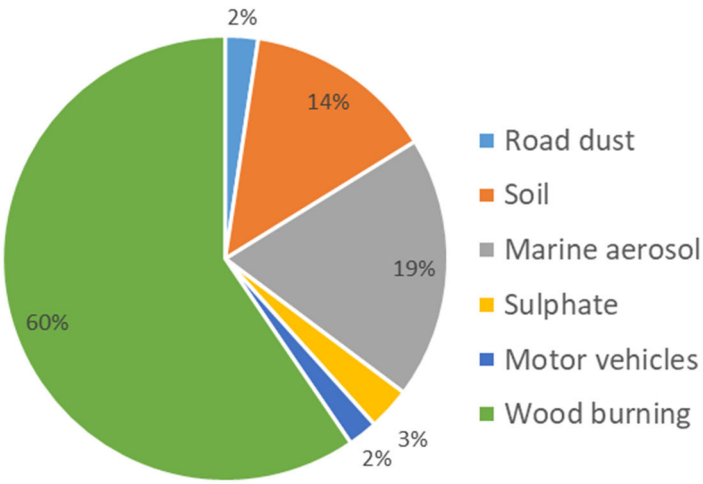

Figure 2.3: Annual source contribution to PM_{10} measured at Masterton West air monitoring station (2002-2004) (Davy, 2007)

3. Methods

3.1 Air monitoring site locations

Air quality and meteorological measurements were obtained from Wellington Central (urban roadside), Upper Hutt (urban background residential) and Masterton West (urban background residential) air monitoring sites shown in Figure 3.1.

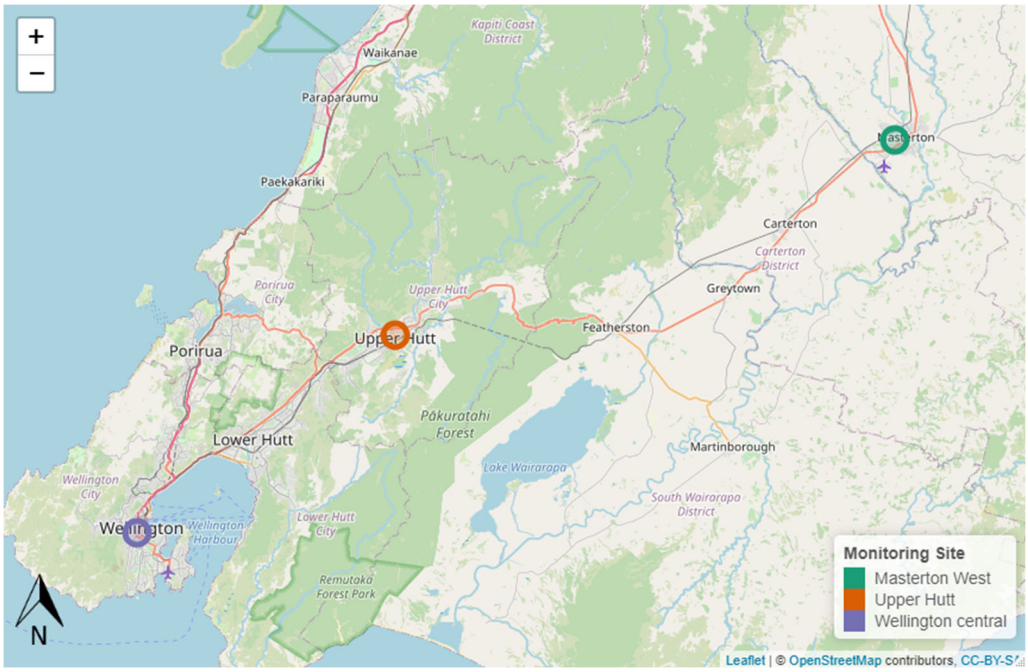

Figure 3.1: Air monitoring site locations: Wellington Central, Upper Hutt and Masterton West

Wellington central monitoring site is located alongside the Wellington Urban Motorway (SH1) close to the intersection of Willis Street and Vivian Street (Appendix A.1). Figure 3.2 shows the location of the station with respect to local roads and available traffic counts for 2019. Approximately 100 public bus trips per weekday pass the Wellington Central monitoring site along Willis Street northbound (https://www.metlink.org.nz/stop/7711). The buses along this route are predominately EURO VI models.

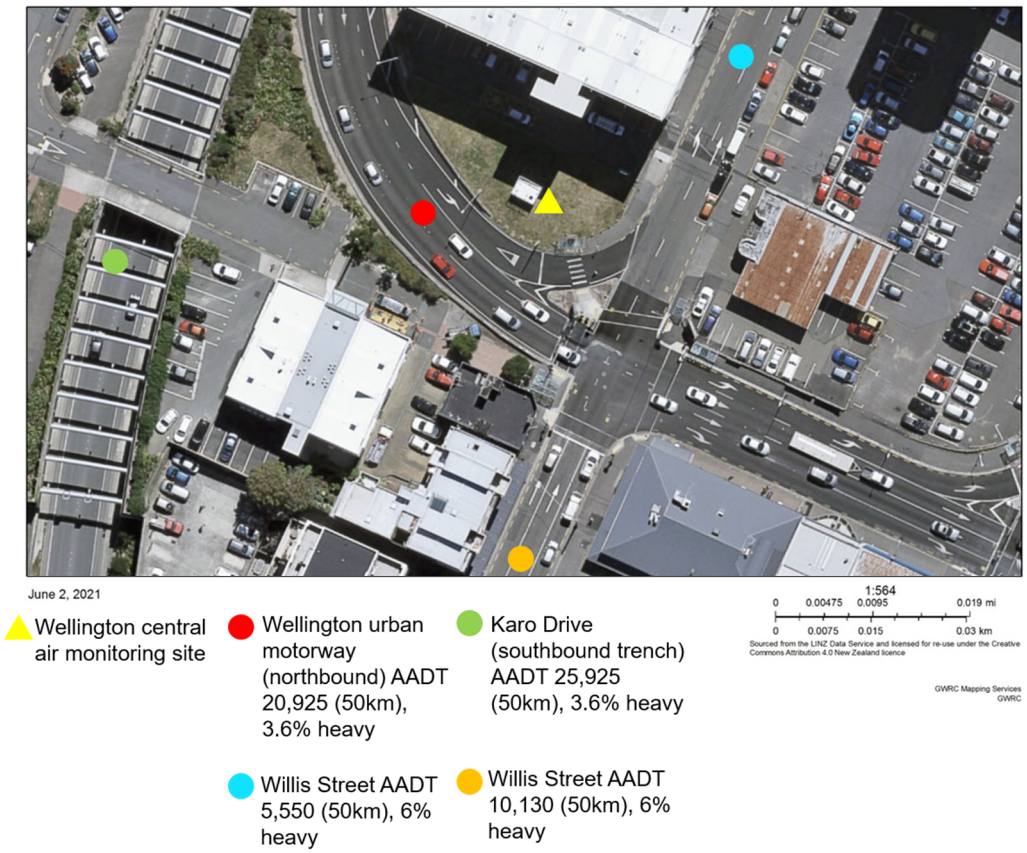

Figure 3.2: Location of Wellington central and annual average daily traffic counts (AADT) for 2019. Wellington urban motorway (Terrace Tunnel southbound NZTA3 Ref 01N11074) and Karo Drive (Terrace Tunnel northbound NZTA Ref 01N21074) and Willis Street (mobileroad.org).

The Upper Hutt monitoring site is situated in a suburban park away from the direct influence of roadside emissions (Appendix A.2) and therefore this site is considered to represent the urban background for traffic-related air pollutants. During winter, urban Upper Hutt is impacted by low level wood burning emissions, and therefore meets the PM_{10} air quality standard. The Masterton West monitoring site is located on the grounds of a local high school and is strongly influenced by home heating emissions during the winter months and therefore fails to meet the PM_{10} air quality standard (Appendix A.3).

3.2 Data sources

3.2.1 Air pollutant measurements

All air pollutant measurements available from each air monitoring site from 1/1/2016 to 30/9/2020 were validated and reported as 1-hour averages (Appendix A4). Wellington central had 10 days missing record for wind speed and NO_2 and NOx during Alert Level 4 due to malfunctioning equipment.

[3] https://www.nzta.govt.nz/resources/state-highway-traffic-volumes/

The black carbon measurements were split into fossil fuel derived and biomass-burning derived concentrations using the AE33 default source apportionment model based on Sandradewi et al. (2008). The identification of the wood burning component of black carbon is based on the principle that organic compounds in aerosols from wood burning absorb light more efficiently in shorter wavelengths compared to aerosols from fossil fuel burning. Although the BC source apportionment results have not been verified for this monitoring location, for instance, by comparing with co-measured chemical markers of wood smoke, plots of winter and summer concentrations are temporally consistent with a BC biomass burning component from winter evening wood smoke (Appendix A5).

Monthly averages for nitrogen dioxide were obtained from Waka Kotahi NZTA and GWRC's passive NO_2 diffusion tube monitoring (NZ Transport Agency, 2020[4]). Passive tube measurements are less accurate than the continuous method and typically overestimate concentrations by 33% compared to reference monitoring (Kuschel & Sridhar, 2020). Despite these limitations they usefully extend spatial coverage of air pollution monitoring data and are a relatively inexpensive and reliable method for estimating long term trends (Air Quality Expert Group, 2004).

3.2.2 Meteorological measurements

Co-located meteorological measurements were obtained from the three monitoring sites. Wind flows across Wellington are almost always from either N-NW or S-SE but how these winds are experienced locally varies due to funnelling through complex topography and between tall buildings. Although the Wellington city area is close to the coast with The Wellington central monitoring site is sheltered by a nearby multi-storey building which blocks wind from the NW-N-NE sectors. Therefore, wind speed and direction measured at this site are highly localised and will not apply to other parts of Wellington city. Due to local sheltering effects and low mast height of 6m, mean wind speeds measured at Wellington central are much lower than recorded at Metservice weather station at Kelburn which has a 10m mast. A mast height of 10m is required to meet World Meteorological Organization standards. Wind roses for 2017 to 2019 are shown for Wellington central, Upper Hutt, Masterton West and for the Kelburn weather stations (Appendix A6).

3.2.3 Traffic counts

Hourly traffic counts were downloaded from NZTA's State Highway Traffic Monitoring System (TMS). These counts were aggregated to average daily totals (midnight to midnight) for use in the analysis. Annual average daily traffic (AADT) estimates for local roads were obtained from Waka Kotahi NZTA[5]. Locations of Waka Kotahi traffic monitoring sites are shown in relation to Wellington Central air monitoring site in Figure 3.3.

[4] https://www.nzta.govt.nz/resources/air-quality-monitoring/
[5] https://www.nzta.govt.nz/resources/state-highway-traffic-volumes/

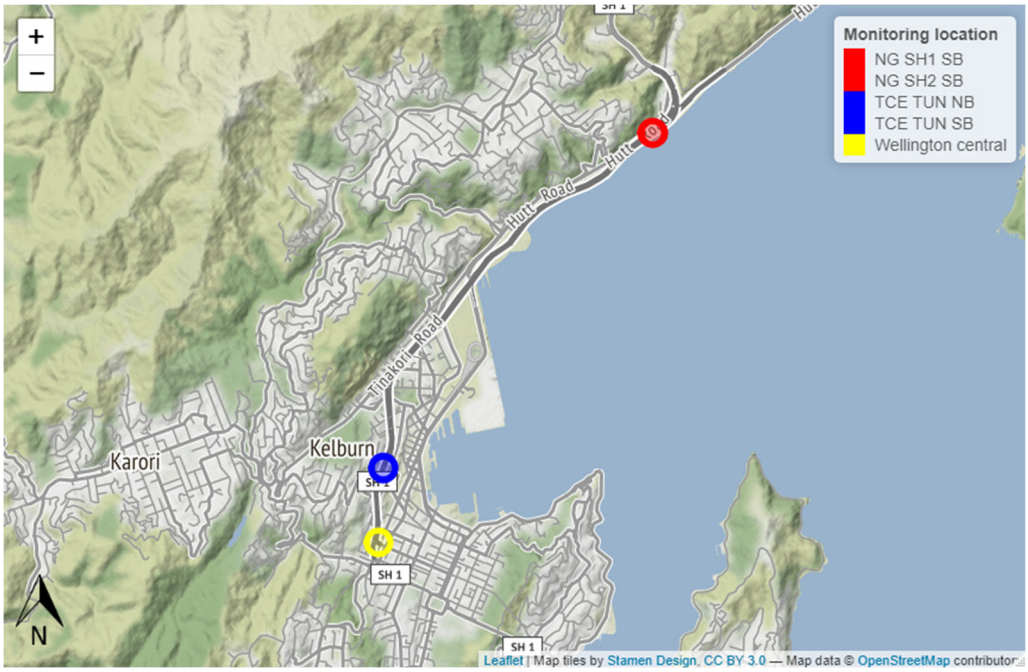

Figure 3.3: Location of NZTA regional continuous traffic counting sites on state highway at Ngauranga (red) and north of the Terrace Tunnel (blue). The location of Wellington central air monitoring station is shown in yellow.

3.3 Modelling to predict BAU air pollutant concentrations

A critical issue for evaluating changes in air quality is determining whether the change is due to variations in meteorology or emissions intensity (Grange & Carslaw, 2019). A simple example, is the impact of winter meteorological conditions on traffic-related pollutant concentrations which are higher in winter than in summer. Building predictive models to explain air quality is difficult due to the complex interactions between emissions, meteorology and atmospheric chemistry. Machine learning techniques, such as random forest (RF) regression, have been successfully developed and used to detect effects of interventions on air quality trends by controlling for the effect of meteorology through 'meteorological normalisation' (Grange & Carslaw, 2019) and modelling short-term relationships between traffic and air pollution (Kamińska, 2018).

More recently, RF modelling has been used to quantify the effects of the COVID-19 lockdown on air quality, for example, in NZ (Talbot et al. 2021), the United Kingdom (Jephcote et al., 2020), Austria (Lovrić et al., 2020) and in Switzerland (EMPA, 2020). This approach trains a random forest model to explain pollutant concentrations based on meteorological and time variables for a training period under past BAU conditions (eg, 2016 to 2019). The model built on the past training period is then used to predict pollutant concentrations using observed meteorological and time variables for the periods of COVID-19 restrictions in 2020. This approach accounts for the effect of any unusual or atypical meteorological conditions during 2020 which can obscure underlying air quality trends.

RF models were trained using 1-hour air pollution observations (Appendix A4) and meteorological and time variables (Table 3.1) up to 29 February 2020 to capture pre COVID-19 BAU emissions patterns. Traffic emissions (eg, kg/hr of NOx emitted) were not available as a model inputs, instead hour of the day was used to capture the diurnal variation in traffic emissions. Day of the year (Julian Day) was used to represent seasonal patterns in emissions, for example, solid fuel emissions which are present during winter.

The RF modelling was carried out using the *rmweather* R package (Grange et al. 2018; Grange & Carslaw, 2019) which uses the underlying *ranger* R package (Wright & Ziegler, 2017). The default *rmweather* hyperparameters were used; 0.8 training fraction, 300 trees grown for the forest, minimum node size of 5 and 300 random samples (with replacement) for the predictions (Grange & Carslaw, 2019). The RF BAU models (2016 to 2019) were trained on 80% of the data (randomly selected) so that 20% of the data is withheld as a testing data set. In this way the RF model trained on 80% of the input variables can then be used to predict pollutant values from the 20% of data that was not used in the model training. This feature allowed examination of the model residuals (the observed pollutant values minus the predicted values) in the test data set to determine whether there were any biases in prediction under the BAU scenario that might affect the validity of any predictions made for 2020 COVID-19 alert levels.

Table 3.1: Meteorological and temporal predictor variables model inputs

Variable type	Predictors	Description
Meteorological	Temperature	Degrees Celsius
	Wind direction	Degrees (true north)
	Wind speed	m/s
	Relative humidity	%
	Barometric pressure	hPa
Temporal	Unix date	Number of seconds since 1/1/1970. Represents the long-term time trend in air pollutants
	Julian day	Day of the year, representing seasonal variation
	Weekday	Day of the week, represents variation by day of the week
	Hour	Hour of day represents diurnal variation as a proxy for traffic emissions

Using a similar approach to Jephcote (et al. 2020) and EMPA (2020), the BAU RF models were used to predict or forecast hourly pollutant concentrations from 1 March 2020 to 30 September 2020 using site-specific meteorological observations and time variables. Wind speed is a critical data input for the

predictions, so where wind speed is not available, predictions are not made. This produced a 1 March 2020 to 30 September 2020 time series of 1-hour average observed pollutant concentrations and BAU-predicted pollutant concentrations.

The numerical difference between observed pollutant concentrations and BAU-predicted pollutant concentrations was used to estimate the effect of the COVID-19 movement restrictions on air quality. For each Alert Level period, the difference between predicted 1-hour averages and observed pollutant 1-hour averages was calculated and a paired two-sided Student's t-test (R Core Team, 2020) used to determine whether the average difference between each 1-hour value (observed and predicted) was statistically different from zero at the 95% Confidence Limit.

The 1-hour average observed pollutant concentrations and BAU-predicted pollutant concentrations were also averaged for each Alert Level period (Table 3.2). For each Alert Level period the percentage change (BAU-predicted concentration minus observed concentration divided by BAU-predicted concentration) was used to estimate the reduction or increase in air pollution attributable to changes in emissions occurring under COVID-19. Where this difference was statistically significant (based on the paired Student's t-test for the difference between each 1-hour observed and predicted value) an assessment was made whether this difference was due to changes in emissions intensity. These percentage changes were compared to percentage change in average daily traffic counts (where available) between 2020 and 2019 for each Alert Level period.

Table 3.2: Dates Alert Levels in force

Alert Level	Start date	End date
L2	21/3/2020	23/3/2020 13:29
L3	23/3/2020 13:30	25/3/2020 23:59
L4	26/3/2020 00:00 [25/3 11:59]	27/4/2020 23:59
L3	28/4/2020 00:00	13/5/2020 23:59
L2	14/5/2020 00:00	8/6/2020 23:59
L1	9/6/2020 00:00	11/8/2020
L3/L2	12/8/2020 [12:00]	22/9/2020

4. Results

4.1 Evaluation of BAU pollutant modelling

Models trained on air pollutant measurements, meteorological time variables (1 January 2016 to 30 February 2020) to represent the pre COVID-19 BAU situation performed well for nitrogen dioxide (NO_2), oxides of nitrogen (NOx), carbon monoxide (CO), ozone (O_3) and black carbon (BC) with R^2 values ranging from 0.70 to 0.88 (Appendix A7).

The accuracy of the BAU model predictions were assessed by looking for any patterns in the 1-hour residuals from 2016 to 2019. All pollutants showed a tendency for under prediction as concentrations increased. There were also temporal variation in the residuals, with under prediction typically on Tuesday, Wednesday, Thursday and Friday representing a different emissions pattern than on the weekend and Mondays which are more affected than other days of the week by public holidays.

Models for $PM_{2.5}$ and PM_{10} had reduced explanatory power, with R^2 ranging from 0.45 to 0.66, reflecting that different source components of PM respond differently to the same meteorology, for example, the sea salt component is elevated during high wind conditions, but combustion-derived PM sources are typically much lower under the same conditions. Therefore, RF model BAU-predictions were not made for PM_{10} and $PM_{2.5}$ and instead average PM concentrations observed during the different Alert Levels were compared to average concentrations observed on the same Julian days from 2016 to 2019.

4.1.1 Wellington central

At this roadside location, hour of the day was the most important explanatory variable for black carbon (fossil fuel), NOx and CO concentration as it was strongly correlated with diurnal traffic flow patterns and therefore traffic emissions. Wind speed was also an important explanatory variable due to the moderate negative correlation between wind speed and pollutant concentrations, ie, lower wind speeds are associated with higher pollutant concentrations due to less effective dispersion conditions. Day of the week, although less important than hour and wind speed, was a predictor for black carbon (fossil fuel) and NOx due to weekday/weekend difference in air pollutant concentrations, most likely due to fewer diesel-powered commercial or heavy vehicles operating on weekends compared to weekdays. BC biomass burning was also dependent on temperature and Julian day reflecting the contribution of winter wood burning. NO_2 showed some dependence on Julian day due to seasonal factors, such as background ozone availability, which may contribute to higher NO_2 levels during winter.

Carbon monoxide was dependent on Unix date due to the strong linear downward trend in this pollutant over time from improvements in the vehicle fleet emissions control technology (Bromley & Gray, 2017). Ozone was dependent on wind speed, but unlike the other pollutants, ozone was positively correlated with wind speed, perhaps reflecting that under higher wind speeds there is enhanced dilution of NOx and therefore less quenching of O_3. Ozone may also be positively correlated with wind speed as ozone levels peak during spring (Adeeb & Shooter, 2004) coinciding with spring (September to October) generally being the windiest season (Chappell, 2014).

4.1.2 Upper Hutt and Masterton West

Air pollutant concentrations measured at the Upper Hutt site showed the same dependence on wind speed and hour of day for NOx and NO_2, but with temperature being a more important predictor variable than at Wellington Central. The dependence of CO on temperature and Julian Day reflects the seasonal contribution of winter home heating emissions from wood burning at Upper Hutt and Masterton West.

4.2 Observed concentrations of pollutant gases and black carbon during COVID-19 Alert Levels compared to BAU-predictions

The RF models with $R^2 > 0.75$ were used to predict (forecast) black carbon, NOx, NO2, carbon monoxide and ozone hourly concentrations from 1 March 2020 to 30 September 2020. These BAU-predictions were made from 1-hour meteorological data available for each site and temporal variables (ie, hour, Julian day, day of week) except for Unix date. The BAU-predicted and observed pollutant concentrations were then averaged for each Alert Level period. The percentage change[6] between BAU-predictions and observations and whether the difference between BAU-predictions and observations is statistically significantly different from zero at the 95% confidence level (two-sided, paired Student's t-test) for each pollutant is presented in Appendix A8 and shown in Figures 4.1 and 4.2.

[6] (BAU-predicted –observed / BAU-predicted) * 100

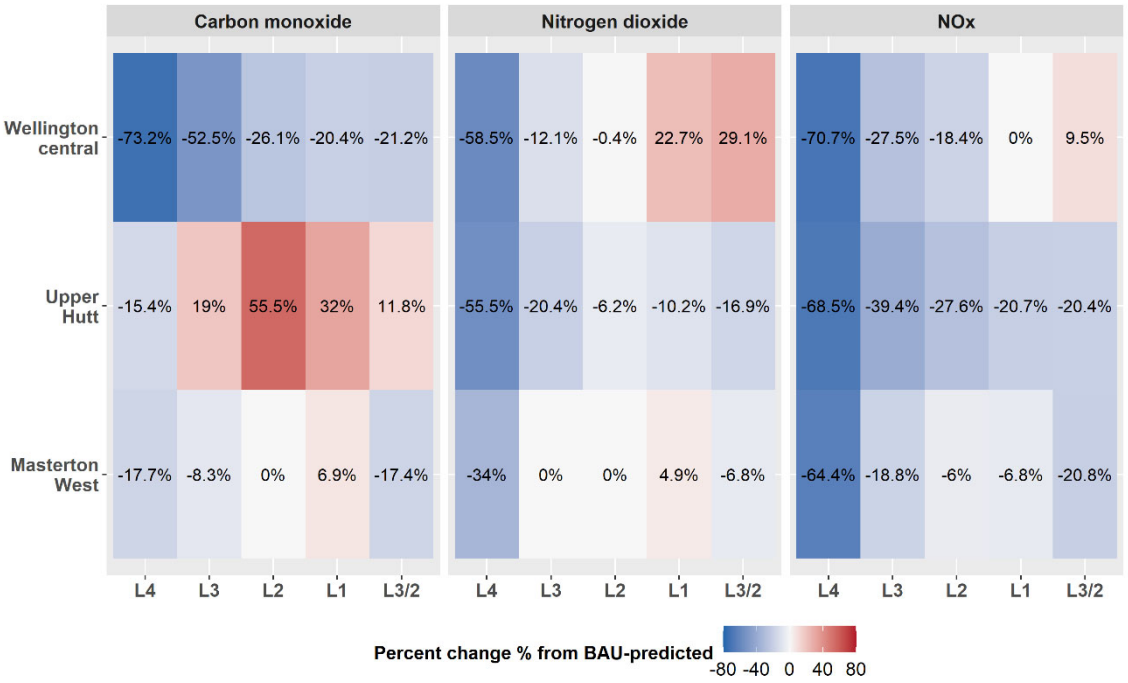

Figure 4.1: Percentage difference between observed carbon monoxide, nitrogen dioxide and NOx and BAU-predictions across all three sites by Alert Level. Where the percentage difference was not different from zero at the 95% confidence level, the percentage difference is shown as 0%.

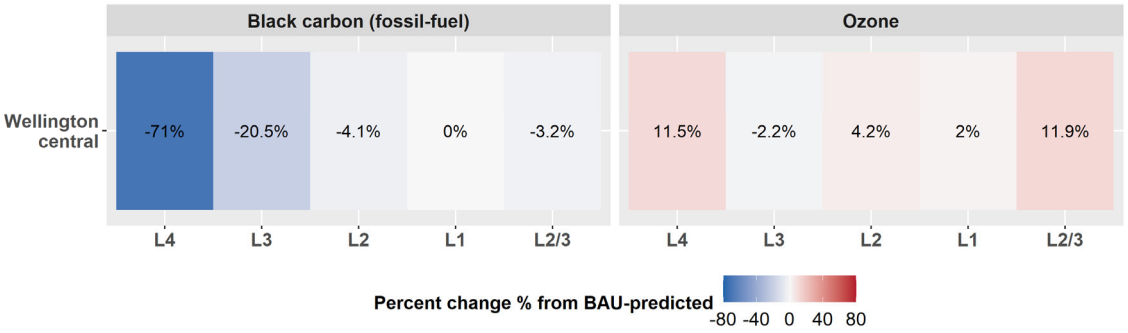

Figure 4.2: Percentage difference between observed black carbon (fossil-fuel) and ozone and BAU-predictions across at Wellington central by Alert Level. Where the percentage difference was not different from zero at the 95% confidence level, the percentage difference is shown as 0%.

Nitrogen oxides (NOx) were substantially reduced at all sites during L4 compared to BAU-predictions, reflecting the dramatic reduction in traffic volumes. NOx remained lower than predicted for the remainder of the restriction periods, apart from L1 and L3/2 at Wellington central, where NOx levels returned to normal or slightly higher than predicted. Nitrogen dioxide levels were mostly below predicted, but to a much lesser extent than NOx. The exception was during L1 and L3/2 at Wellington central, where NO2 levels were much higher than expected. Interestingly, ozone levels at Wellington central were higher than predicted during L4 and L3/2. Black carbon at Wellington

central showed a similar magnitude of reduction as seen in NOx during L4 and L3, but returned to near-normal levels from L2 onwards. There were differences in carbon monoxide between sites, with Wellington central showing a substantial reduction in L1 and remaining below predicted levels for the remainder of the Alert Levels. Conversely, Upper Hutt and Masterton West showed a much lower reduction during L1 and Upper Hutt recorded higher than expected carbon monoxide levels, particularly during L2.

A more granular picture of how observed pollutant levels differed from BAU-predicted the 1-hour averages were aggregated to 24-hour averages (midnight to midnight) and plotted for each pollutant and monitoring site (Figures 4.3 to 4.13). Deltas or differences between 24-hour average observed pollutant levels and BAU-predicted levels are shown. A positive difference indicates that observed air pollutant levels are higher than expected under BAU for the given weather conditions and a negative difference indicates air pollutants are lower than expected for BAU and the weather conditions. The plots are presented as 7-day moving averages to make the trends easier to visualise.

The moving 7-day averages for BAU-predictions and observations (Figures 4.3 to 4.13) for all pollutants at all sites clearly show the disruption to normal emission patterns during L4. From L2 onwards the BAU-predictions and observations are correlated showing temporal coincidence of peaks and troughs due to dependence on weather conditions which suggests a lower level of emissions but with pollutant levels responding to meteorology as in past years. For Wellington central, the 24-hour average difference between observed black carbon, NOx and NO_2 concentration and BAU-predicted values clearly show the outlier of the 1 June 2020 Queens Birthday Monday public holiday with a high negative value representing lower than expected air pollution, as fewer cars were on the road compared to a ‘normal’ Monday (Figures 4.3 to 4.5). The Queen’s Birthday effect is less noticeable at the urban background monitoring sites in Upper Hutt and Masterton West.

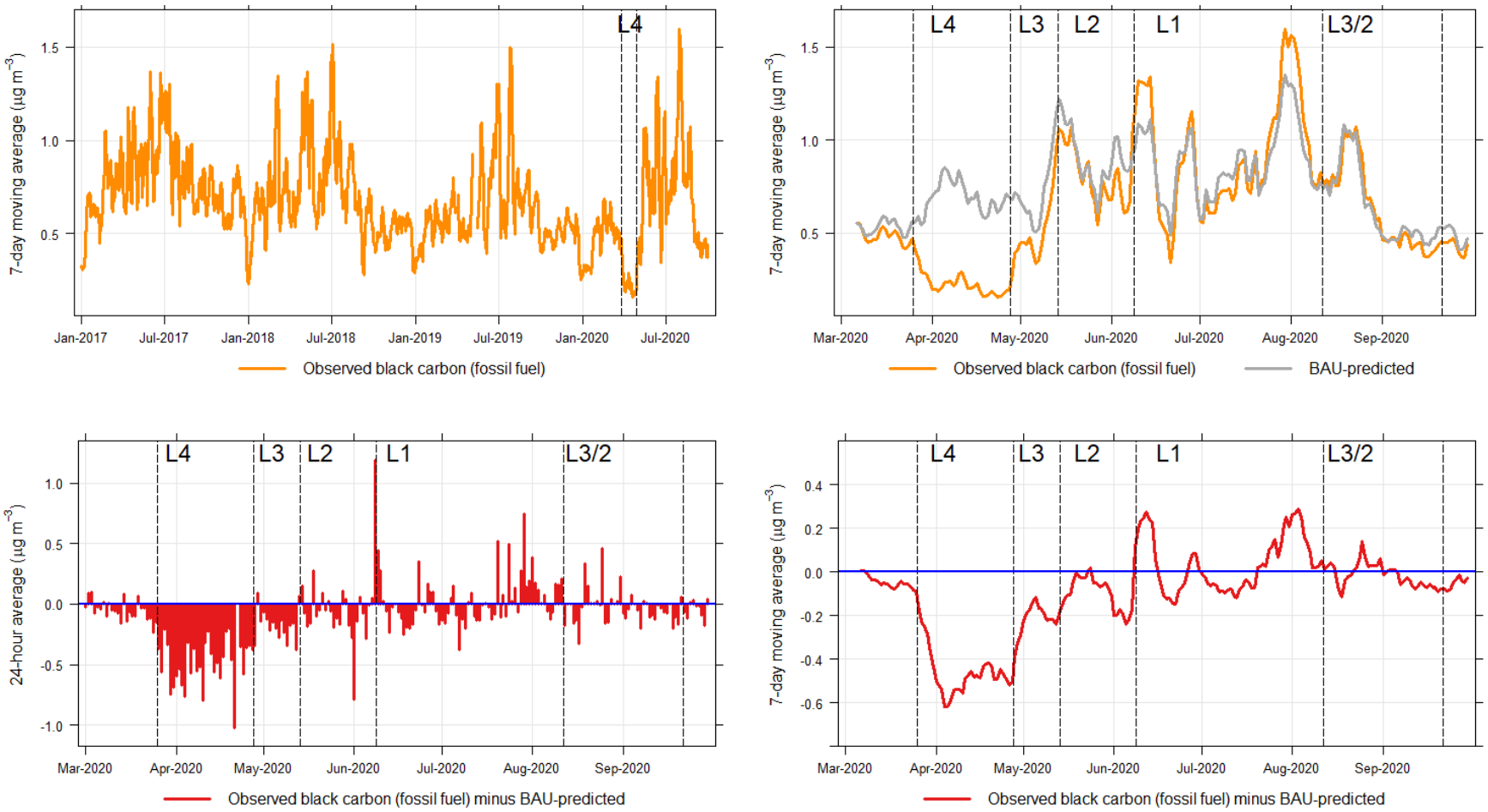

Figure 4.3: Wellington central back carbon (fossil fuel derived). Top left: 7-day moving average observations 1/1/2017 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

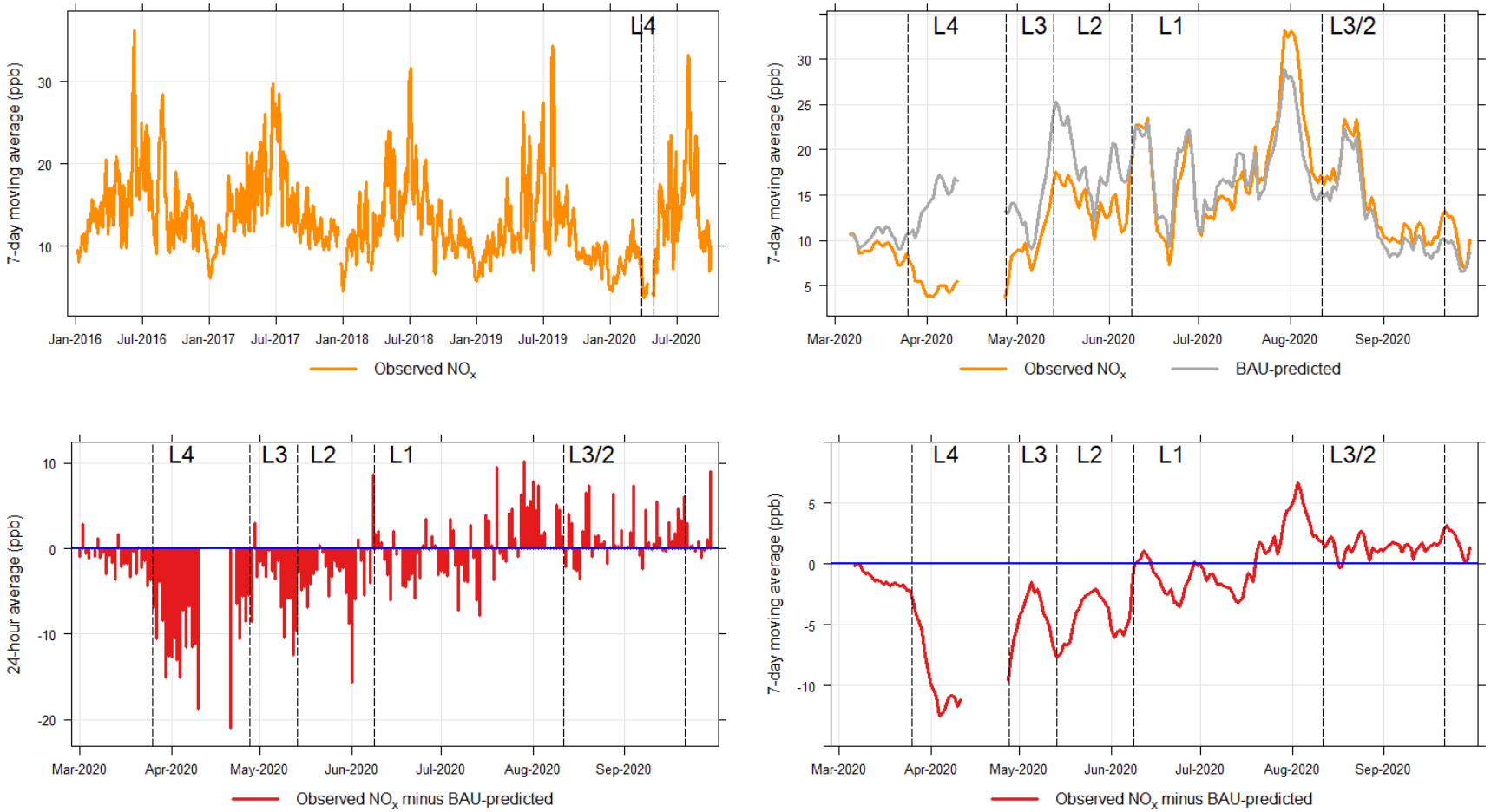

Figure 4.4: Wellington central nitrogen oxides (NOx). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

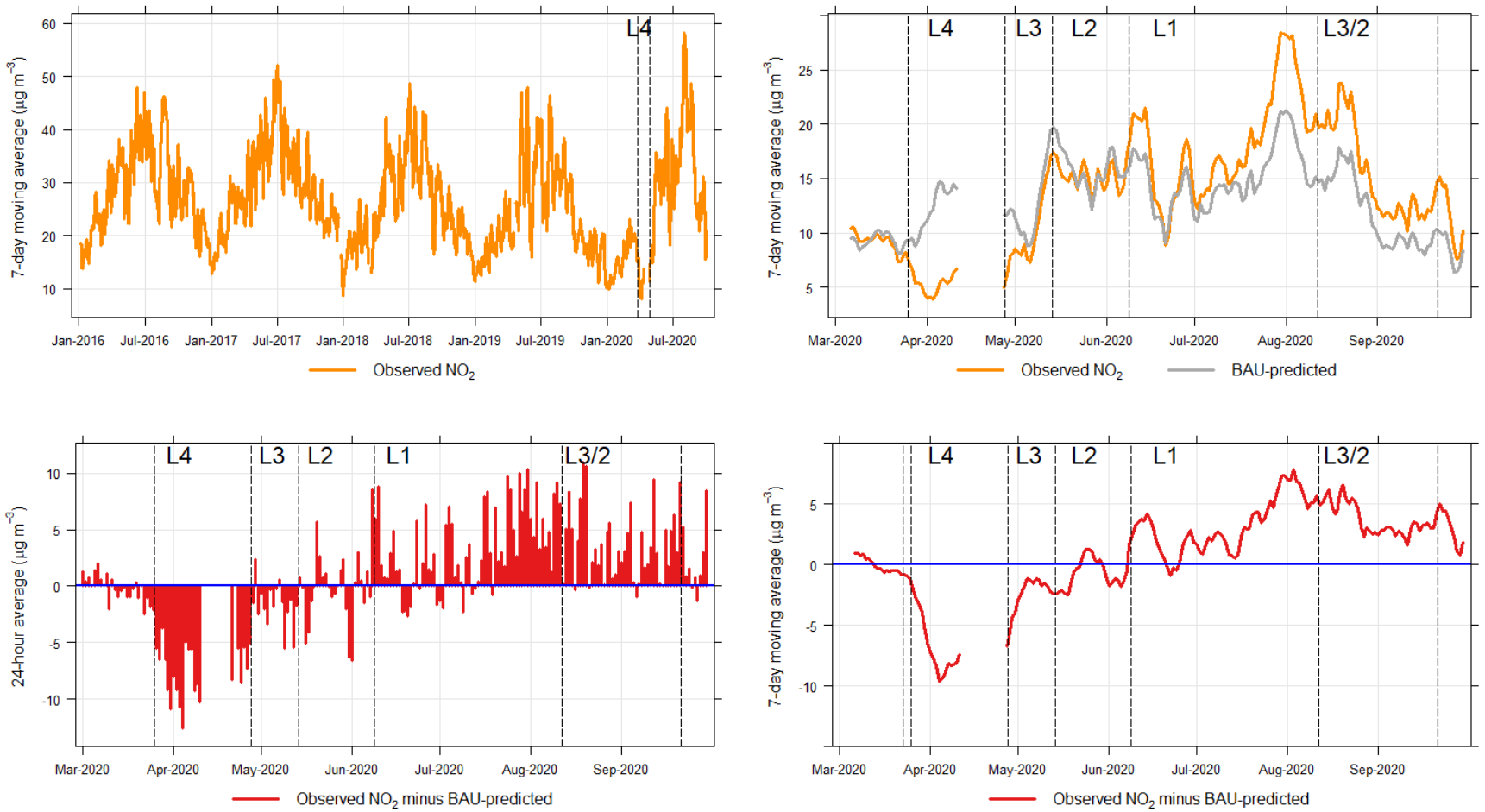

Figure 4.5: Wellington central nitrogen dioxide (NO_2). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

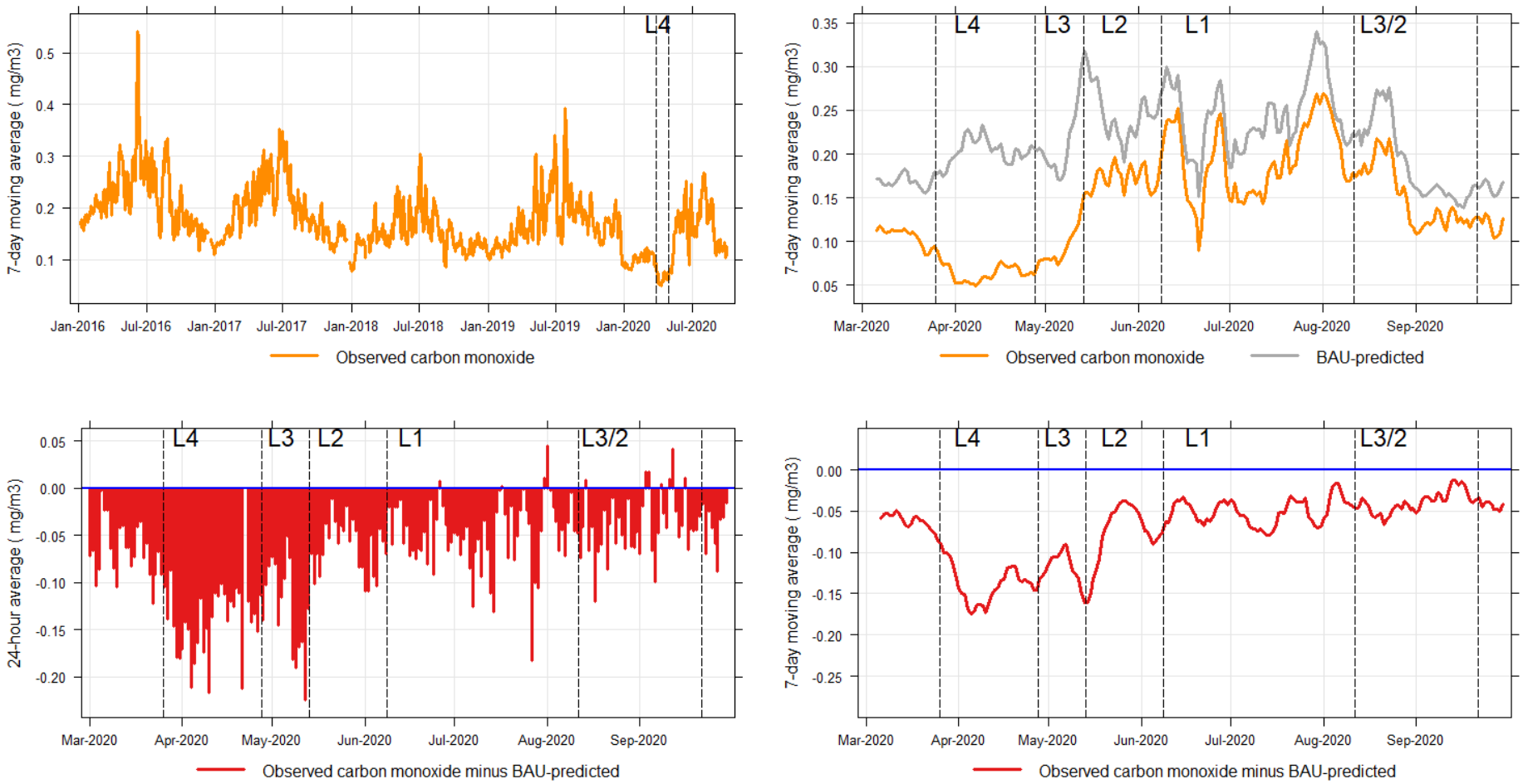

Figure 4.6: Wellington central carbon monoxide (CO). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

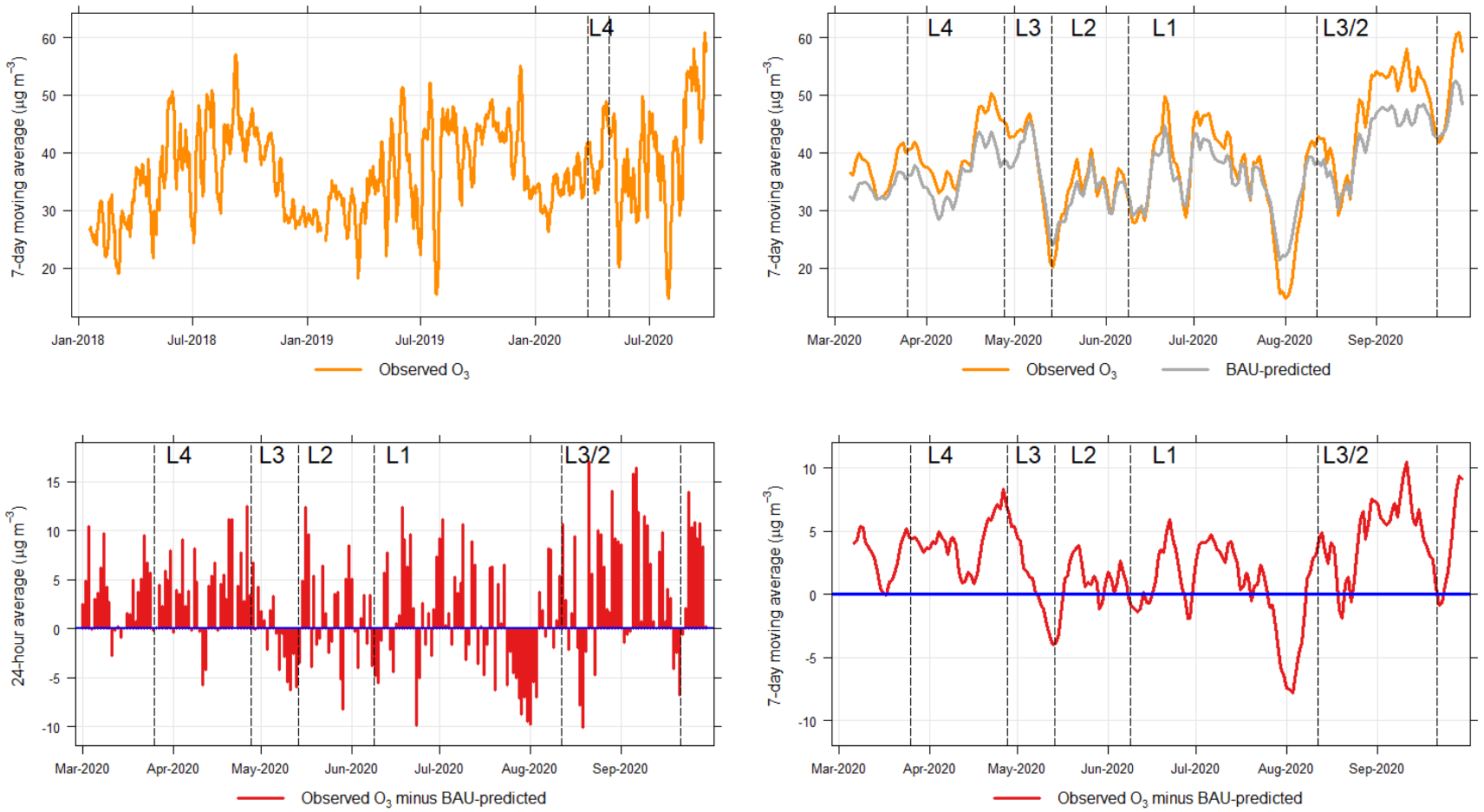

Figure 4.7: Wellington central ozone (O_3). Top left: 7-day moving average observations 1/1/2018 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

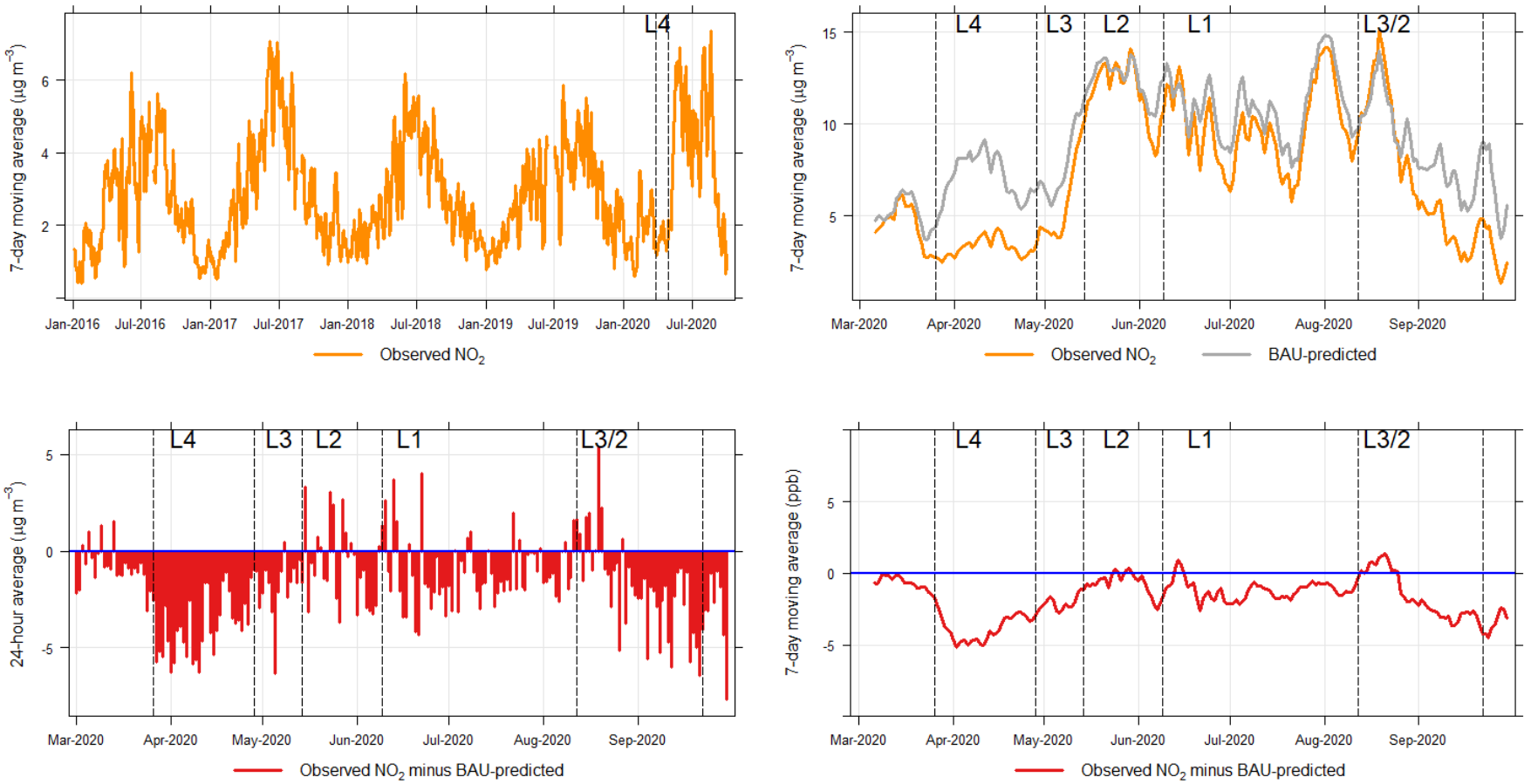

Figure 4.8: Upper Hutt nitrogen dioxide (NO2). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

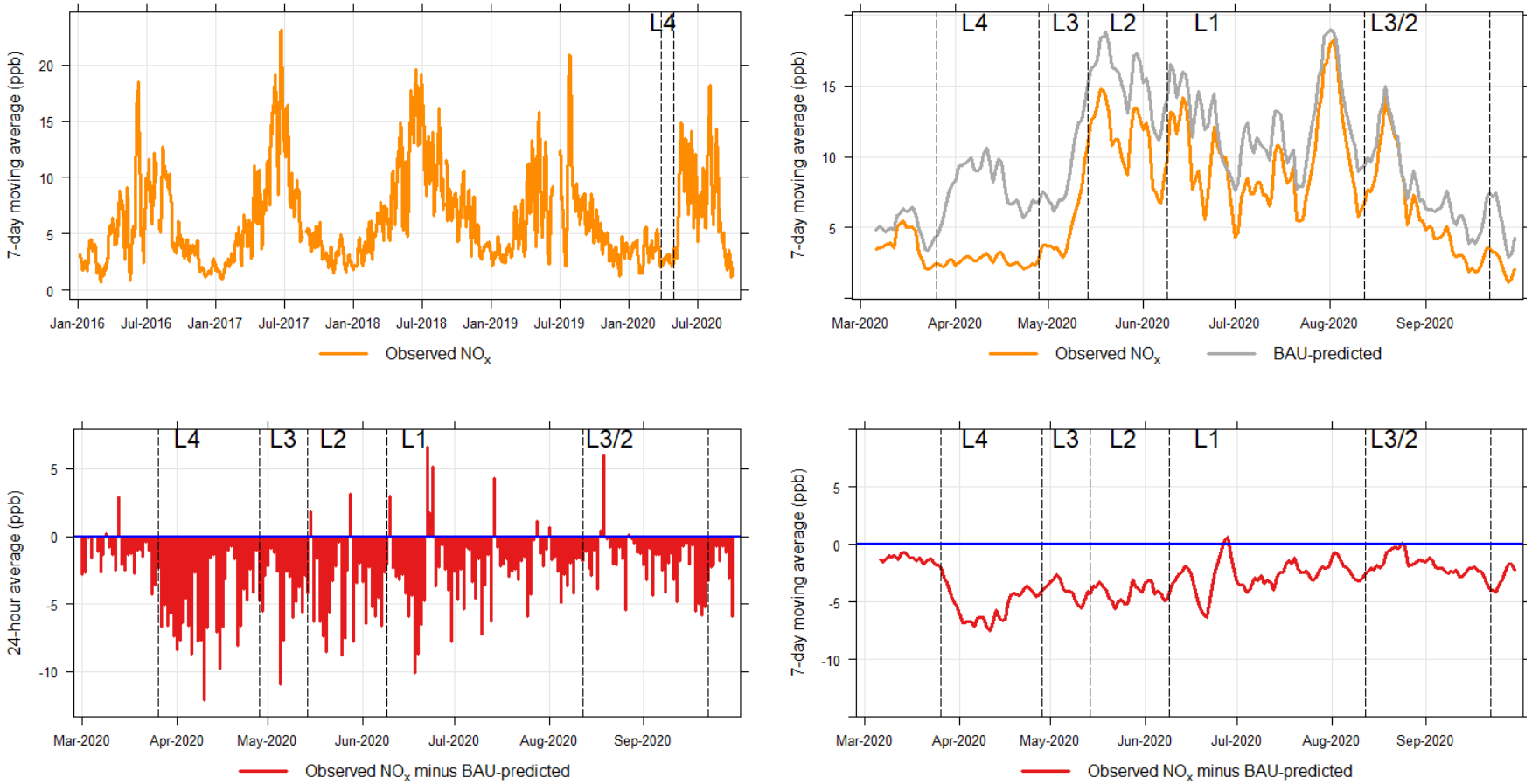

Figure 4.9: Upper Hutt nitrogen oxides (NOx). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

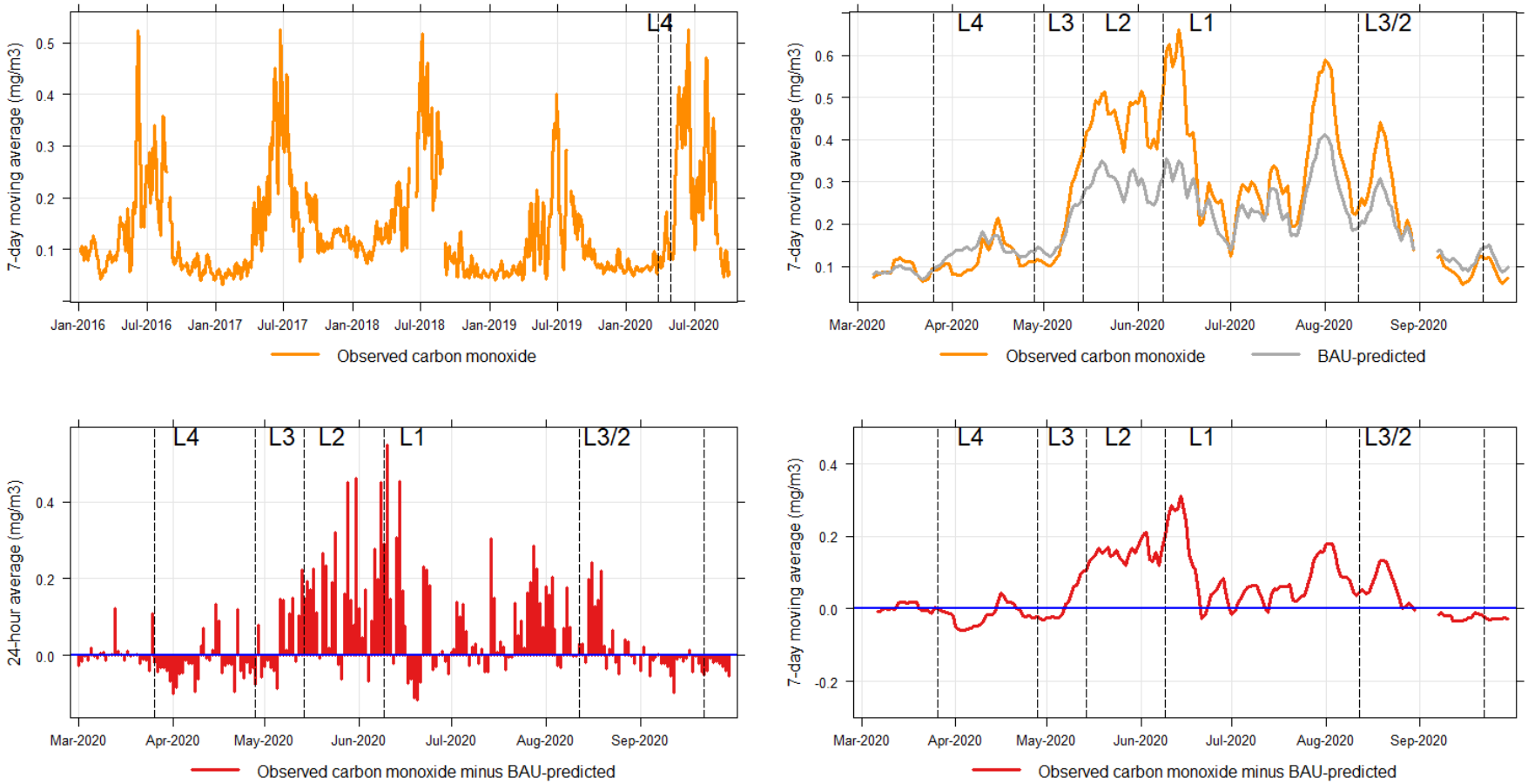

Figure 4.10: Upper Hutt carbon monoxide (CO). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

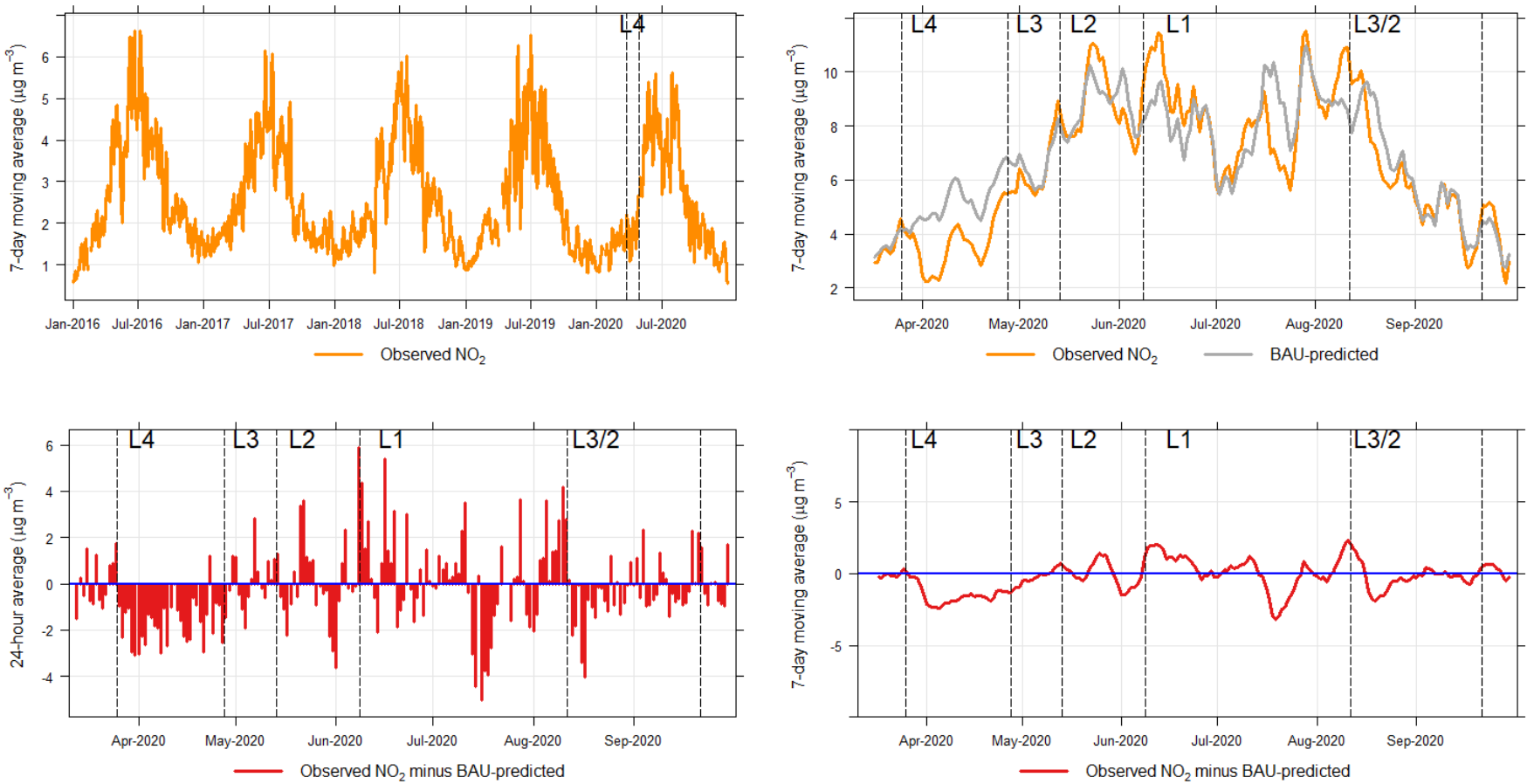

Figure 4.11: Masterton West nitrogen dioxide (NO_2). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 13/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 13/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 13/3/2020 to 30/9/2020.

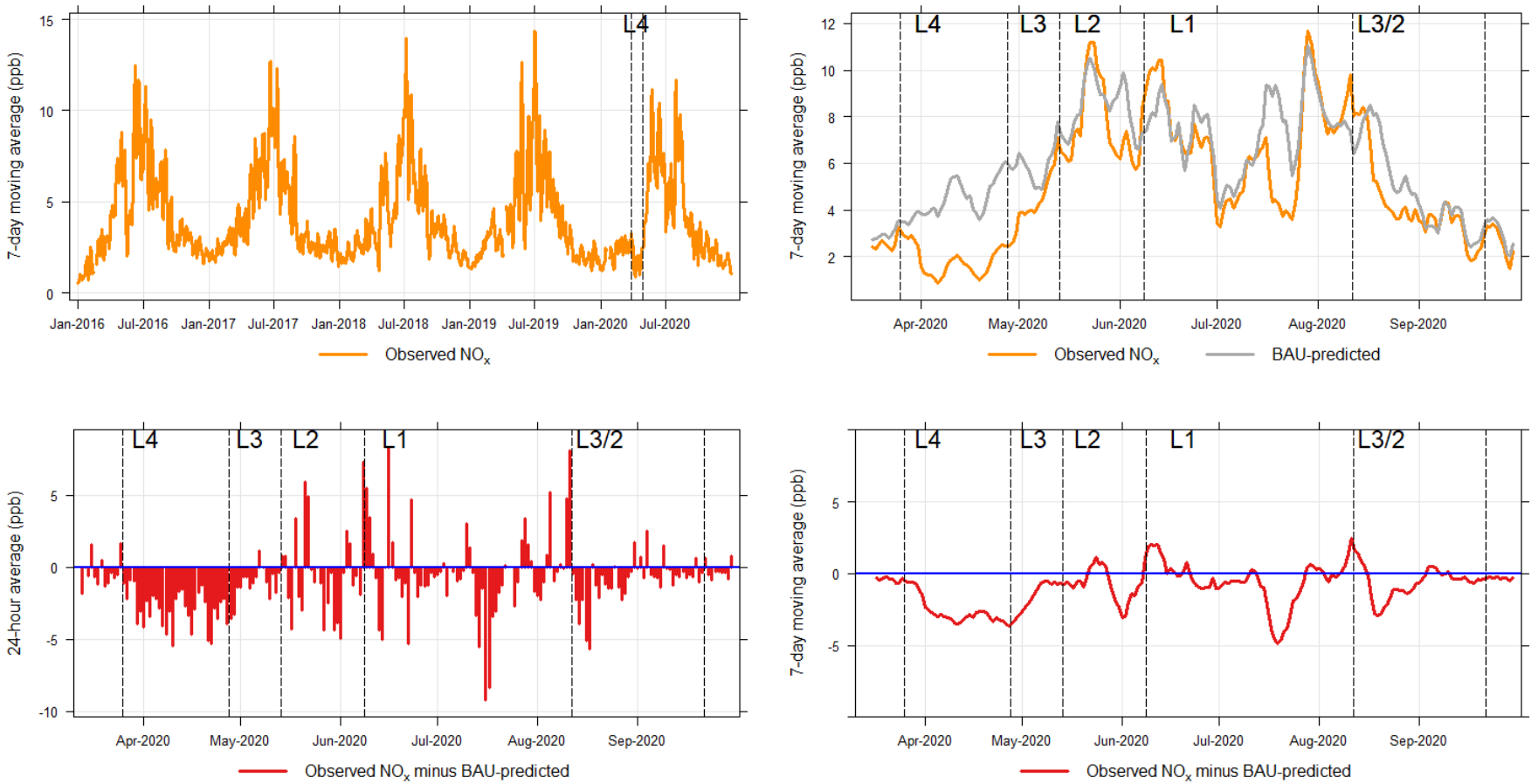

Figure 4.12: Masterton West nitrogen oxides (NOx). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 13/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 13/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 13/3/2020 to 30/9/2020.

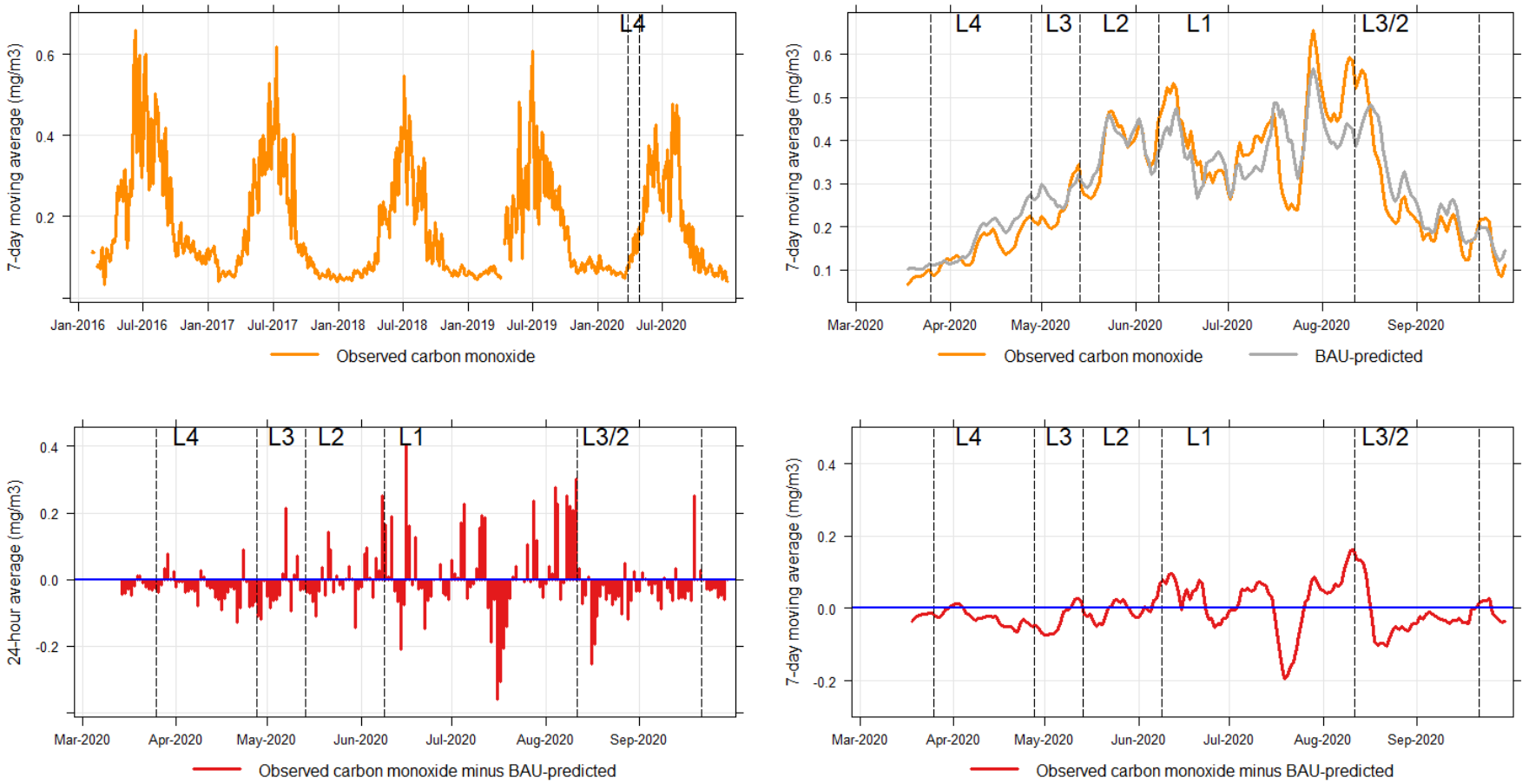

Figure 4.13: Masterton West carbon monoxide (CO). Top left: 7-day moving average observations 1/1/2016 to 30/9/2020. Top right: 7-day moving average observations (orange) and BAU-predicted (grey) 1/3/2020 to 30/9/2020. Bottom left: 24-hour average observations minus BAU-predicted 1/3/2020 to 30/9/2020. Bottom right: 7-day moving average observations minus BAU-predicted 1/3/2020 to 30/9/2020.

4.3 Observed concentrations of PM_{10} and $PM_{2.5}$ during COVID-19 Alert Levels compared to long-term mean

Random forest predictive models built using temporal and meteorological variables did not explain sufficient variation in PM to be useful for predicting BAU. Therefore, a simpler approach of comparing average particulate matter concentrations during each Alert Level with the average for the same period during the last four years was used. Comparisons between the short term (2020) period and long-term period (2016 to 2019) are shown as a percentage change and difference in average concentration where the difference between periods was different from zero at the 95% confidence level (Student's t-test) (Figure 4.14 and Appendix A10). Box plots showing the distribution of 24-hour average PM_{10} and $PM_{2.5}$ by Alert Levels for the two time periods are shown in Appendix A11.

Figure 4.14: Percentage difference[7] between observed PM and long-term (2016 to 2019) mean across all three sites by Alert Level. Where the percentage difference was not different from zero at the 95% confidence level, the percentage difference is shown as 0%.

To gain further insight on the potential reasons for changes in PM from the long-term average, diurnal plots comparing observations during Alert Levels to those in the past four years were also produced (Figures 4.15 to 4.17).

[7] LTM – 2020 observations / LTM * 100

PM_{10} at Wellington central was on average lower during L4, L3 and L2 compared to the same period from 2016 to 2019, with the largest reduction of 17% (-2.0 µg/m^3) observed in L4. $PM_{2.5}$ at Wellington central was 15% (-0.7 µg/m^3) lower in L4 but not statistically different from the long term average during the other Alert Levels. The magnitude of percentage reductions in PM during L4 are much smaller than was observed for the other traffic-related air pollutants (NOx, CO and black carbon) estimated when the impact of meteorology was accounted for by machine learning. Whilst $PM_{2.5}$ over L3, L2 and L1 was not, on average, different from previous years, the diurnal patterns appear to show a higher and later evening peak. The diurnal plots also show that the typical am and pm peaks were occurring later in the day (Figure 4.15).

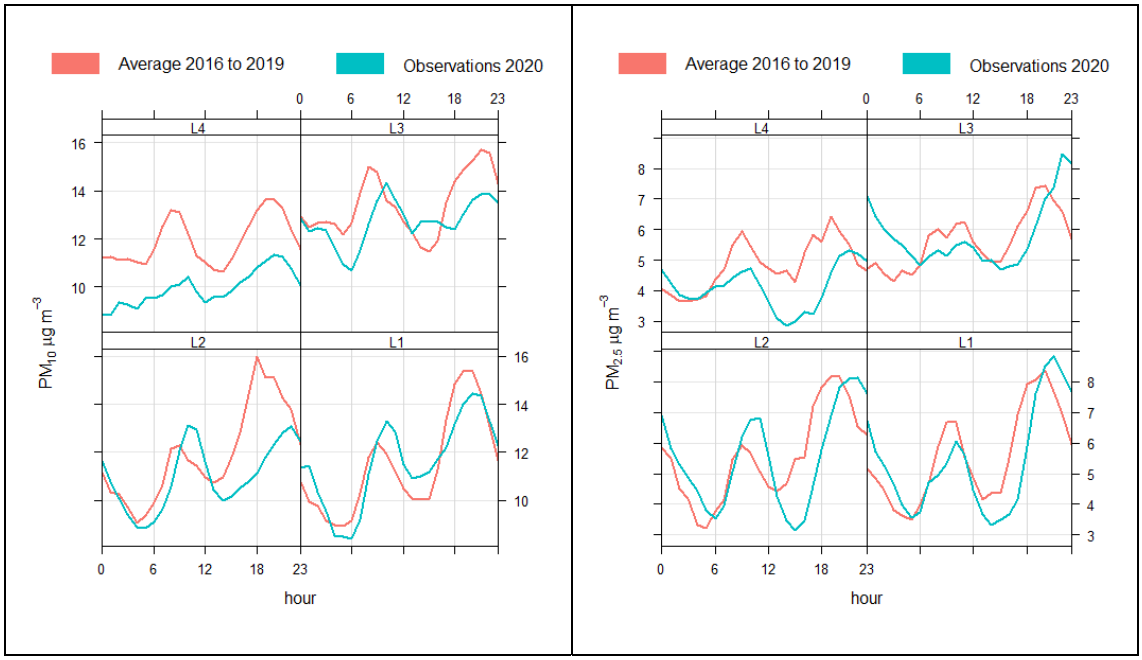

Figure 4.15: Wellington central PM_{10} (left) and $PM_{2.5}$ (right) diurnal plots

At Upper Hutt air monitoring station, observed PM_{10} was on average 15% lower in L4 (-1.4 µg/m^3) and 17% (-1.9 µg/m^3) lower in L3 and 13% lower (-1.6 µg/m^3) in L1. These reductions were much smaller than were observed for traffic-related NOx levels when meteorology was accounted for. The diurnal plots suggest a higher and later evening PM_{10} peak in L2 which is also seen in carbon monoxide consistent with home heating emissions and lower overnight temperatures (Figure 4.15). Although Upper Hutt appeared to have additional wood burning activity the PM_{10} winter average and winter daily PM_{10} maxima in 2020 were the same as in 2019 despite lower average wind speed than in the previous four years.

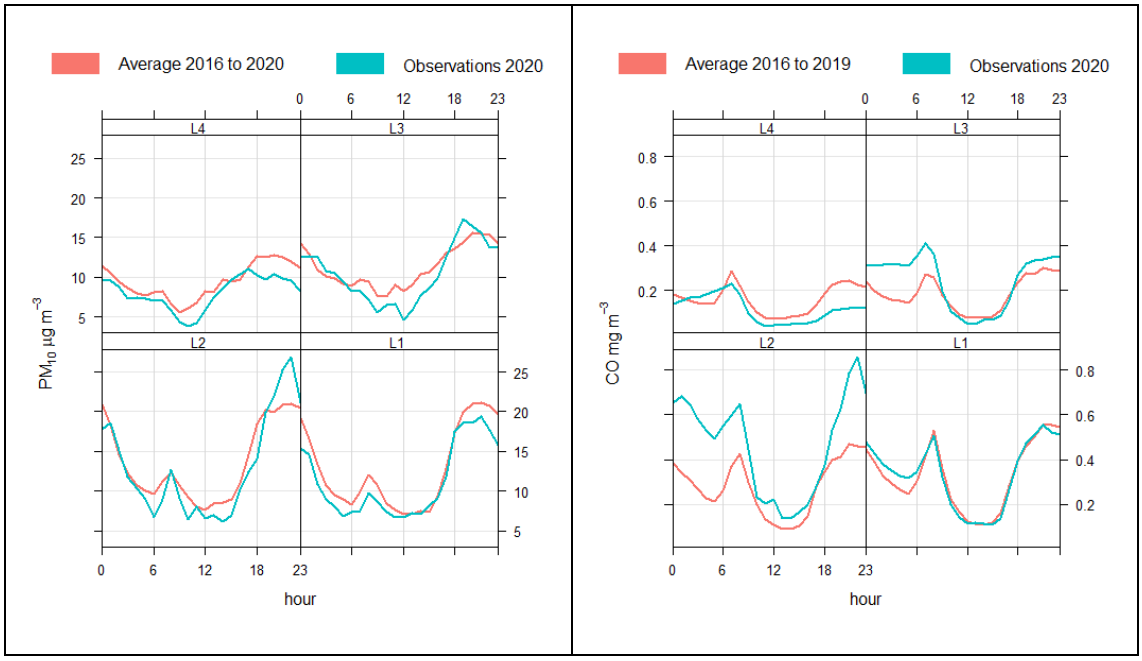

Figure 4.16: Upper Hutt PM_{10} (left) and carbon monoxide (right) diurnal plots

Masterton West experienced a 10% decrease (-1.2 µg/m^3) in average PM_{10} in L4, but saw a 16% (+2.3 µg/m^3) and 30% (+4.8 µg/m^3) increase above average in L3 and L2 respectively. The $PM_{2.5}$ increase in L3 and L2 coincides with the home heating season and appeared to be driven by some days with elevated evening peaks (Figure 4.16). The higher than average in $PM_{2.5}$ in L2 may be related to the lower than average air temperature and wind speed particularly for the overnight period. However the elevated $PM_{2.5}$ levels compared to previous years did not result in more high pollution days than typically observed in previous years. During winter 2020, Masterton West recorded 29 days that were above the World Health Organization guideline of 25 µg/m^3, four fewer than in 2019. Between 2016 and 2019, the annual number of PM2.5 'exceedances' has ranged from 19 in 2016 to 33 in 2019.

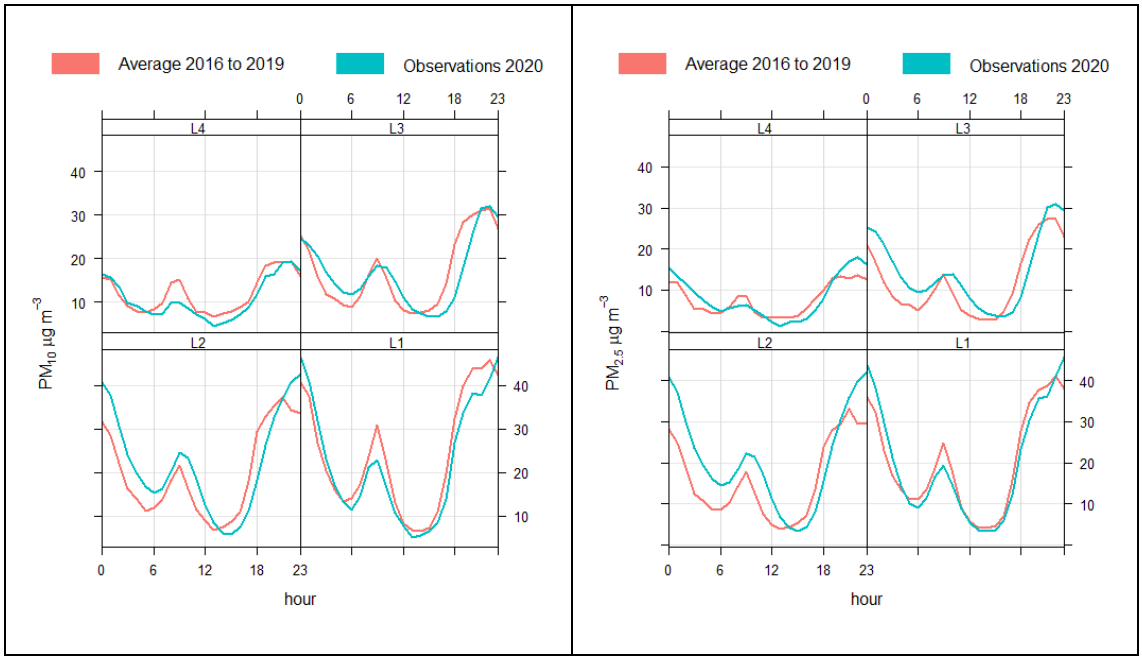

Figure 4.17: Masterton West PM_{10} (left) and $PM_{2.5}$ (right) diurnal plots

4.4 Traffic volumes during COVID-19 Alert Levels compared to 2019

From 2016 to 2019 daily and hourly traffic counts from the Terrace Tunnel and Ngauranga were only weakly or moderately correlated with daily and hourly pollutant levels at Wellington Central (Appendix A9). Finely resolved traffic data, ie, disaggregated by vehicle type and speeds, for the roadside and streets close to the monitoring station were not available to correlate with observed pollutant levels. In this study the traffic counts at Ngauranga and Terrace Tunnel were used to give a general indication of changes from BAU (2019) in traffic activity arising from different Alert Levels (Table 4.1).

Traffic volumes fluctuate depending on time of year being lower during Easter and the December/January holiday period. For comparison with the air pollutant data, 7-day average traffic counts in 2020 are shown as a percentage of 2019 values for Ngauranga SH1 southbound split by heavy and light duty vehicle fleet (Figure 4.11).

Table 4.1: Percentage difference between average daily traffic in 2020 and 2019 by Alert Level at NZTA traffic monitoring sites with continuous data available

Location	AADT (2019)	L4	L3	L2	L1	L2/3
Terrace Tunnel southbound NZAT Ref. 01N11074	20,925	-78.8	-38.4	-9.2	-3.6	-5.8
Terrace Tunnel northbound NZTA Ref. 01N21074	26,571	NA	NA	NA	NA	NA
Ngauranga SH1 southbound NZTA Ref. 01N11068	23,166 (light 96.4%)	-82.7	-54.7	-20.1	-2.6	-4.8
	865 (heavy 3.6%)	-77.2	-20.9	-5.5	+2.3	-6.7
Ngauranga SH2 southbound NZTA Ref. 00210979	22,057 (light 95.6%)	-84.2	-53.7	-20.4	-1.9	-4.3
	1,015 (heavy 4.4%)	-80.9	-22.9	-19.1	-19.1	-26.5

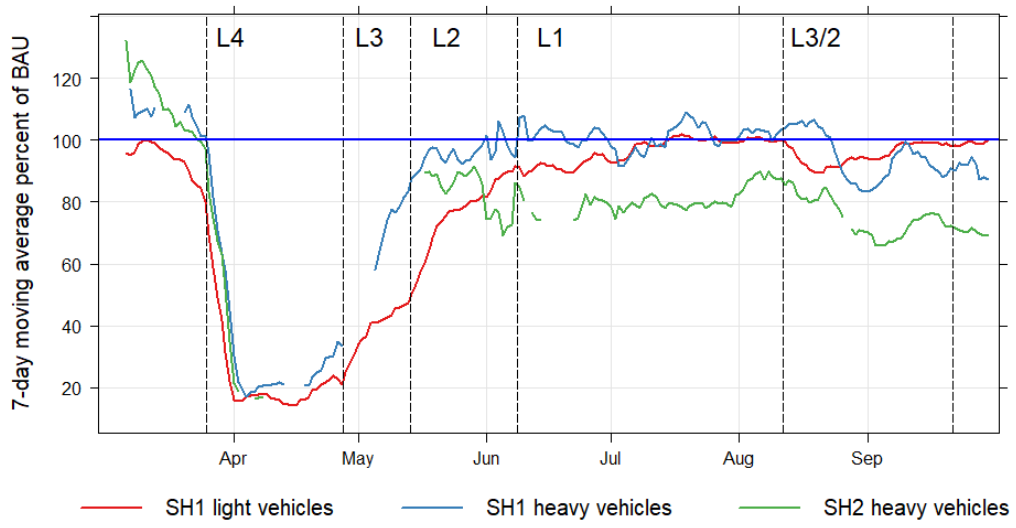

Figure 4.18: 7-day moving average NZTA SH traffic counts as a percentage of 2019 counts by Alert Level 1/3/2020 to 30/9/2020

4.5 Meteorology during COVID-19 Alert Levels compared to previous years

The Random Forest models with R^2 above 0.75 were able to adequately account for the impact of meteorology on air pollutant concentrations. However, predicting particulate matter concentrations from temporal and meteorological variables was less successful as the influence of meteorology depends on the source of PM and its size fraction (coarse or fine). A semi

quantitative analysis was undertaken to investigate whether the meteorological conditions during the different Alert Level periods differed significantly from the same period in previous years. Wind roses showing wind speeds and directions measured at each monitoring site by Alert Level compared to the previous four to five year period are shown in Appendix 12. Differences between average wind speed and temperature by Alert Levels compared to historical values are shown in Appendix 13 and in wind speed and temperature diurnal profiles to aid interpretation in Appendix 14.

A regional analysis of seasonal minimum and maximum temperature and daily mean wind anomalies against the 1981 to 2010 reference period found that during autumn 2020 (MAM) regional temperatures were warmer on average; although Masterton experienced cooler than average night-time temperatures. Virtually all of the region experienced below average winds due to the influence of a blocking anticyclone east of New Zealand (GWRC 2020a). The winter period (JJA) was characterised by a long corridor of anomalous high pressure extending all the way from the sub-Antarctic waters south of Australia to the east of New Zealand. This set up contributed to weakening the normal westerly flow and the southerly fronts, bringing very mild temperatures and prolonged dry periods. Warmer than average temperatures continued for the region, although Masterton night-time temperatures were much closer to average, Virtually all the region experienced well below average wind speeds as a result of the influence of the blocking anticyclone east of New Zealand, with fewer frequent fronts and prolonged stable periods (GWRCb).

The regional analysis was not always consistent with comparisons with previous years based on local meteorological observations from the air monitoring stations because the air monitoring station observations used a shorter and more recent reference period (ie, the previous four or five years) and examined different time periods (ie, duration of different Alert Levels rather than a seasonal average). Compared to previous years, Wellington central, had average wind in L4 but lower than average winds in L3, L2 and L1 and temperatures were cooler than average during L4 and L3. Upper Hutt experienced slightly above average wind speeds in L4 and below average winds in L3, L2 and L1. In Upper Hutt, temperature was cooler than average in L4, L3 and L2, particularly for over-night temperatures. Compared to previous years, Masterton West was windier in L4, the same in L3, and less windy in levels L2 and L1. Masterton air temperatures were cooler in L4, L3, and L2 and about average in L1.

4.6 NO_2 passive diffusion tube monitoring

Due to COVID-19 movement restrictions the NO_2 tubes were unable to be exchanged and therefore were exposed for a two-month period (March to April 2020) instead of the scheduled monthly exchange, which is +/- two days of the first Wednesday of each calendar month. The NO_2 tubes were exposed for 64 days from 2-3 March 2020 to 4 May 2020 which included 33 days of Alert Level 4 and the first nine days of Level 3.

The impact of the COVID-19 lockdown on NO_2 levels measured by the passive monitoring tube network sites was estimated by comparing the two-month average for the March to April 2020 period to that measured in the previous year (March-April 2019). It was not possible to account for differences in meteorology between the two years as the 2-month resolution is too coarse.

Figures 4.19 to 4.21 show the percentage reduction in 2020 NO_2 concentrations relative to 2019. Across all regional monitoring sites (not shown) the percentage reductions ranged from 31% to 67%. The percentage reduction for the average of the triplicate NO_2 tubes (WEL073-WEL075) collocated at Wellington Central air monitoring station was 42% which was also the median percentage reduction for all NO_2 tube sites. Traffic reductions (as measured at Ngauranga SH1 southbound traffic counter) for the March to April period in 2020 were 54% lower than in 2019. It is also likely to be larger uncertainty associated with a 2-monthly average compared to an annual average.

These results show there was high spatial variation in the reductions in traffic-related air pollution across the region during March-April 2020. Without site-specific traffic data, it was not possible to explore whether this variation was due to local traffic changes that may not be captured by the NZTA state highway traffic counts. However, there was a strong linear relationship (R^2=0.90) between March to April concentrations measured in 2019 and the absolute reduction in NO_2 concentration for each site measured in 2020 (Appendix A15). This suggests that the NO_2 reductions observed across the network were proportional to the magnitude of 2019 concentrations.

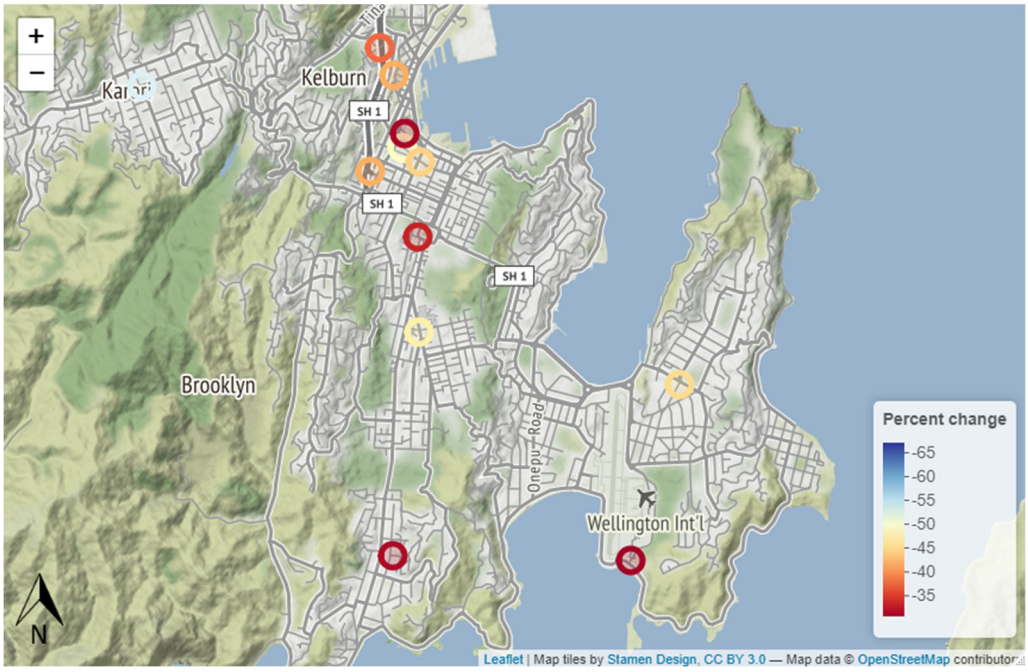

Figure 4.19: Wellington City percentage reduction in average NO_2 measured by passive diffusion tubes March-April 2020 compared to March-April 2019

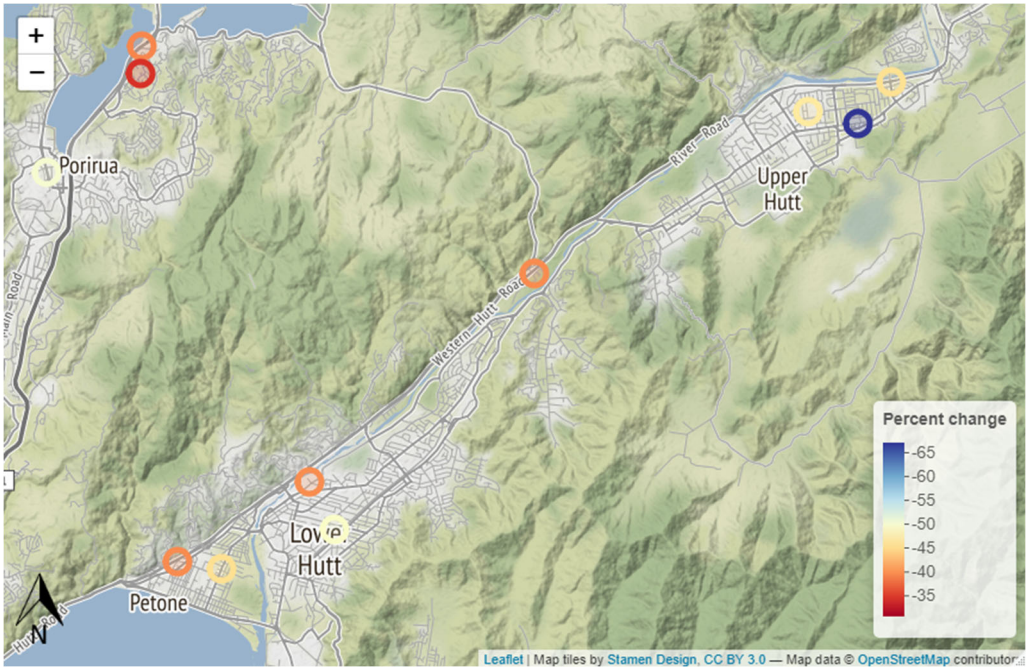

Figure 4.20: Hutt Valley and Porirua percentage reduction in average NO_2 measured by passive diffusion tubes March-April 2020 compared to March-April 2019

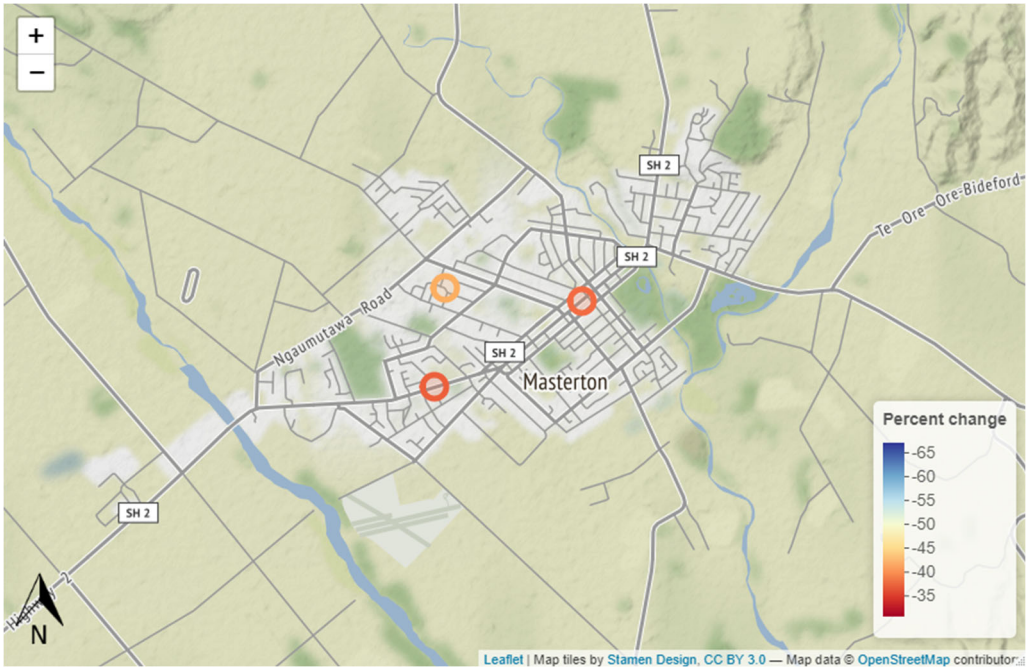

Figure 4.21: Masterton percentage reduction in average NO2 measured by passive diffusion tubes March-April 2020 compared to March-April 2019

5. Discussion

5.1 Changes in black carbon and NOx during COVID-19 Alert Levels

Black carbon and NOx at Wellington central air monitoring station had a similar magnitude of reduction (as a percentage change from BAU-predicted) during L4 and L3 (Figure 5.1) which broadly matched the reduction in heavy vehicle traffic on southbound SH1 and SH2 (counted at Ngauranga) (Figure 4.18). During L4 concentrations of black carbon and NOx were both reduced by 71%. GNS Science radiocarbon analysis of weekly grass samples taken at Wellington central air monitoring station found fossil fuel derived CO_2 concentrations dropped by around 75% during L4 (Jocelyn Turnbull, pers. comm. 10/03/2021). This preliminary analysis suggests that roadside black carbon and NOx measurement may be a useful proxy for tracking changes real-world traffic-derived CO_2 measurements. More research is needed to investigate the relationships between local air pollutant and CO_2 measurements.

At Wellington central, BC and NOx followed broadly the same trend until the beginning of September 2020, after which their patterns diverged (Figure 5.1). NOx remained elevated whist BC concentrations declined. It is not possible to explain this divergence using available traffic count data. Black carbon levels returned to almost to BAU in L2 and then were higher than BAU at the beginning of L1. This may reflect an increase in local traffic around Wellington city and the monitoring site that is not captured in the SH count data and/or a change in the usual fleet profile, for example, more light diesel commercial delivery vans. NZ post reported that online retail sales peaked in late April as the country moved to L3 as non-essential purchases could then be delivered. Over L2 and L1 the weekly online spend continued to be well above where it was before lockdown (https://thefulldownload.co.nz/Covid-spotlight).

At Upper Hutt during L4 there was a 69% reduction in NOx compared to BAU-predicted. In contrast to Wellington central, NOx and NO_2 concentrations at Upper Hutt remained lower than BAU throughout L1 and L2/3. This difference between the roadside site in Wellington and urban background site in Upper Hutt is probably a reflection of the Wellington site being influenced by a busy arterial route on which traffic flows quickly returned to near normal, whereas the urban background site is located in a park area away from the direct road influence and so represents the impact of diluted and dispersed aggregate traffic emissions from further afield. Masterton West, whilst showing a similar magnitude in NOx reduction as Wellington central and Upper Hutt during L4, it appears to show a smaller reduction in NOx and NO_2 than seen at Upper Hutt over L3, L2 and L1.

Black carbon data is available in near real-time as the data, and unlike carbon monoxide and NOx, does not require adjustment for instrument drift based on results of monthly or three-monthly calibrations. Black carbon is also chemically inert and is not influenced by atmospheric chemistry. However, there is a relatively short record of black carbon measurements available for building a predictive model that accounts for meteorology which introduces uncertainty

in the model for predicting concentrations under meteorological conditions not used for model training. Furthermore, there is only one permanent monitoring station that measures black carbon so it is uncertain how generalisable the black carbon results are to other roadsides in the region.

At Wellington central BC, NOx and NO_2 were substantially higher than predicted for the meteorological conditions on Monday 8 June 2020 (last day of Alert Level 2) and Tuesday 9 June 2020 (first day of Level 1) despite traffic volumes not being back to expected levels on these two days. This may have been due the combination of low wind speeds on those days (0.8 and 0.7 m/s) as well as a possible increase in local traffic activity not captured by the SH traffic counting sites used for comparison.

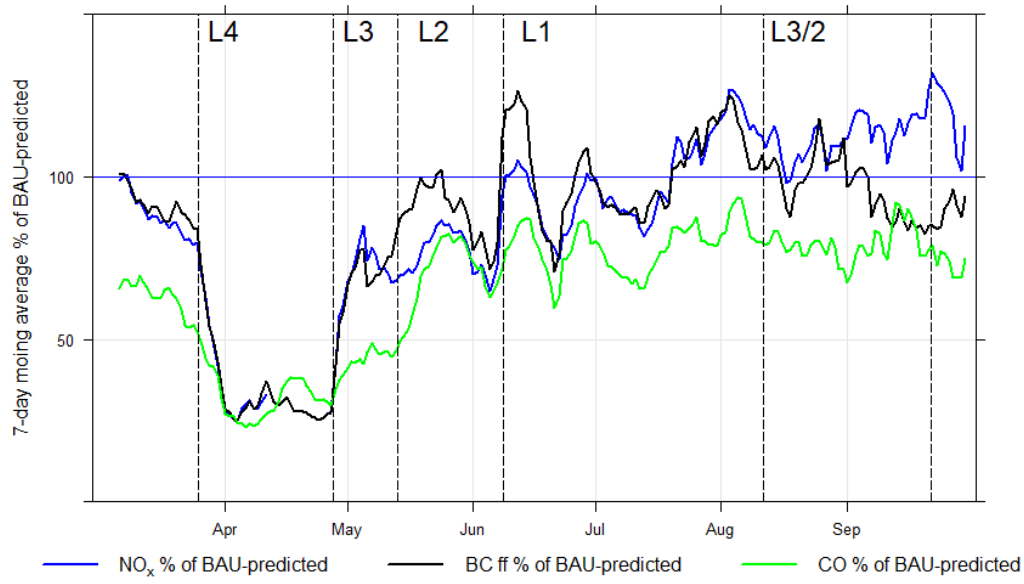

Figure 5.1: Wellington Central 7-day moving average NOx, black carbon (fossil-fuel) and carbon monoxide (CO) shown as percentage of BAU-predictions with the blue line at 100% which is the air pollutant concentration expected for the time of year and weather conditions under typical (pre COVID-19) traffic emissions

5.2 Changes in nitrogen dioxide and ozone during COVID-19 Alert Levels

Wellington central recorded a percentage decrease of 58.5% in NO_2 during L4 which is similar to that reported (56.9%) for a suburban roadside monitoring site in Henderson, Auckland with comparable ADDT to the Wellington Central site (Patel et al., 2020). The L4 lockdown had the effect of reducing NO_2 levels at the Wellington central roadside to what would typically be measured at the Upper Hutt urban background site for that time of year (ie, 5-7 µg/m^3). Wellington central also measured enhanced ozone levels (+11%) during L4 as reductions in primary NO emissions from lower traffic volumes meant less ozone was being depleted through NOx titration. Higher than expected ozone levels due to reduced NOx during lockdowns overseas have also been reported (Lovrić, et al. 2020).

At Wellington central and Upper Hutt during L4 and L3 NO_2 reductions, as a percentage of BAU-predicted, were less pronounced compared to NOx (Figure 5.2 and 5.3). This can be explained by the non-linear atmospheric chemistry involved when traffic-emitted NO is rapidly oxidised to NO_2 through a reaction with ozone. At Wellington central although concentrations of NO from March to May 2020 were lower than previous years (Figure 5.4) proportionally more of the emitted NO was able to be converted NO_2 as the reaction was not ozone-limited. This finding is consistent with a higher NO_2/NOx ratio from March to July in 2020 compared with previous years (Figure 5.5). Overseas studies have also reported smaller reductions in NO_2 concentration compared to reductions in NOx concentrations attributable to shifts in the NO_2/NOx ratio in favour of NO_2 (Shi, et al. 2021).

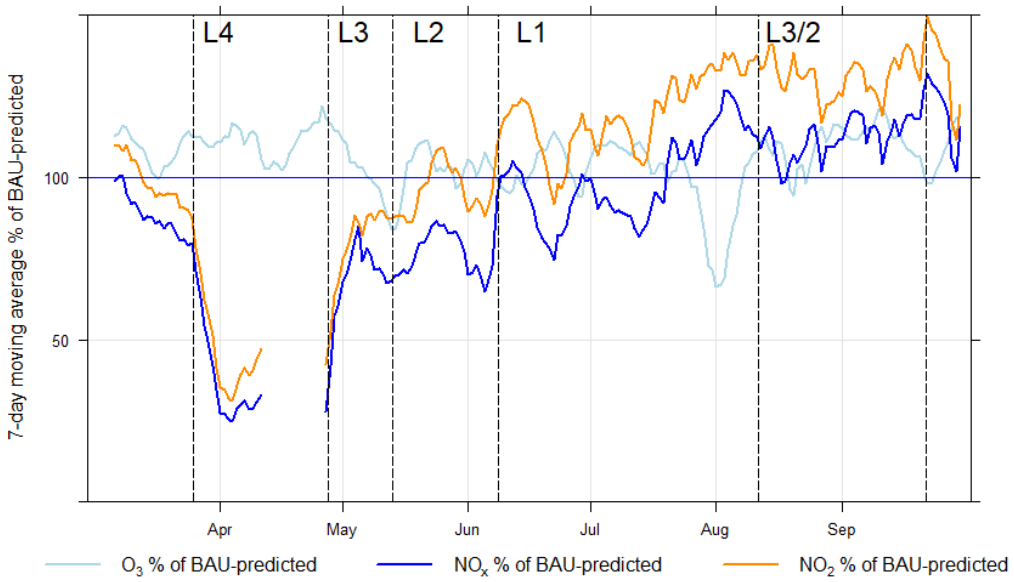

Figure 5.2: Wellington Central 7-day moving average NOx, nitrogen dioxide (NO_2) and ozone (O_3) shown as percentage of BAU-predictions with the blue line at 100% which is the air pollutant concentration expected for the time of year and weather conditions under typical (pre COVID-19) traffic emissions

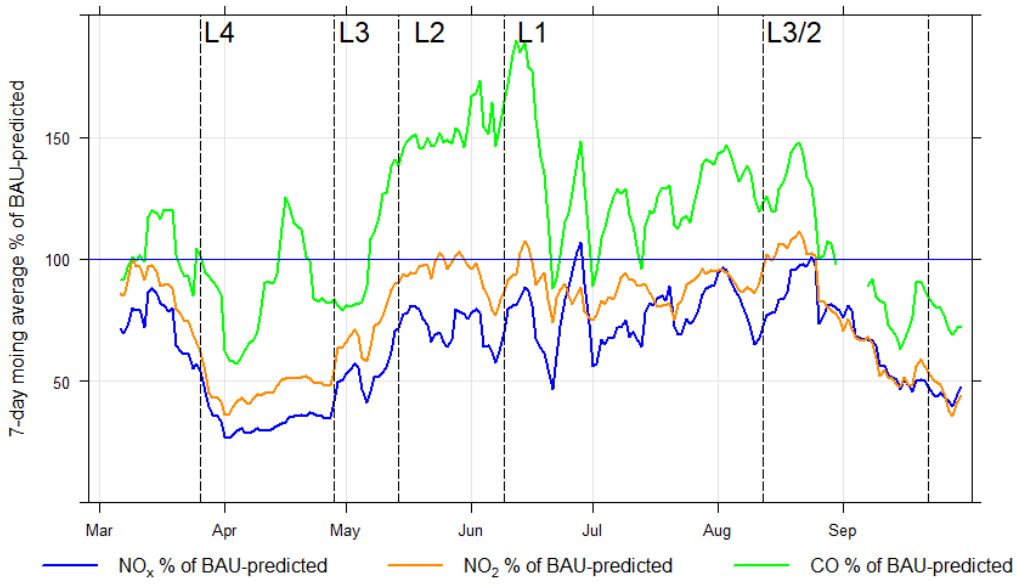

Figure 5.3: Upper Hutt 7-day moving average NOx, nitrogen dioxide (NO_2) and carbon monoxide (CO) shown as percentage of BAU-predictions represented with the blue line at 100% which is the air pollutant concentration expected for the time of year and weather conditions under typical (pre-COVID-19) traffic and home heating emissions

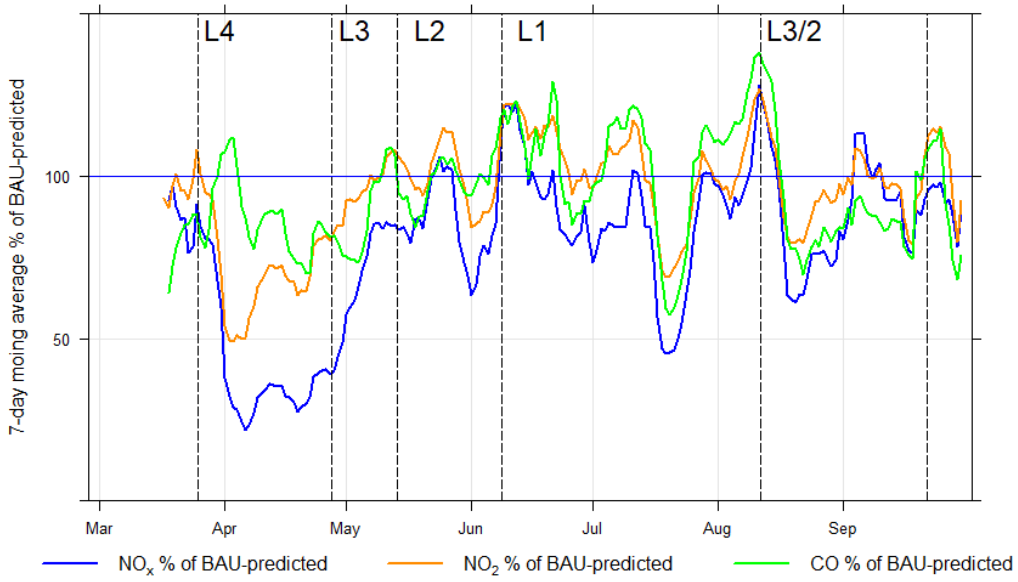

Figure 5.4: Masterton West 7-day moving average NOx, nitrogen dioxide (NO_2) and carbon monoxide (CO) shown as percentage of BAU-predictions represented with the blue line at 100% which is the air pollutant concentration expected for the time of year and weather conditions under typical (pre-COVID-19) traffic and home heating emissions

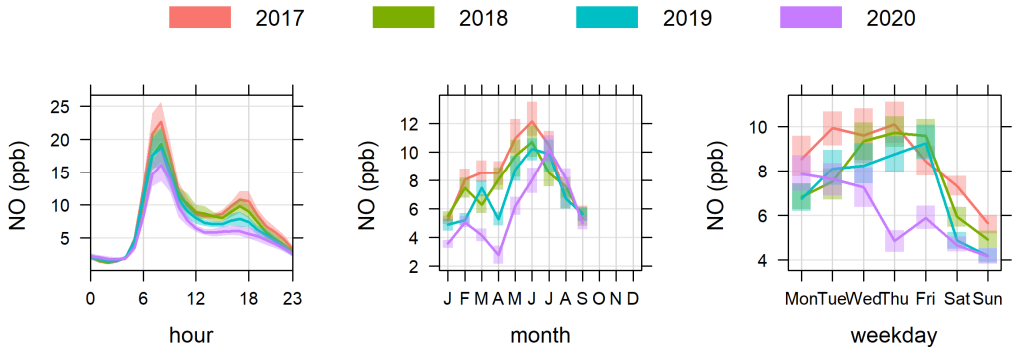

Figure 5.4: Wellington Central diurnal, monthly and weekly variation in NO (ppb) by year January to September. Shading shows 95% confidence interval in mean

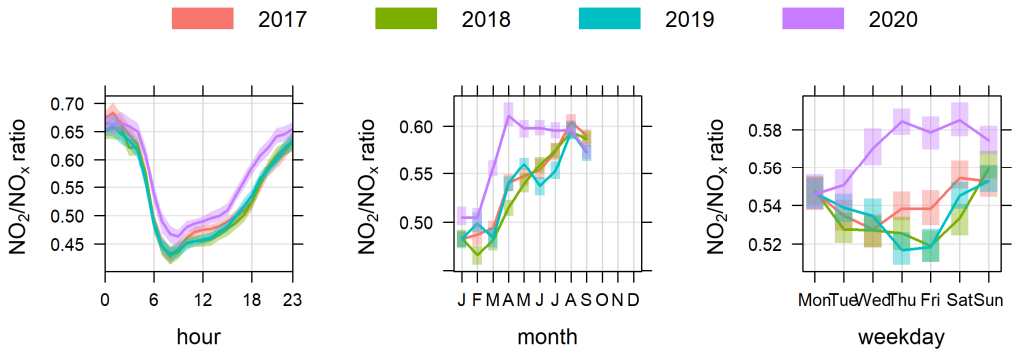

Figure 5.5: Wellington central diurnal, monthly and weekly variation in NO_2/NOx (ppb) ratio by year January to September. Shading shows 95% confidence interval in mean

Unexpectedly NO_2 levels at Wellington central were higher than BAU-predicted in L1 and L2/L3 despite no significant increase in SH traffic counts (measured at Ngauranga). This is attributed to a combination of increase in local traffic activity during L1 as suggested by the black carbon data, impact of atypical meteorology not captured by the model's training dataset and enhanced conversion of NO to NO_2 due to high background ozone levels in September 2020.

The RF modelling showed that wind speed is more important than hour of the day (a proxy for emissions) for predicting NO_2, whereas for NOx the reverse is true. Therefore, it is suggested that the higher than BAU-predicted NO_2 levels in L1 are due to meteorological effects not previously captured in the RF models. There was a run of days from 24 July to 3 August 2020 where wind speeds were very low coinciding with high pressure system that lasted for several days resulting in stable atmospheric conditions that typically restrict dispersion and dilution of air pollutant emissions as well as favour ongoing accumulation of NO emissions for conversion to NO_2. Over this period all traffic-related pollutants were elevated and ozone was noticeably depleted.

Measured ozone concentrations in September 2020 (L3/L2) were higher than recorded in the previous two years (Figure 5.6). This could be due to enhanced

stratospheric ozone intrusion, ie, naturally occurring high altitude ozone being transported into the lower atmosphere due to strong gusty westerly winds associated with spring (Adeeb & Shooter, 2004). In Wellington September 2020 was characterised by a "fierce" westerly regime, the windiest since 1990 (Pezza, 2020). If these extreme wind conditions lead to ozone enhancement, then conversion of precursor NO emissions to NO_2 would not be limited by ozone availability resulting in elevated NO_2. However, higher wind speeds also lead to dispersion and therefore dilution of primary NO emissions which lead to lower NO_2 concentrations.

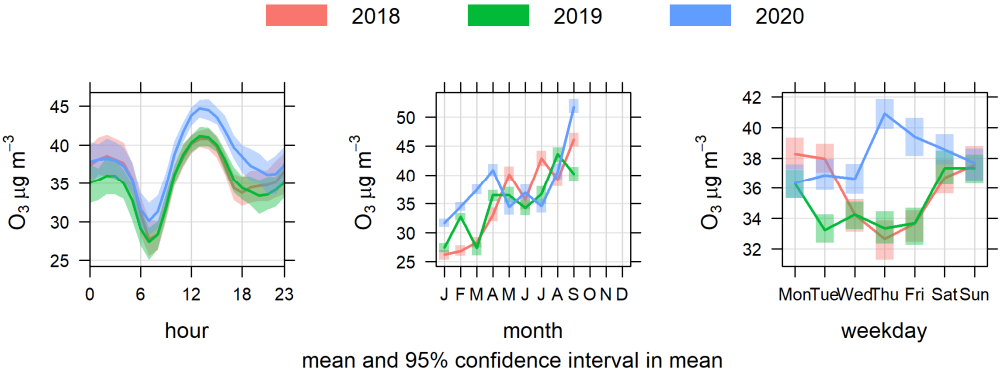

Figure 5.6: Wellington central diurnal, monthly and day of the week variation in ozone (O_3) by year (January to September)

5.3 Changes in carbon monoxide during COVID-19 Alert Levels

Pre COVID-19 and through all the Alert Levels, carbon monoxide concentration measured at Wellington central air monitoring station remained lower than predicted by BAU scenario modelling (Figure 5.1). Carbon monoxide was already lower than normal from 23 December 2019 onwards, most likely due to disruptions to local traffic on nearby Willis Street as sections of road were closed for works needed fix a collapsed wastewater underground pipe.

At the Wellington central air monitoring station the reductions in carbon monoxide (as a percentage change from BAU) closely matched the percentage reductions in daily traffic counts on southbound SH1 and SH2 (measured at Ngauranga) during L4, L3 and L2. However, carbon monoxide levels remained about 20% lower than normal during L1 and L2/L3 despite SH traffic levels being on average 2% to 4% lower than previous year. Smoothed 7-day average carbon monoxide appears to follow the same general upward trajectory as light vehicle counts as SH1 southbound (Ngauranga), but unlike traffic counts, did not return to 100% of BAU during the latter part of L1 (Figure 4.6). Carbon monoxide concentrations may be responding to reduced light vehicle passenger journeys as a consequence of an estimated 13% of Wellington private vehicle commuters continuing to work from home (Waka Kotahi NZ Transport Agency, 2020).

Carbon monoxide is an indicator of petrol-fuelled vehicles but is no longer widely measured in New Zealand as the concentrations are well below the National Environmental Standard and ambient air quality guidelines due to

improved vehicle technologies (Bluett et al. 2016) and progressively stricter emission limits (Bromley & Gray 2017). This analysis shows that, despite low concentration, carbon monoxide remains a useful indicator for traffic-related emissions.

Carbon monoxide at Upper Hutt showed a very different trend than Wellington central with higher than expected concentration in L3 and L2. The increase in CO coincided with the colder winter months and may be the result of increased home heating emissions due to more 'stay at home' behaviour. The Upper Hutt urban residential site is more affected by wood burning than the Wellington roadside site. The 2020 winter diurnal pattern of CO concentration was distinctly different from previous years, showing a higher overnight component and larger morning peak. This may be due to people staying up later and re-lighting their fires in the morning instead of leaving home for work. The evening PM_{10} concentration peak was relatively high over L3 and L2 compared to previous years, which is consistent with diurnal profile for wood burner emissions. This analysis shows carbon monoxide is a useful indicator of home heating emissions in an urban area impacted by home heating emissions but with comparatively low PM_{10} concentrations.

5.4 Changes in PM_{10} and $PM_{2.5}$ during COVID-19 Alert Levels

Ambient particulate matter is composed of a mixture of sources that are emitted at different times (eg, by hour of day and by season) and arise from both local (eg, wood burning) and regional sources (eg, marine aerosol). The response of PM to meteorology varies by source and time, for example, marine aerosol is ubiquitous in the region and fairly constant throughout the year. However, on shorter time scales, wind speed is an important influence, with higher marine aerosol under high wind speeds and at night higher particulate from wood burning under low wind speeds. RF performs better when there is a dominant source of PM that responds to meteorology in a consistent way.

PM was reduced compared to previous years at all three sites, particularly in L4, which is attributed to lower re-suspended road dust and traffic exhaust particulate correlating with the dramatic reduction in vehicle movements. This is particularly evident at the Wellington central air monitoring site, which is close to the roadside, and therefore expected to have a larger contribution of traffic-derived particulate than the other sites which are more distant from busy roads. The Wellington central site showed a return to 'average' PM_{10} in L1 coinciding with the return to near-normal traffic counts on the state highway. The relatively smaller reduction in PM compared to the other gaseous traffic-related pollutants is attributed to a larger proportion of PM arising from natural sources than from traffic-related sources.

Masterton West, which typically experiences elevated winter PM due to wood burning, recorded above average $PM_{2.5}$ in Levels 3 and 2. During L2, elevated evening and overnight concentrations were observed which may be linked to more "stay at home' behaviour as well as lower than average overnight temperatures and wind speed increasing atmospheric stability with less

dispersion of emissions occurring. A possible increase in wood burning during L2 was also seen at Upper Hutt, with a higher than average evening PM and carbon monoxide late evening peak. The pollution rose shows lower wind speed and a greater proportion of wind arose from the north-east sector than usual (Figure 5.7). As seen in past years, the north-east direction contributes the most to overall PM concentrations, most likely due to cold air katabatic drainage from the valley head leading to an accumulation of home heating emissions at the Upper Hutt monitoring site.

Likewise, at Wellington central there appeared to be a small increase in $PM_{2.5}$ evening peak in L2 and L1 which may be related to wood burning as the same higher than usual evening peak also appears in black carbon apportioned to biomass burning (Figure 5.8). The impact of the potential increase in wood burning tentatively ascribed to more stay at home behaviour, did not result in any PM_{10} or $PM_{2.5}$ exceedances, despite the lower than average wind speeds across the region and at these sites.

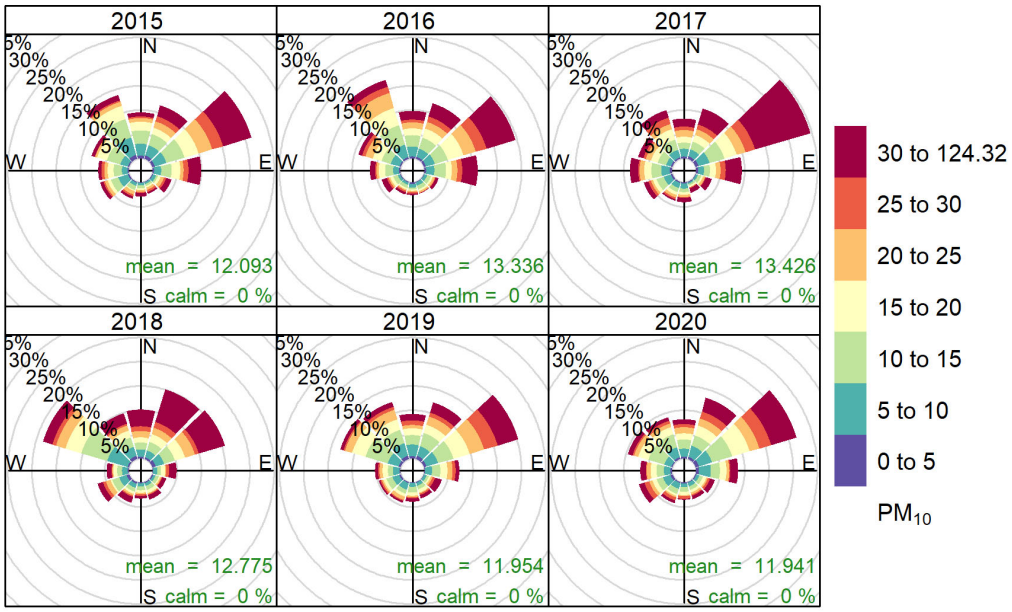

Figure 5.7: Upper Hutt PM_{10} pollution rose showing the percentage contribution to average winter PM10 (May to August) by wind direction sector

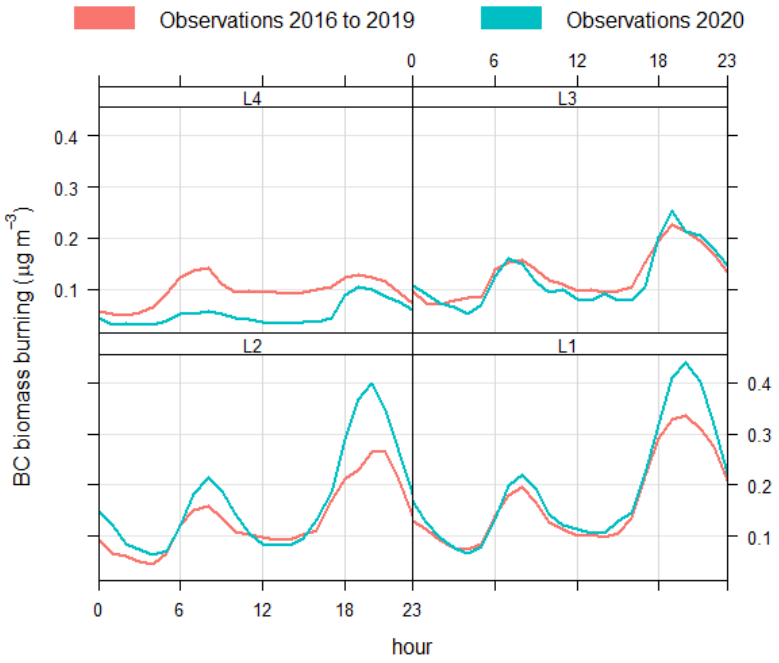

Figure 5.8: Wellington central black carbon from biomass burning diurnal profile by Alert Levels

5.5 Study limitations

As shown by the NO_2 passive diffusion tube network, the likely impacts of changes in traffic activity during Alert Levels on air pollution across the region vary by location. Therefore, results from three air monitoring stations will not capture the range of impacts. Compared to many other countries, NZ has had a relatively short period of disruption which can make trend detection more difficult. There is also the unanswered question of what percentage change or absolute change in pollutant concentrations are environmentally significant versus statistically significant.

As the RF models predict a mean response for pollutant levels there will be more uncertainty in the predictions that for minima and maxima pollutant concentrations. Furthermore, although the models explained between 75% and 88% of the variation, there is still un-explained variation (noise) present. Not all meteorological variables were available for modelling, for example, rainfall which has an important role in removal of air pollutants, especially particulate matter, including black carbon (Cape et al., 2012). Furthermore, there are other meteorological features, such as temperature inversion strength or boundary layer height, which are very relevant for air pollution episodes, but were not available to be used as predictor variables.

Another significant limitation is the lack of site-specific traffic profile data which if available may be able to be more strongly correlated with air pollutant measurements. In future modelled traffic emissions such as annual NOx (kg/km/day) will available at for road links from the NZTA Vehicle Mapping

Tool[8] currently in development. These emissions estimates may prove useful for matching with observed NOx and NO_2 concentrations.

[8] https://www.nzta.govt.nz/roads-and-rail/highways-information-portal/technical-disciplines/air-quality-climate/planning-and-assessment/vehicle-emissions-mapping-tool/

6. Conclusion

This study has shown how concentrations of air pollutants at a roadside monitoring site in Wellington city and urban background sites in Upper Hutt and Masterton responded differently to changes in emissions resulting from the travel, work and social disruptions under COVID 19 Alert Levels. The key findings for the two main sources of pollutant emissions are as follows:

a) Traffic-related air pollutants and emissions

- A significant reduction in directly emitted tail-pipe traffic-related pollutants (NOx, carbon monoxide and black carbon) occurred during L4 due to dramatically decreased traffic volumes when meteorological effects were accounted for.

- During L4 the reduction in nitrogen dioxide was less pronounced than the other traffic-related air pollutants (black carbon and carbon monoxide) as there was a non-linear response between emissions and pollutant concentrations due to atmospheric chemistry. This phenomena led to higher than expected ozone levels in L4 and higher than expected nitrogen dioxide levels near the roadside as traffic returned to near normal levels in L1.

- There was spatial variation in magnitude of reductions in nitrogen dioxide between sites across the passive diffusion tube monitoring network, but no obvious patterns were found.

- The observed reductions in PM_{10} during L4 were much less marked than for the other traffic-related air pollutants which is due to PM also arising from non-transport related sources, such as marine aerosol. The L4 reduction in PM_{10} and $PM_{2.5}$ is most likely due to lower levels of re-suspended road dust and PM material from other sources such as construction or road works.

b) Home heating-heated pollutants and emissions

- Increased stay at home behaviour may have increased the frequency and duration of wood burner use compared to previous years leading to higher than expected residential emissions of particulate matter and carbon monoxide from solid fuel fires. There may have also been an impact of lower than average wind speeds during the winter period leading to conditions that favour build-up of particle air pollutant levels.

The study confirms the usefulness of black carbon, carbon monoxide and nitrogen oxides as indicators of short term temporal trends in traffic-related emissions. The findings also highlight the need for black carbon, NOx and CO monitoring at key passive NO_2 monitoring roadside to help evaluate impacts of changes in mode shifts to active travel, public transport, decarbonisation and new travel patterns, such as people working from home. The NO_2 tube

monitoring data is still the most practical way of tracking trends in exposure to NO_2, especially in residential areas where people spend most of their time.

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Many thanks to Darren Li and Rhys Evans (GWRC Environmental Science) for operating and maintaining air quality monitoring sites and instruments. Thank you to Dr Perry Davy (GNS Science) for providing information on the sources of particulate at Wellington Central and for useful feedback and comments on the analysis. Thank you to Dr Nick Talbot (independent consultant and Auckland University researcher) for peer review and recommendations which greatly improved this report.

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Appendix A1: Wellington Central air monitoring site

A1.1: Wellington Central air monitoring site location (lat -41.293605, lon 174.771924)

A1.2: Wellington Central air monitoring station

Appendix A2: Upper Hutt air monitoring site

A2.1: Upper Hutt air monitoring site location (lat -41.121565, lon 175.070358)

A2.2: Upper Hutt air monitoring station

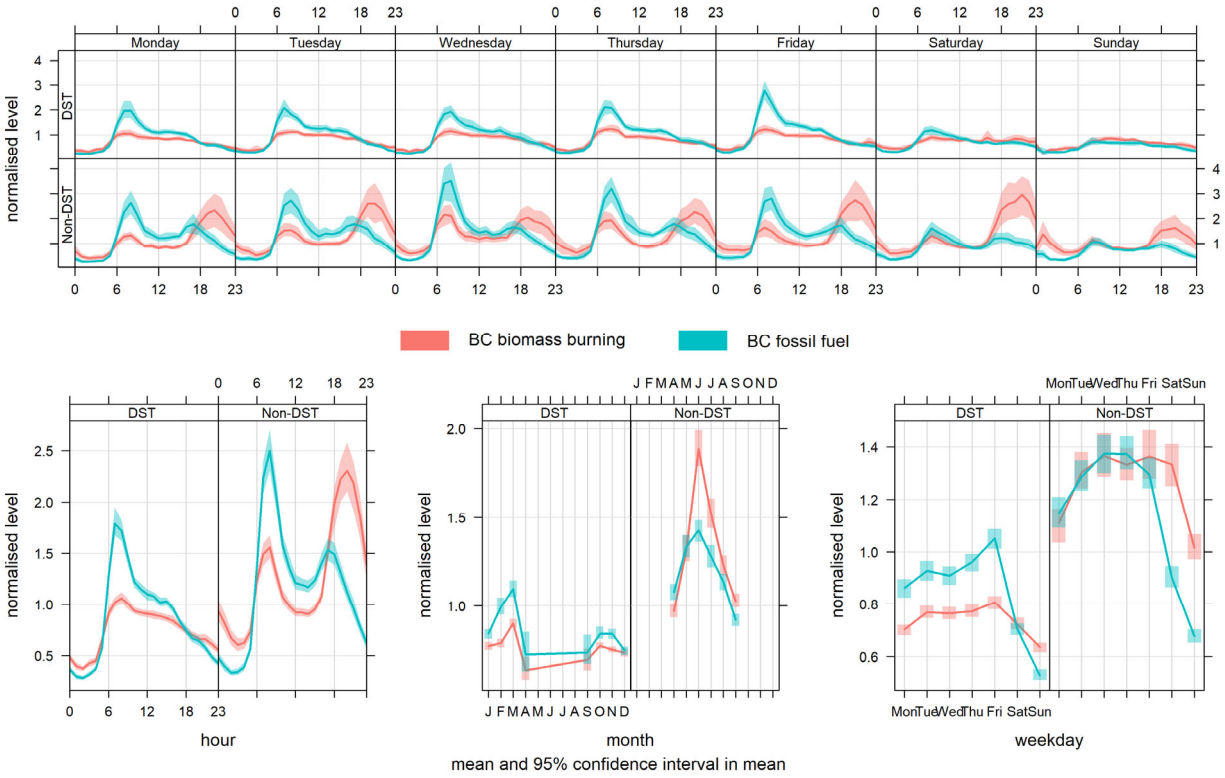
Appendix A3: Masterton West air monitoring site

A3.1: Masterton West air monitoring site location (lat -40.952364, lon 175.646546)

A3.2: Masterton West air monitoring station

Appendix A4: Wellington Central variation in black carbon by source

A4.1: Wellington Central time variation in fossil fuel derived and biomass burning derived black carbon (2017:2019). The observations are conditioned by daylight saving time with non-DST representing the period where wood smoke (biomass burning) may be present and DST representing the summer months without a wood burning source. The two sources are normalised as the wood smoke component is very low compared to the fossil-fuel derived component.



Appendix A5: Air pollutant datasets by air monitoring site

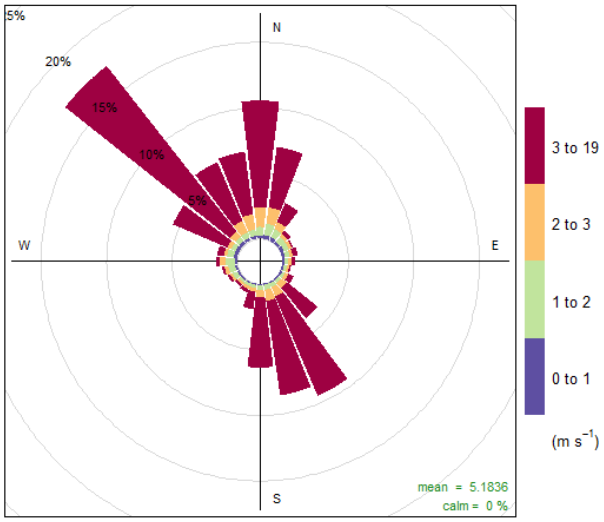
A5.1: Air pollutant data and monitoring methods

Site and pollutants	Method	Period
Wellington central		
Black carbon	Aethalometer AE33 (Magee Scientific)	1/1/2016 to 30/9/2020
NOx	Chemiluminsecence M200E (API)	1/1/2016 to 30/9/2020
CO	Gas Correlation M300E (API)	1/1/2016 to 30/9/2020
O_3	M400E (API)	12/1/2018 to 30/9/2020
PM_{10} and $PM_{2.5}$	5030 Sharp and 5014i (Thermo Scientific)	9/1/2016 to 15/1/2018 and 18/1/2018 to 30/09/2020
Upper Hutt		
NOx	Chemiluminsecence M200E (API)	1/1/2016 to 30/9/2020
CO	Gas Correlation M300E (API)	1/1/2016 to 30/9/2020
PM_{10}	FH62 (Thermo Scientific)	1/1/2016 to 30/9/2020
Masterton West		
NOx	Chemiluminsecence M200E (API)	1/1/2016 to 30/9/2020
CO	Gas Correlation M300E (API)	1/1/2016 to 30/9/2020
PM_{10} and $PM_{2.5}$	5014i (Thermo Scientific)	1/1/2016 to 30/9/2020

Appendix A6: Wind roses

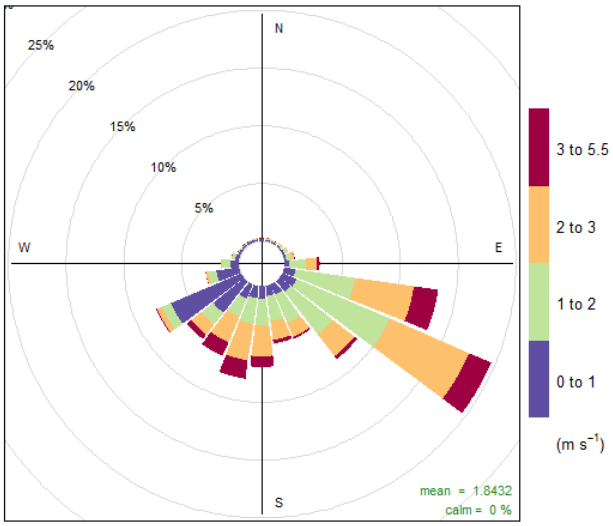
A6.1: Wind roses (2017 to 2019)

Kelburn weather station



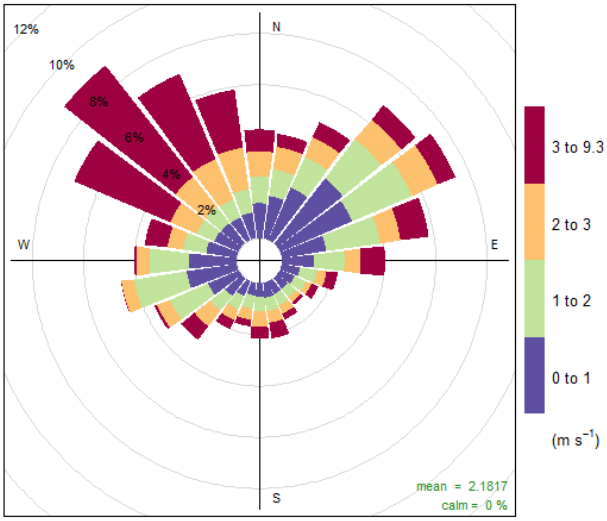
Frequency of counts by wind direction (%)

Wellington central air monitoring station



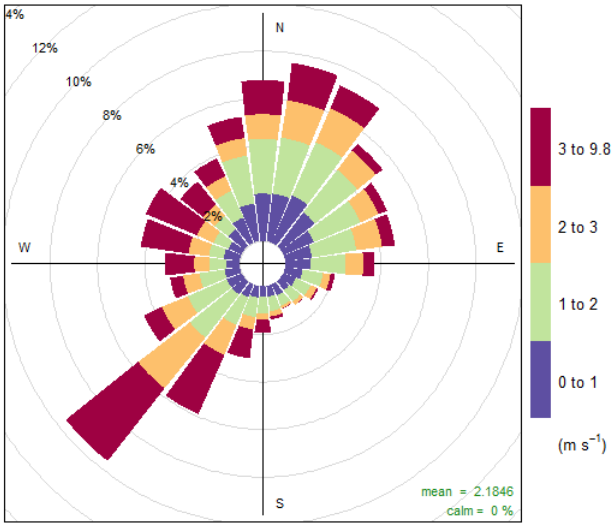
Frequency of counts by wind direction (%)

Upper Hutt air monitoring station



Frequency of counts by wind direction (%)

Masterton West air monitoring station



Frequency of counts by wind direction (%)

Appendix A7: Predictive models

A7.1: Random forest models grown for pollutants at each monitoring site

Monitoring sites and pollutants	n (hours)	R^2	Most important predictor variables
Wellington central			
Black carbon	28,404	0.79	hour, ws, weekday
BC fossil-fuel	28,404	0.79	hour, ws, weekday
BC biomass burning	28,403	0.75	hour, ws, temperature /Julian day
NOx	32,001	0.83	hour, ws, weekday
NO_2	31,426	0.85	ws, hour, Julian day
CO	31,885	0.82	hour, ws, Unix date
O_3	19,078	0.88	ws, Julian day, Unix date/ temperature
PM_{10}	28,098	0.65	Unix date, Julian day, temperature, rh
$PM_{2.5}$	27,659	0.45	Julian day, Unix date, temperature, hour
Upper Hutt			
NOx	26,887	0.78	ws, hour, Julian day, temperature
NO_2	30,389	0.80	ws, hour, temperature
CO	30,486	0.84	temperature, Julian day, ws
PM_{10}	27,354	0.59	hour, temperature, Julian day, rh
Masterton West			
NO_x	26,903	0.70	hour, ws, temperature, Julian day
NO_2	26,887	0.78	ws, hour, Julian day, temperature
CO	26,349	0.78	ws, temperature, Julian day, hour
PM_{10}	27,076	0.64	hour, temperature, Julian day, ws
$PM_{2.5}$	27,028	0.66	temperature, hour, Julian day, ws

Appendix A8: Percentage differences – gases and black carbon

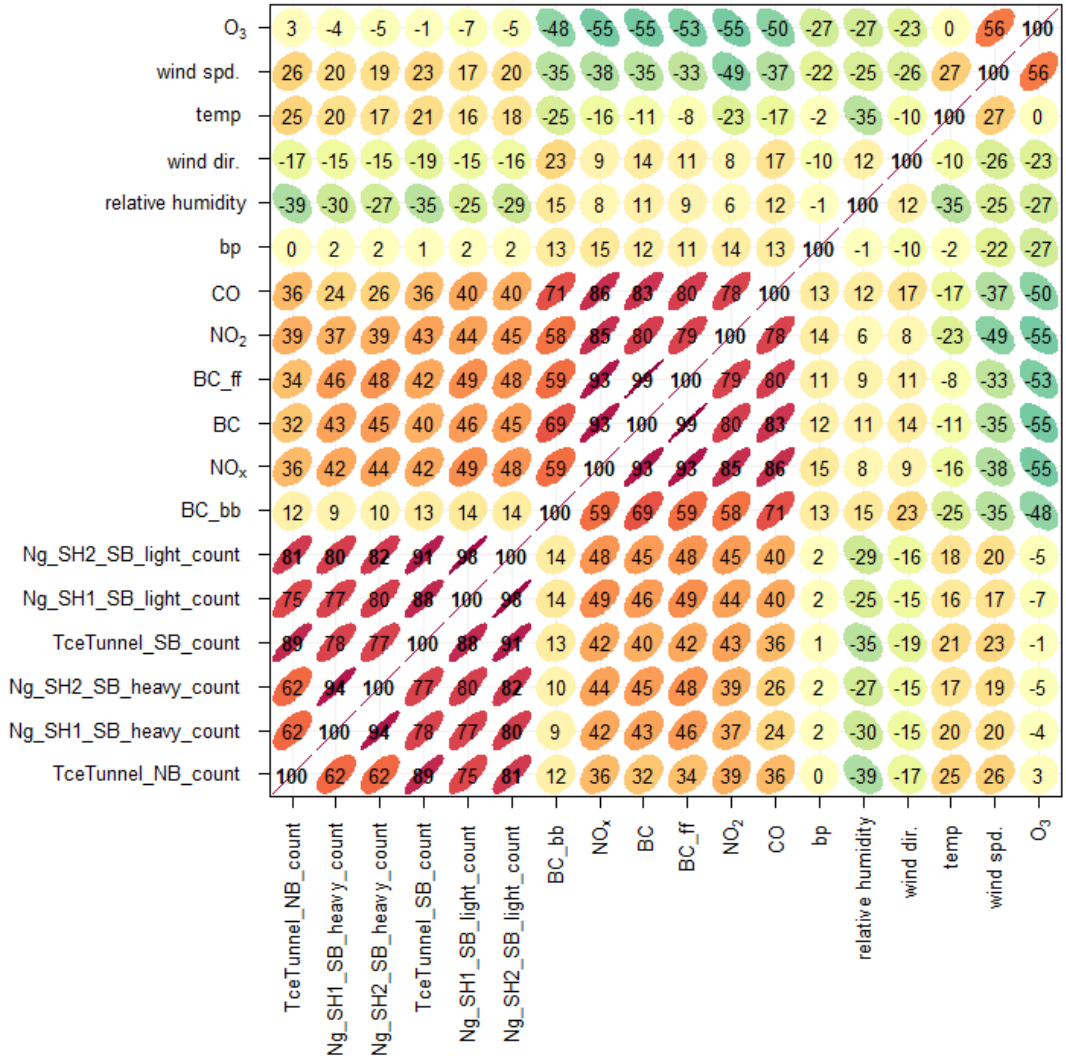
A8.1: Average observed and BAU-predicted pollutant levels for each Alert Level period and percentage change from BAU-predicted

Site & pollutant	Alert Level	Observations	BAU-predictions	Observations minus BAU-predictions	Percentage pollutant change from BAU-predicted	p-value (Student's t.test)
Wellington central						
BC – fossil fuel (µg/m^3)	L4	0.199	0.686	-0.49	-71.0	p < 0.001
	L3	0.679	0.854	-0.17	-20.5	p < 0.001
	L2	0.831	0.867	-0.04	-4.1	p < 0.001
	L1	0.903	0.876	0.03	3.0	ns
	L2/L3	0.619	0.639	-0.02	-3.2	ns
NO_2 (µg/m^3)	L4	5.3	12.7	-7.44	-58.5	p < 0.001
	L3	12.7	14.4	-1.75	-12.1	p < 0.001
	L2	15.33	15.38	-0.06	-0.4	p < 0.001
	L1	18.3	14.9	3.38	22.7	p < 0.001
	L2/L3	15.04	11.65	3.39	29.1	p < 0.001
NOx (ppb)	L4	4.3	14.8	-10.46	-70.7	p < 0.001
	L3	12.4	17.1	-4.69	-27.5	p < 0.001
	L2	14.45	17.70	-3.25	-18.4	p < 0.001
	L1	18.2	17.9	0.26	1.5	ns
	L2/L3	13.70	12.51	1.19	9.5	p < 0.001
CO (mg/m^3)	L4	0.06	0.22	-0.16	-73.2	p < 0.001
	L3	0.11	0.24	-0.13	-52.5	p < 0.001
	L2	0.18	0.24	-0.06	-26.1	p < 0.001
	L1	0.19	0.24	-0.05	-20.4	p < 0.001
	L2/L3	0.15	0.19	-0.04	-21.2	p < 0.001
O_3 (µg/m^3)	L4	40.88	36.66	4.22	11.5	p < 0.001
	L3	33.62	34.37	-0.75	-2.2	p < 0.001
	L2	34.80	33.40	1.40	4.2	p < 0.001
	L1	36.00	35.30	0.70	2.0	p < 0.001
	L2/L3	47.71	42.63	5.08	11.9	p < 0.001

Upper Hutt						
NO_2 (µg/m^3)	L4	3.1	7.1	-3.92	-55.5	p < 0.001
	L3	7.1	9.0	-1.83	-20.4	p < 0.001
	L2	11.7	12.5	-0.77	-6.2	p < 0.001
	L1	9.7	10.8	-1.10	-10.2	p < 0.001
	L2/L3	7.3	8.7	-1.47	-16.9	p < 0.001
NOx (ppb)	L4	2.5	7.8	-5.36	-68.5	p < 0.001
	L3	6.5	10.7	-4.23	-39.4	p < 0.001
	L2	11.0	15.2	-4.20	-27.6	p < 0.001
	L1	9.6	12.1	-2.50	-20.7	p < 0.001
	L2/L3	6.3	7.9	-1.62	-20.4	p < 0.001
CO (mg/m^3)	L4	0.117	0.138	-0.02	-15.4	p < 0.001
	L3	0.244	0.205	0.04	19.0	p < 0.001
	L2	0.469	0.302	0.17	55.5	p < 0.001
	L1	0.333	0.253	0.08	32.0	p < 0.001
	L2/L3	0.195	0.174	0.02	11.8	p < 0.001
Masterton West						
NO_2 (µg/m^3)	L4	3.5	5.3	-1.73	-34.0	p < 0.001
	L3	7.1	7.1	0.0	0	ns
	L2	8.9	8.8	0.14	1.1	ns
	L1	8.6	8.2	0.37	4.9	p =0.005
	L2/L3	5.5	5.9	-0.40	-6.8	p=0.0012
NOx (ppb)	L4	1.6	4.5	-2.9	-64.4	p < 0.001
	L3	5.2	6.4	-1.2	-18.8	p < 0.001
	L2	7.9	8.4	-0.5	-6.0	p=0.004
	L1	6.9	7.4	-0.5	-6.8	p = 0.03
	L2/L3	3.8	4.8	-1.02	-20.8	p < 0.001
CO (mg/m^3)	L4	0.156	0.185	-0.03	-15.7	p < 0.001
	L3	0.264	0.288	-0.02	-8.3	p = 0.0085
	L2	0.398	0.392	-0.01	1.5	ns
	L1	0.417	0.390	0.03	6.9	p=0.002
	L2/L3	0.232	0.281	-0.05	-17.4	p < 0.001

Appendix A9: Correlations between pollutants and vehicle counts

A9.1: Correlation plot 1-hour average pollutants at Wellington Central air monitoring site and Waka Kotahi traffic counts 2016 to 2019

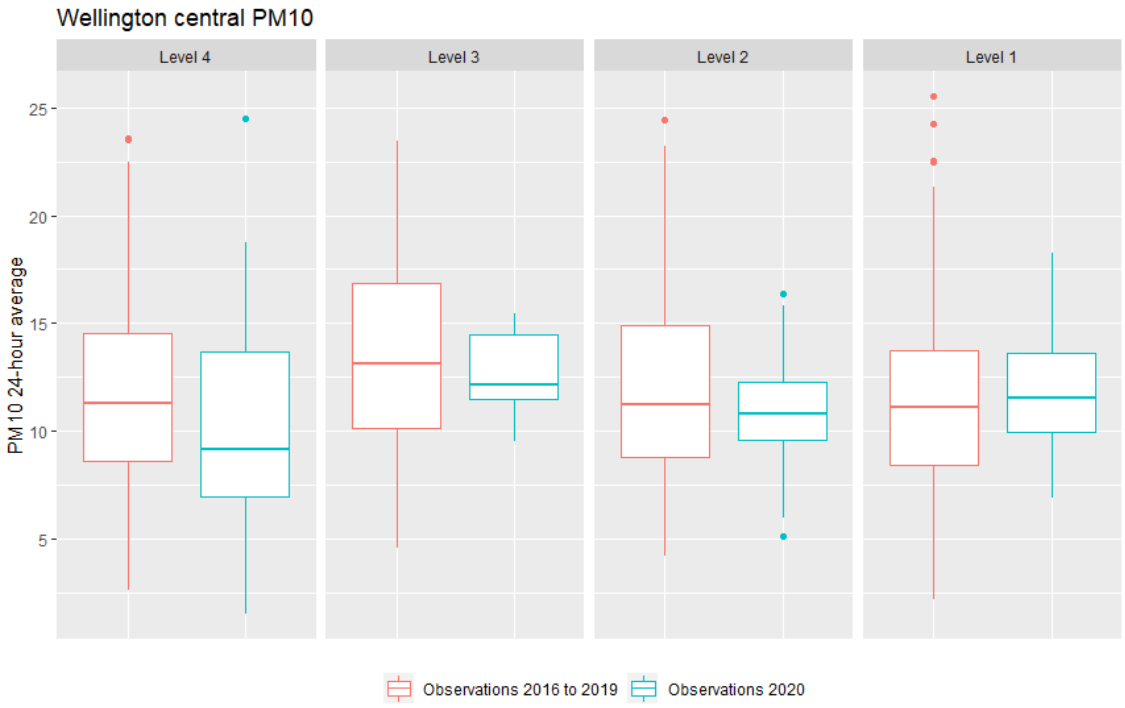


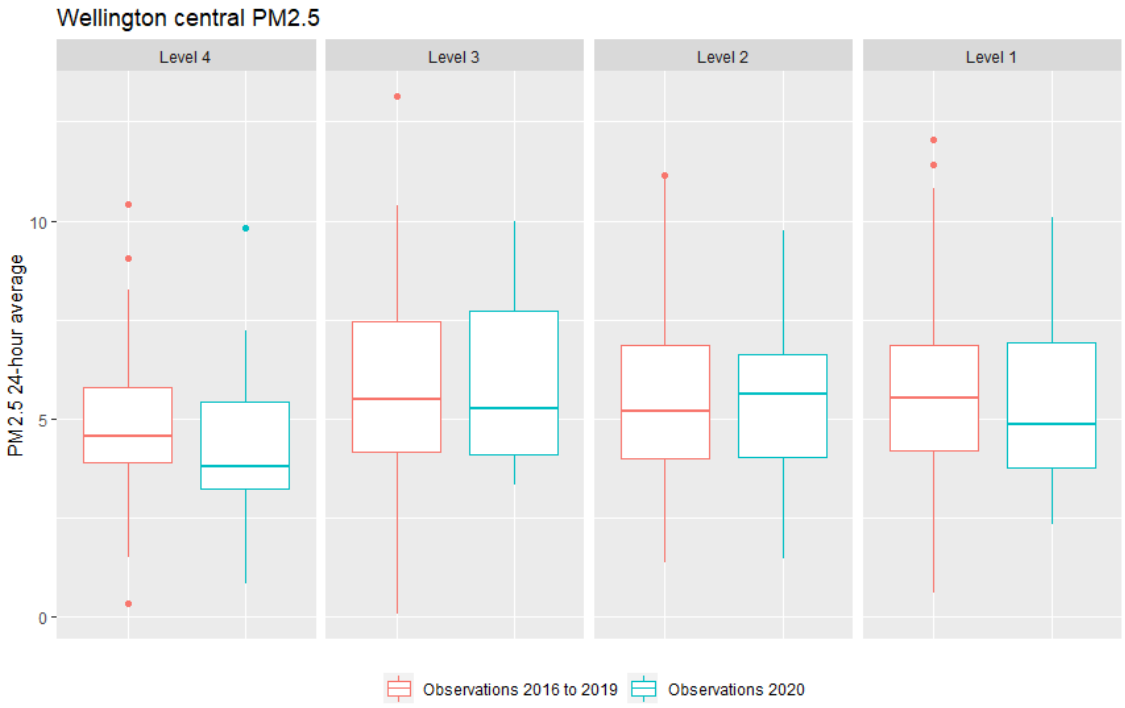
Appendix A10: Observed and long term average PM levels by Alert Levels

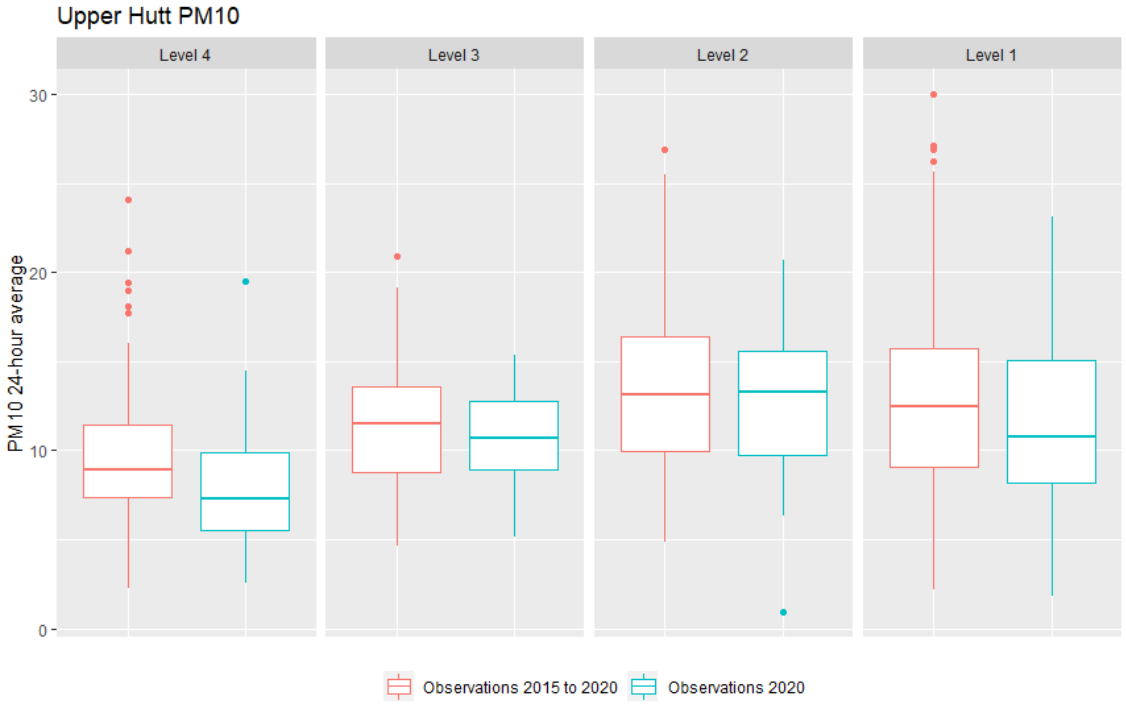
A10.1: Average observed and long term average pollutant levels for each Alert Level period (2020) and percentage change from long term average (2015 to 2019)

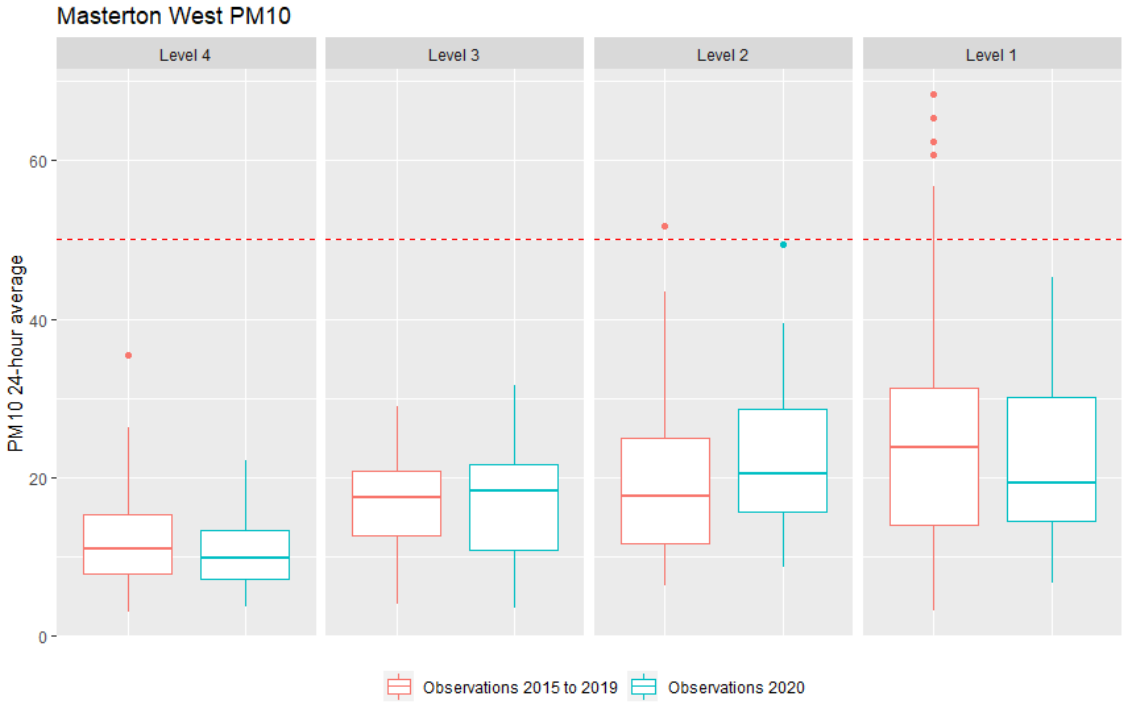
Site & pollutant	Alert Level	Observations	Long term average	Observations minus long term average	Percentage pollutant change from long term average	P-value (t-test)
Wellington central						
PM_{10} (µg/m^3)	L4	9.9	11.9	-2.0	-16.8%	p < 0.001
	L3	12.7	13.4	-0.7	-5.2%	p < 0.05
	L2	11.0	11.9	-0.9	-7.6%	p < 0.001
	L1	11.6	11.4	0.2	1.8%	ns
$PM_{2.5}$ (µg/m^3)	L4	4.1	4.8	-0.7	-14.6%	p < 0.001
	L3	5.8	5.6	0.2	2.6%	ns
	L2	5.5	5.6	-0.1	1.8%	ns
	L1	5.4	5.6	-0.2	-3.6%	ns
Upper Hutt						
PM_{10} (µg/m^3)	L4	8.1	9.5	-1.4	-14.7%	p < 0.001
	L3	9.4	11.3	-1.9	-16.8%	p < 0.05
	L2	13.0	13.7	-0.7	-5.1%	ns
	L1	11.2	12.8	-1.6	-12.5%	p < 0.001
Masterton West						
PM_{10} (µg/m^3)	L4	10.8	12.0	-1.2	-10%	p < 0.05
	L3	16.6	16.8	-0.2	-1.2%	ns
	L2	22.4	20.0	2.4	12.0%	p < 0.05
	L1	22.6	24.3	-1.7	-7.0%	ns
$PM_{2.5}$ (µg/m^3)	L4	8.2	8.1	0.1	1.2%	ns
	L3	14.5	12.2	2.3	15.9%	p < 0.05
	L2	20.9	16.1	4.8	29.8%	p < 0.05
	L1	20.1	20.3	-0.2	-1.0%	ns

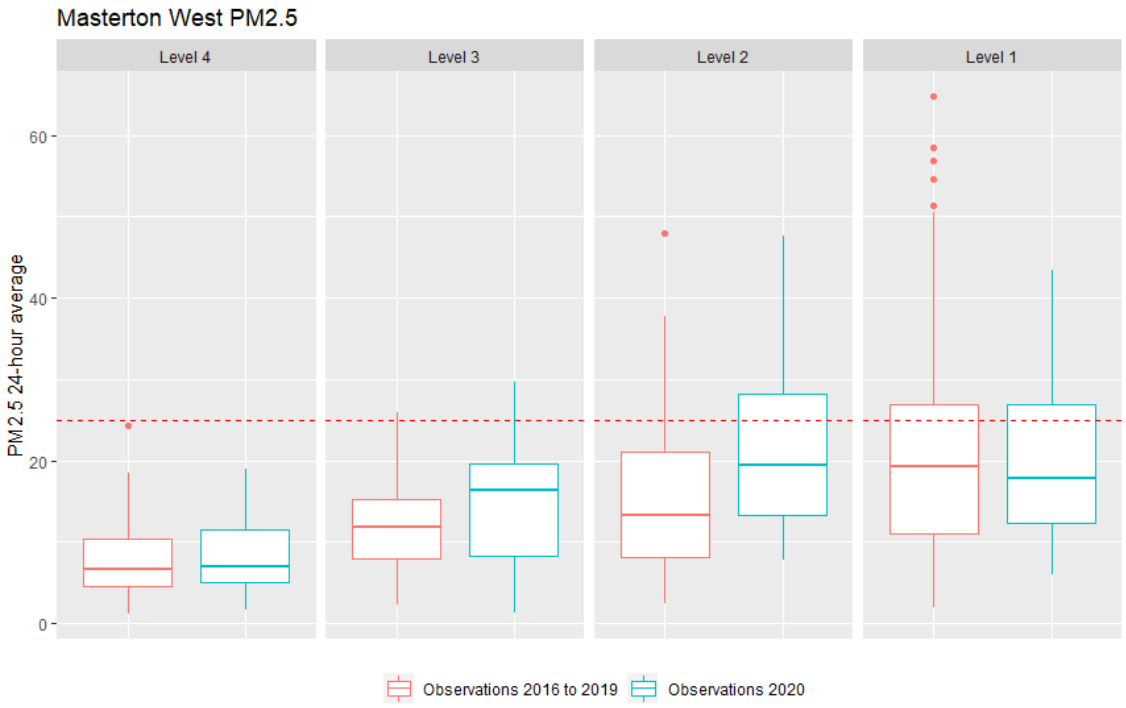
Appendix A11: PM box plots





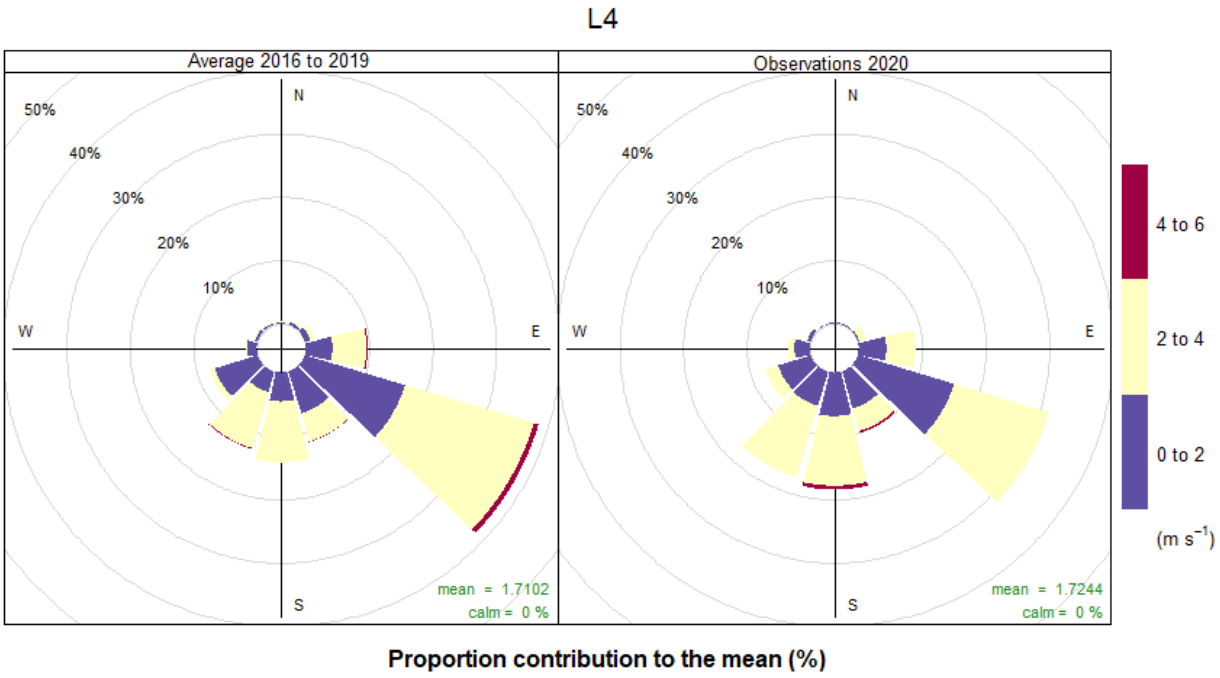

Upper Hutt PM10
Level 4
Level 3
Level 2
Level 1
PM10 24-hour average
30
20
10
0
Observations 2015 to 2020
Observations 2020

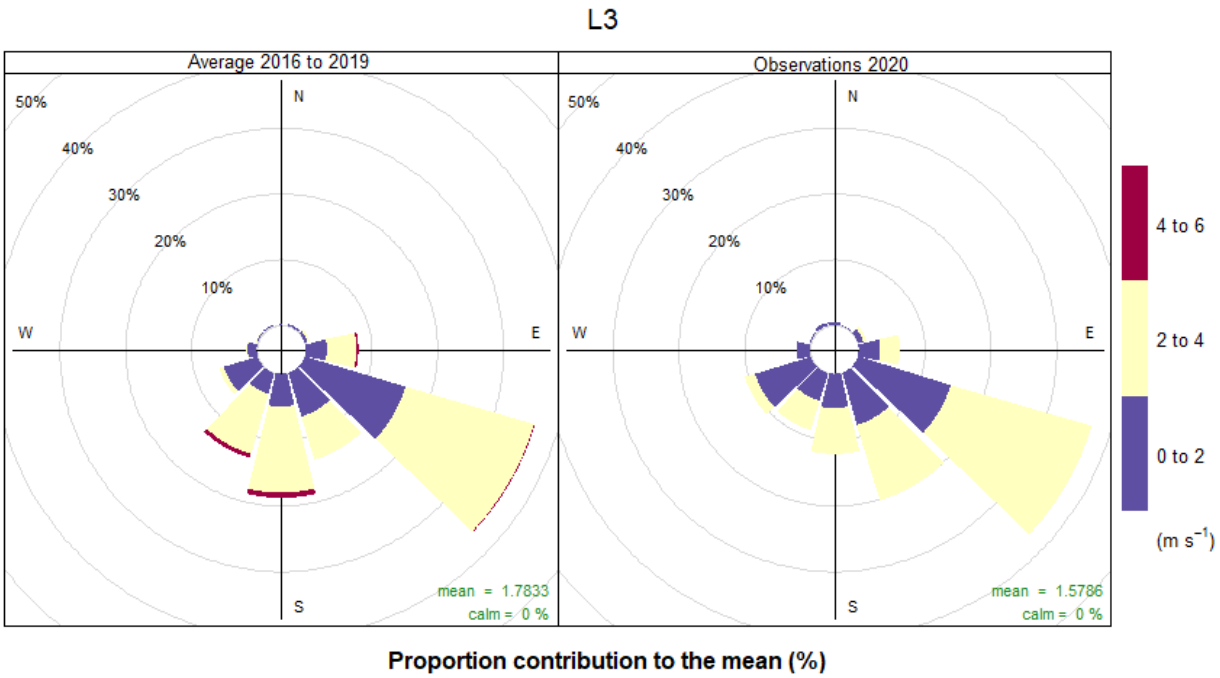

Masterton West PM10
Level 4
Level 3
Level 2
Level 1
PM10 24-hour average
60
40
20
0
Observations 2015 to 2019
Observations 2020


Masterton West PM2.5
Level 4
Level 3
Level 2
Level 1
PM2.5 24-hour average
60
40
20
0
Observations 2016 to 2019
Observations 2020

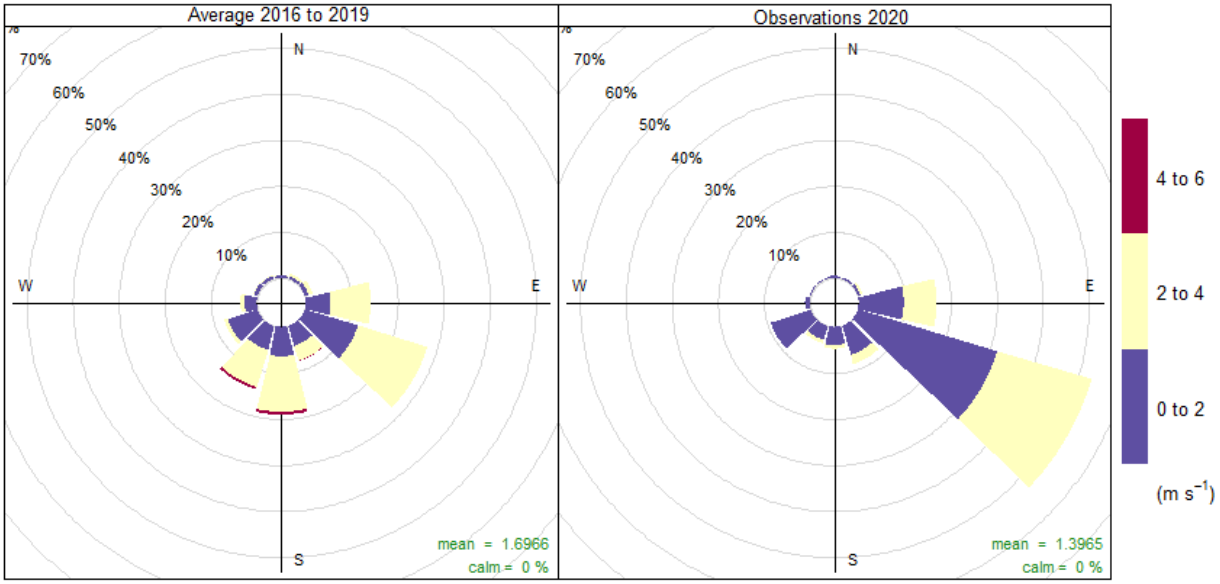
Appendix A.12: Wind roses by alert level period

A.12.1: Wellington central



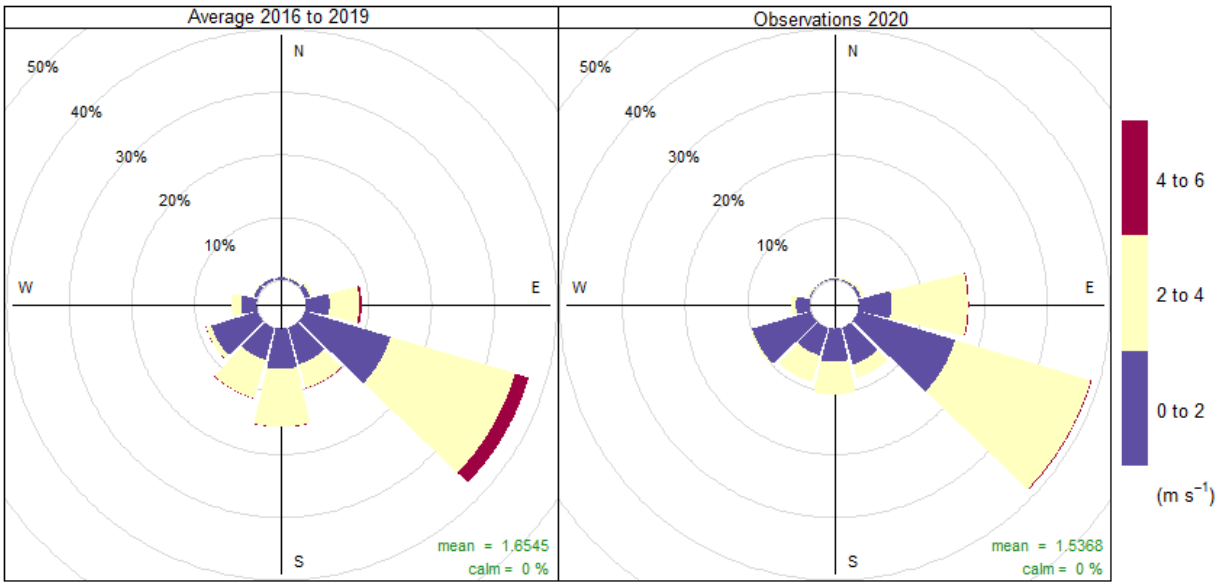


L2



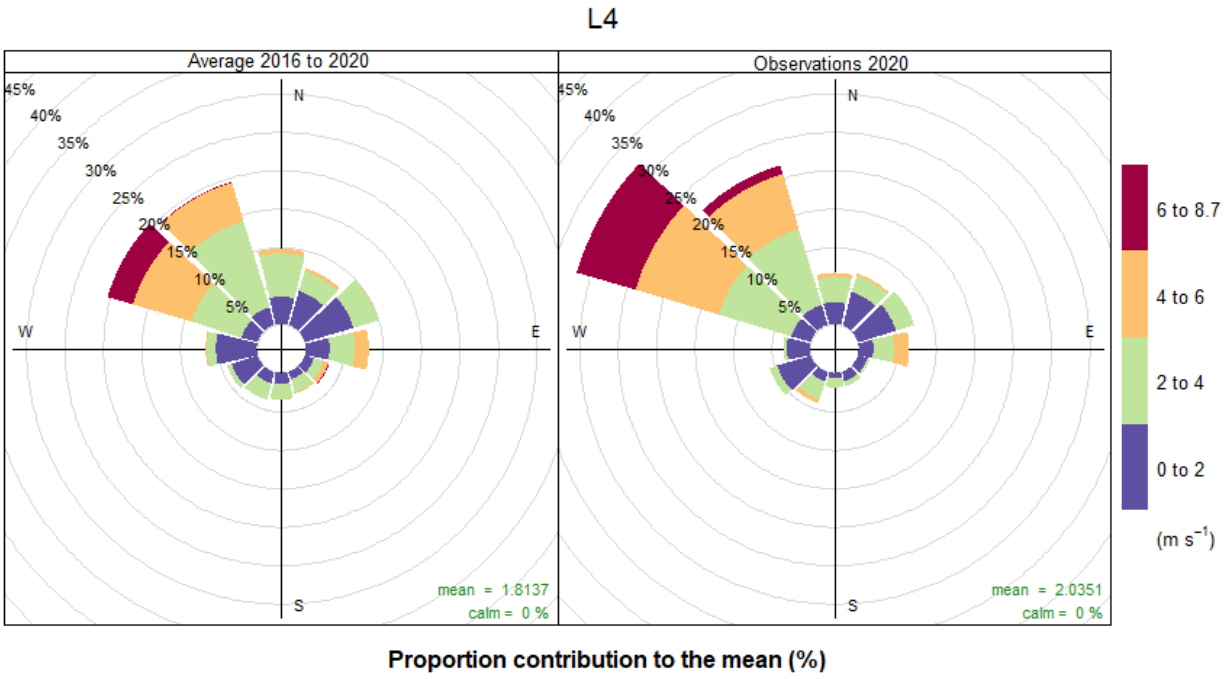
Proportion contribution to the mean (%)

L1

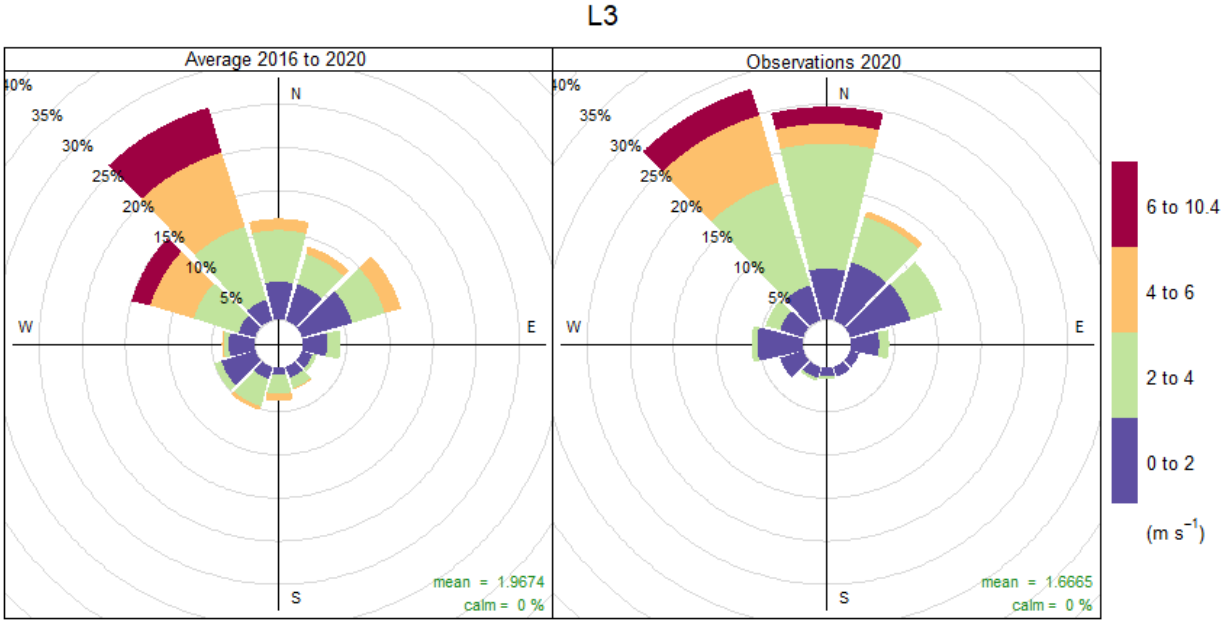


Proportion contribution to the mean (%)

A12.2: Upper Hutt

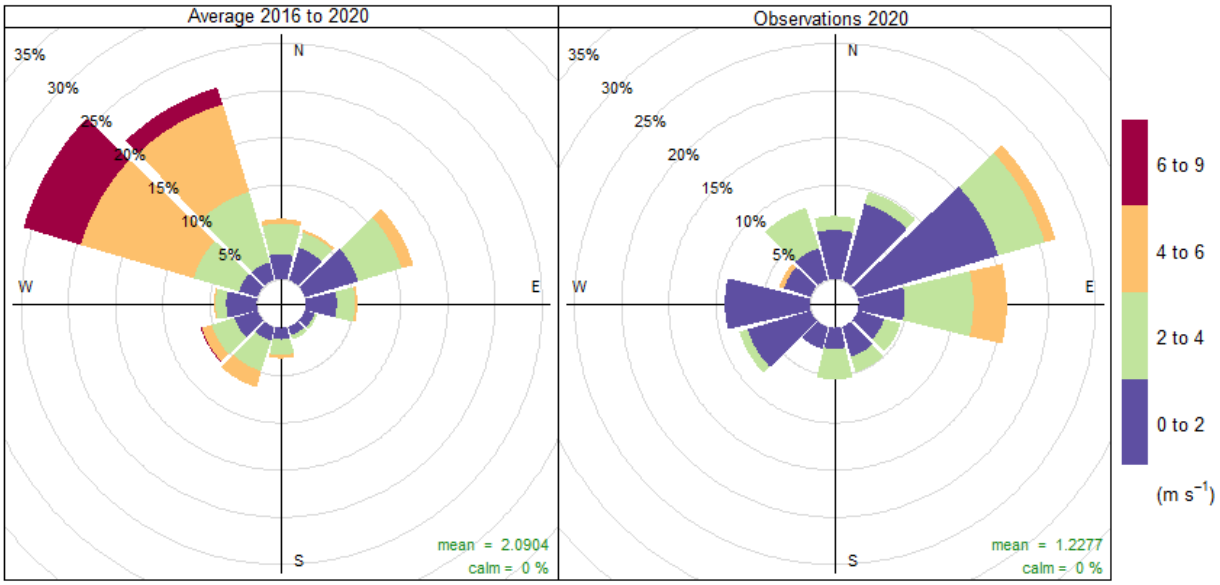


Proportion contribution to the mean (%)



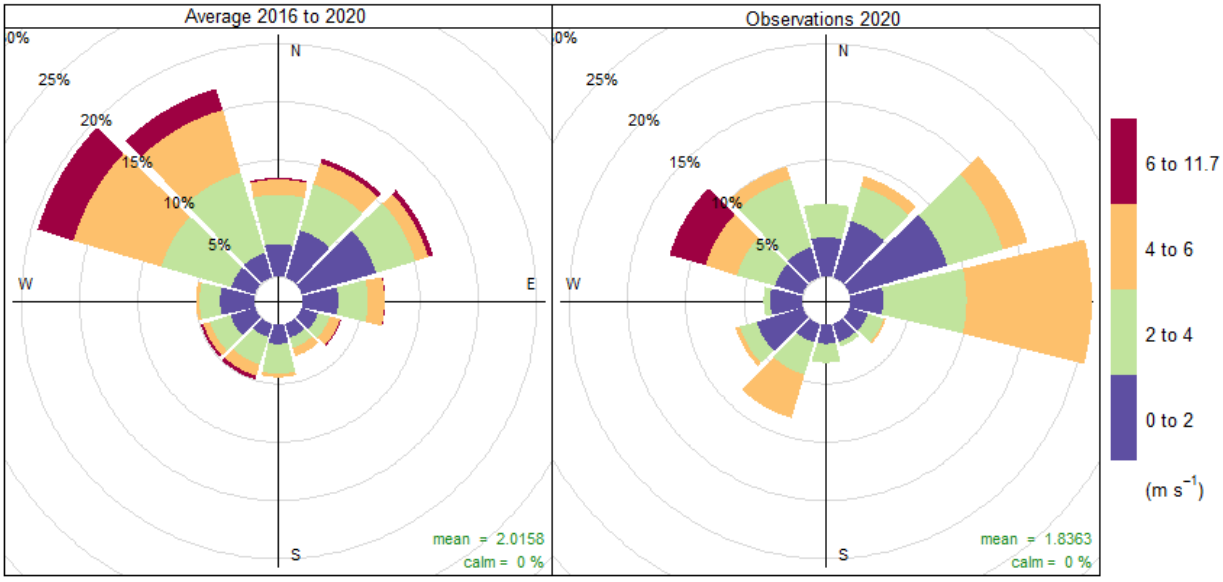
Proportion contribution to the mean (%)

L2



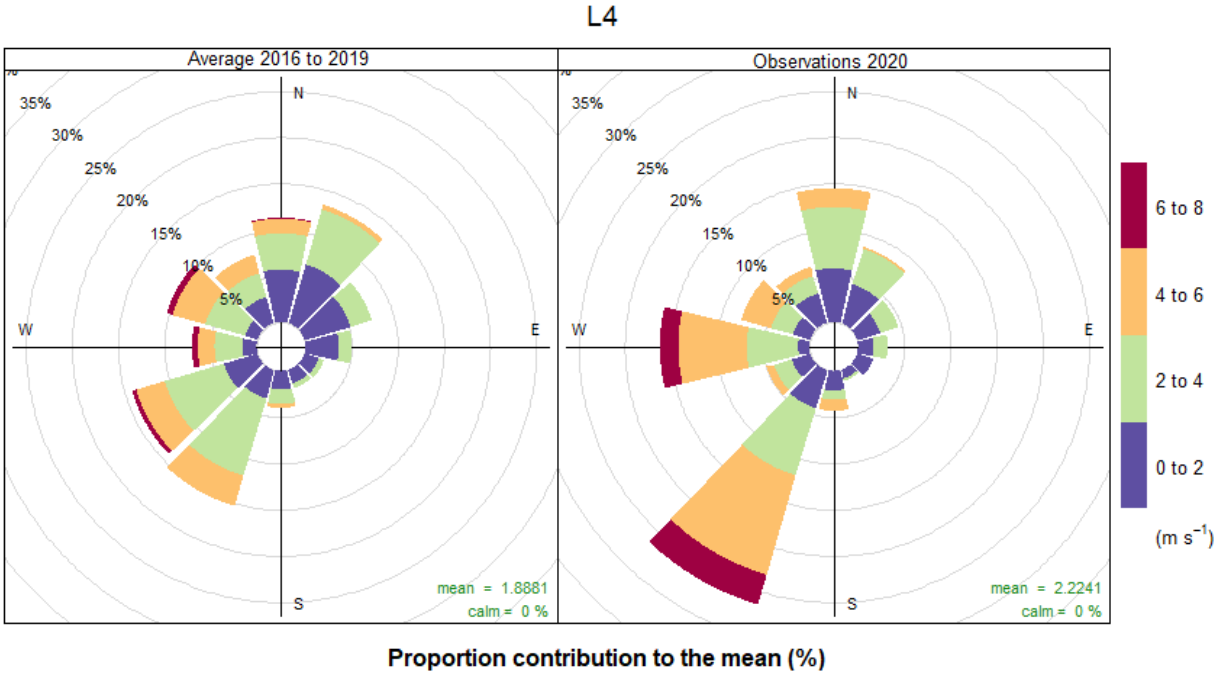
Proportion contribution to the mean (%)

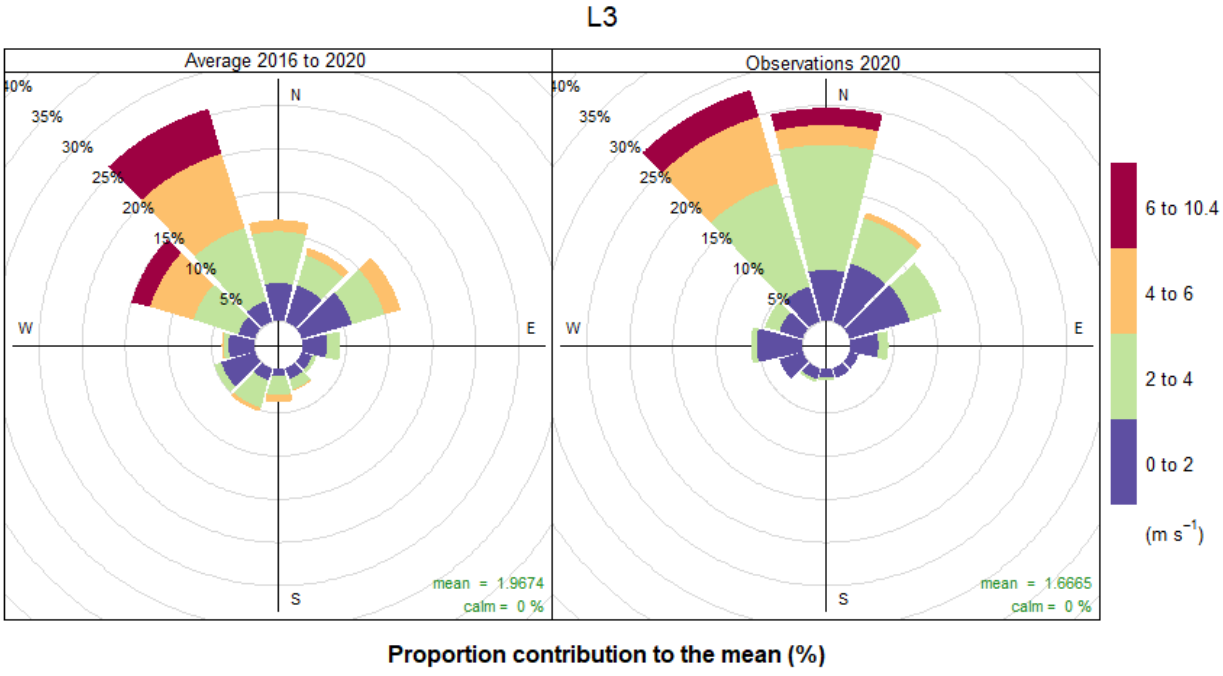
L1



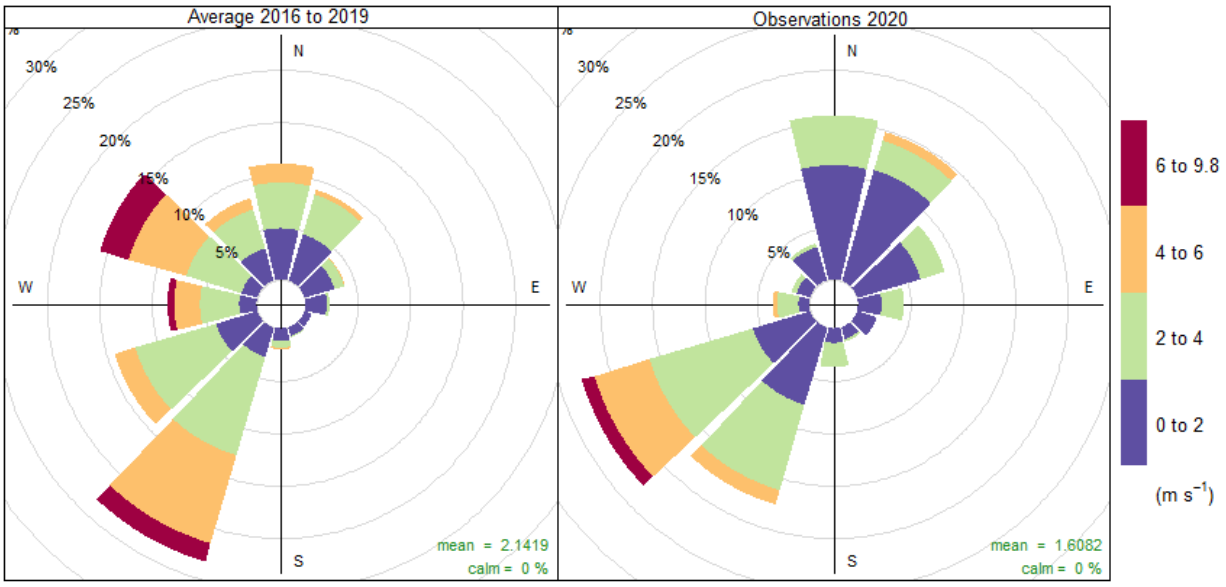
Proportion contribution to the mean (%)

A12.3: Masterton West



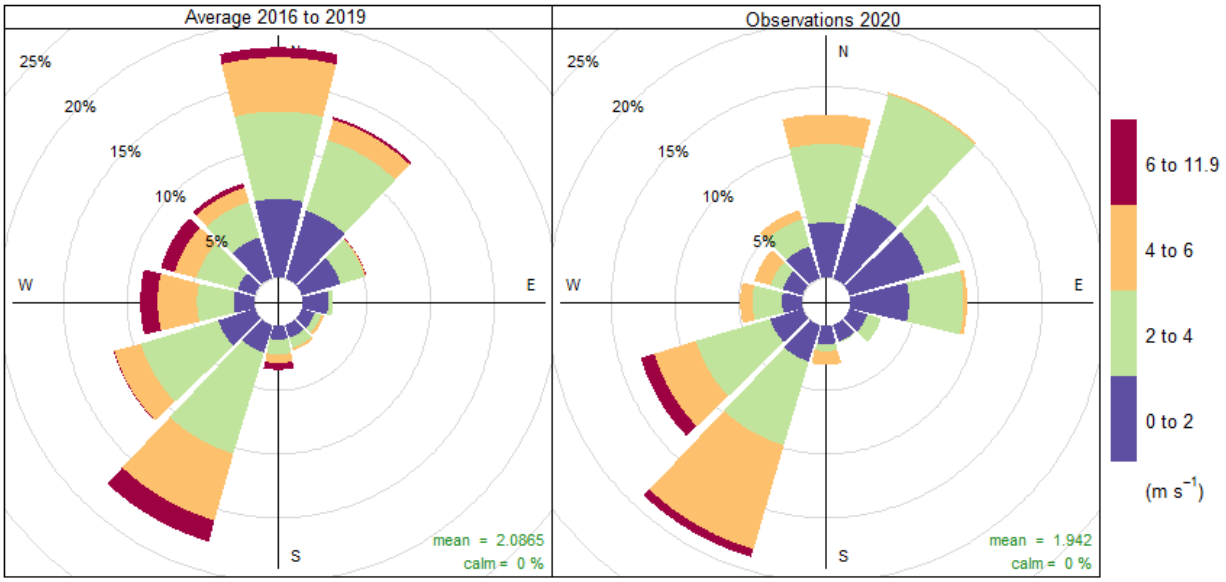


L2



Proportion contribution to the mean (%)

L1



Proportion contribution to the mean (%)

Appendix A.13: Meteorology: wind speed and temperature averages

A13.1: Wellington central 1-hour average

Period	Parameter	Observations 2020	Observations 2016 to 2019
L4	Air temperature	14.3**	14.8
	Wind speed	1.7	1.7
	Wind direction	168.5	162.4
L3	Air temperature	13.8**	14.2
	Wind speed	1.6**	1.8
	Wind direction	168.4	162.0
L2	Air temperature	11.4	11.6
	Wind speed	1.4**	1.7
	Wind direction	152.2	169.2
L1	Air temperature	10.4	10.5
	Wind speed	1.5**	1.7
	Wind direction	165.2	169.3

A13.2: Upper Hutt 1-hour average

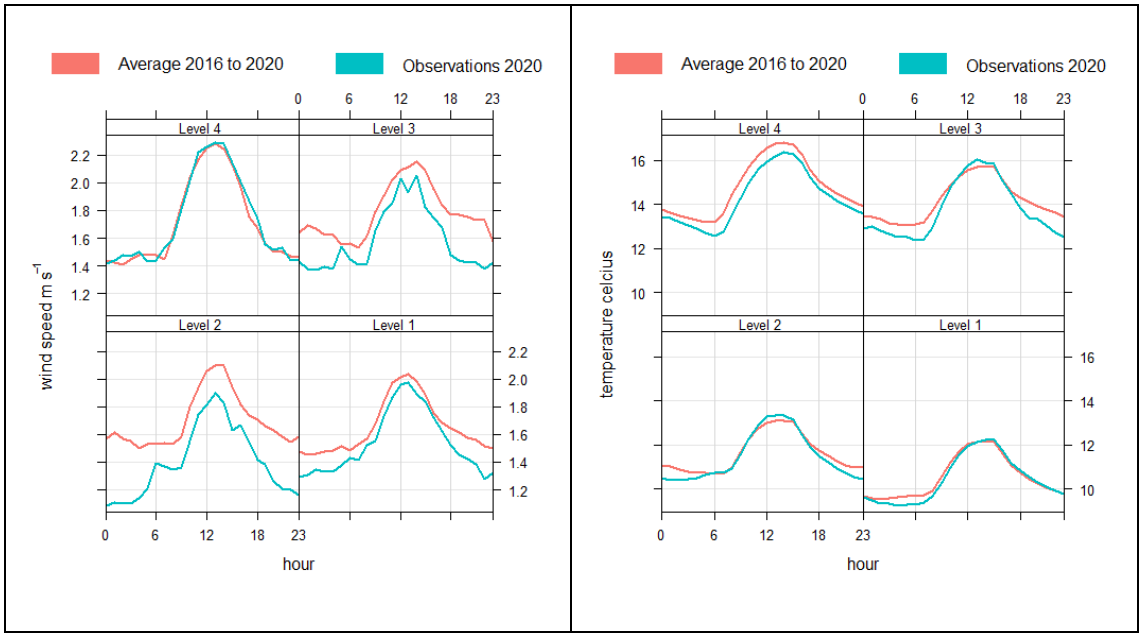
Period	Parameter	Observations 2020	Observations 2015 to 2019
L4	Air temperature	14.1**	15.0
	Wind speed	2.0*	1.9
	Wind direction	199.6	183.5
L3	Air temperature	13.2**	14.3
	Wind speed	1.7**	2.1
	Wind direction	173.8	189.7
L2	Air temperature	10.3**	11.7
	Wind speed	1.2**	1.5
	Wind direction	156.4	183.2
L1	Air temperature	9.9	10.0
	Wind speed	1.8**	2.0
	Wind direction	157.3	179.4

A13.3: Masterton West 1-hour average

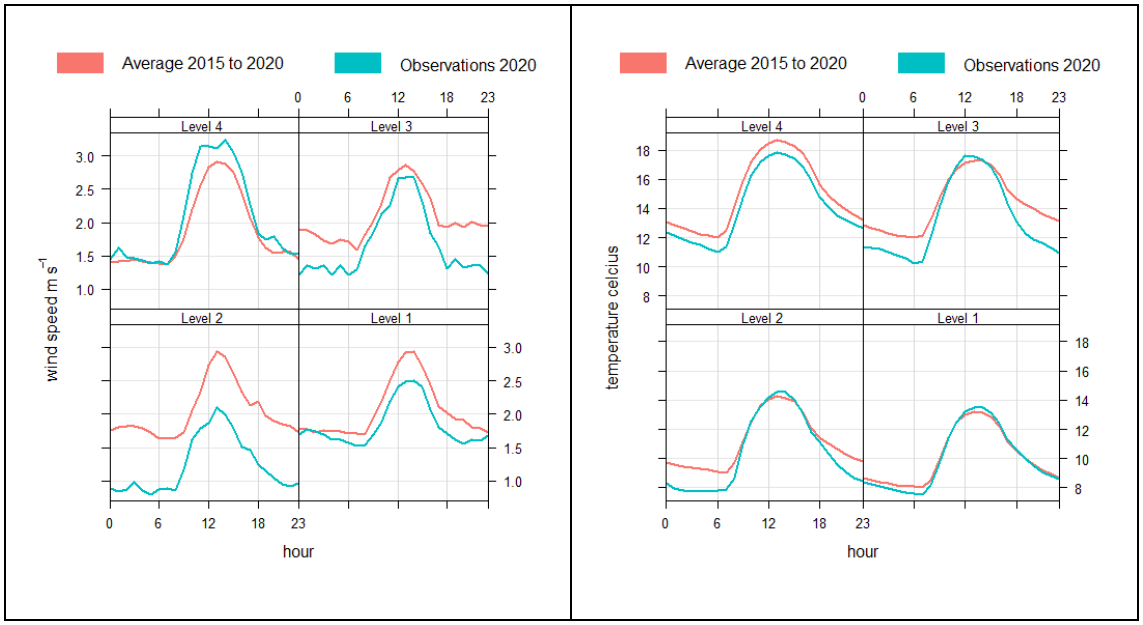
Period	Parameter	Observations 2020	Observations 2015 to 2019
L4	Air temperature	13.5**	14.1
	Wind speed	2.2**	2.0
	Wind direction	188.3	163.8
L3	Air temperature	11.9*	12.7
	Wind speed	2.0	2.1
	Wind direction	157.1	163.4
L2	Air temperature	9.2*	9.6
	Wind speed	1.6**	2.2
	Wind direction	160.9	184.9
L1	Air temperature	8.8	8.9
	Wind speed	1.9**	2.2
	Wind direction	155.3	176.2

Appendix A14: Meteorology: wind speed and temperature diurnals

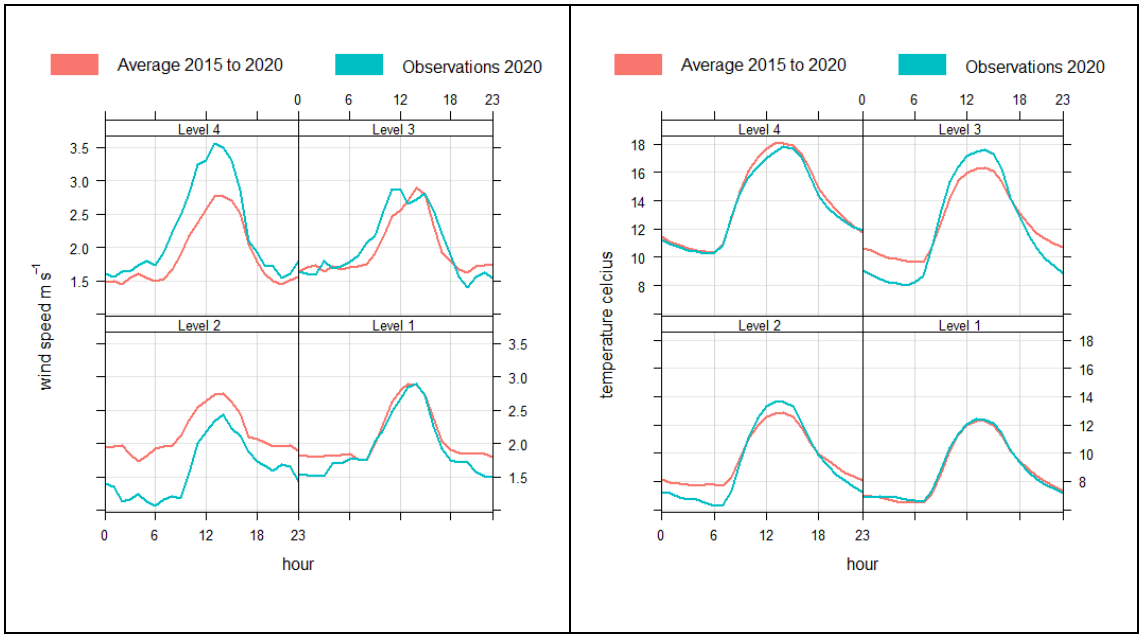
A14.1: Wellington central wind speed (left) and temperature (right)



A14.2: Upper Hutt wind speed (left) and temperature (right)



A14.3: Masterton West wind speed (left) and temperature (right)



Appendix 15: Correlation between 2019 and 2020 passive NO_2 monitoring results

A15.1: NO_2 passive diffusion tube sites. Linear regression of 2019 March-April concentration on reduction in concentration observed in 2020 March-April