The
Air Quality Collective

Quantifying the impact of outdoor burning on air quality in the Alexandra-Clyde basin

Prepared for Otago Regional Council

March 2026

26 February 2026 12.31 pm

Quantifying the impact of outdoor burning on air quality in the Alexandra-Clyde basin

Prepared for Otago Regional Council

Prepared by:
Ian Longley

Report No: 2025_08

Revision History		
Version number	Description	Date
0.1	Draft in preparation/in review	10th October 2025
1.0	Final version for client	24th February 2026

For any information about this proposal please contact:
Ian Longley

Phone: +64-27 657 1389
Ian.longley@theairqualitycollective.org

The Air Quality Collective Ltd
PO Box 2023
Stoke,
Nelson 7041

Contents

Executive Summary 5

1. Introduction 7

1.1 Background 7

1.2 Scope of the project 8

2 Methods 9

2.1 Study design 9

2.2 Instruments 10

2.2.1 Ground-based monitoring 10

2.2.2 Cameras 10

2.3 Siting 10

2.4 Time periods and Stage reviews 11

2.5 Data processing methods 11

2.5.1 Instrument inter-comparability 11

2.5.2 Data adjustment 11

2.6 Survey 14

3 Results 15

3.1 Insight into processes controlling air quality in the basin 15

3.1.1 Emissions 15

3.1.2 Formation/removal: Condensation and evaporation 15

3.1.3 Vertical dispersion: Inversions 15

3.1.4 Lateral dispersion 17

3.2 Visible fires (image analysis) 18

3.2.1 Suitability of the method for identifying fires 18

3.2.2 Subjective quantitative analysis 18

3.2.3 Temporal variation 18

3.2.4 Diurnal variation 19

3.2.5 Fire source locations 19

3.2.6 Plume dynamics 19

3.3 Air quality (all sources) in the basin 19

3.3.1 Long-term mean spatial variation in PM 19

3.3.2 Relationship between air quality in Alexandra and across the basin 21

3.3.3 Compliance with PM_{10} standard 22

3.3.4 Discussion 23

3.4 Contribution of non-combustion sources to air quality in the basin 23
3.5 Contribution of outdoor burning to air quality in the basin 24
    3.5.1 Discussion 24
    3.5.2 Impact of post-inversion fires on basin-wide air quality 25
    3.5.3 Impact of “events” on localised air quality 25
3.6 Survey results 30

4 Discussion 32

4.1 Limitations to analysis 32
    4.1.1 Cloud and fog 32
    4.1.2 Boundary layer depth (inversion intensity) not measured 32
    4.1.3 Persistence (age of aerosol) unknown 32
    4.1.4 Impacts outside the basin.. 32
4.2 Opportunities for extended analysis... 32
    4.2.1 Computer vision and image analysis.. 32
    4.2.2 Perception analysis 33

5 Conclusions 34

6 Policy implications and recommendations for further work 36

7 Acknowledgements.. 37

8 References.. 37

Executive Summary

Otago Regional Council (ORC) is responsible for the management of air quality across the Otago region. Emissions from home heating are the overwhelmingly dominant cause of poor air quality in most towns outside Dunedin. Despite this, many Otago towns and their immediate surroundings, are also impacted by emissions from outdoor burning. This arises from a combination of agricultural (especially horticultural) practices and refuse burning on residential property. Some of these sources may be permitted or consented and some may not.

Emission estimates for outdoor burning used in inventories are highly uncertain. There have also been very few studies that have attempted to observationally verify estimates, or otherwise observe the impact of these emissions on local air quality, and none in Otago. This lack of data inhibits the development and public acceptability of policy actions.

This study was commissioned to fill some of these knowledge and evidence gaps and to help ORC improve its understanding of outdoor burning emissions, particularly in terms of evidence that would inform the definition of airshed boundaries and the suitability of different policy approaches.

This is an under-studied topic and there are no established methods. It was understood that this study – which focussed on the Alexandra-Clyde basin - would therefore be somewhat experimental in nature.

A grid of 17 air monitors were spread across the basin floor whilst 4 cameras positioned on the encircling hills recorded images from the basin. Observations ran from 10th April to 17th September 2025.

We found that air quality was relatively consistent across the basin with average particle levels being 73 % of those measured by ORC in Alexandra. Although most of the basin met the current air quality standards, we estimate that new standards proposed by MfE would have been exceeded several times across the whole basin.

Video images showed outdoor burning to be occurring in the Alexandra-Clyde basin almost every day during the study, and most often between 11 am and 3 pm.

The combination of video images and ground-level monitoring revealed that the impact of smoke from these fires on air quality was highly sensitive to the presence or absence of atmospheric inversions in the basin. An inversion is a natural phenomenon by which vertical dispersion is suppressed leading to smoke emitted below the inversion to be trapped below it, leading to much higher ground-level concentrations than would otherwise be the case.

The Alexandra-Clyde basin is especially prone to the formation of inversions. We inferred the presence of inversions on 63 days out of 146.

Overall, we find that smoke from the majority of outdoor burns was well dispersed and makes a minor or negligible impact on air quality measured at ground-level in the basin, due to its buoyancy and effective vertical dispersion. However, the impact on visibility may still be significant.

Conversely, smoke emitted during inversion conditions can become trapped and have a significant impact across the basin. Emissions in the afternoon are likely to have a greater impact due to the trapping and accumulation of the smoke through the night until the following morning. On a few days per year inversions appear to persist all day. On these occasions smoke form outdoor burning may have a very significant impact on local air quality.

Therefore, we find that whereas domestic home heating is the dominant contributor to air pollution and the risk of exceedance of standards, for a few days per year outdoor burning makes a significant contribution to that risk.

1. Introduction

1.1 Background

Otago Regional Council (ORC) is responsible for the management of air quality across the Otago region. ORC have commissioned the creation of emission inventories for several Otago towns. These inventories indicate that emissions from home heating are the overwhelmingly dominant source in most towns outside Dunedin. Analysis of air quality monitoring data also appears to corroborate this conclusion as poor air quality is mostly limited to cold winter evenings with low winds. Despite this, Otago towns - and especially their immediate surroundings - are also undoubtedly impacted by emissions from outdoor burning. This arises from a combination of agricultural (especially horticultural) practices and refuse burning on residential property. Some of these sources may be permitted or consented and some may not.

Emission estimates for outdoor burning used in inventories are highly uncertain. There have also been very few studies that have attempted to observationally verify estimates, or otherwise observe the impact of these emissions on local air quality, and none in Otago. This lack of data inhibits the development and public acceptability of policy actions.

Outdoor burning emissions tend to be odorous and highly visible, whereas home heating emissions create much smaller plumes mainly at times when it is dark, and most people are indoors. This may create a perception bias where local people may think that outdoor burning is more significant than it is, and home heating less significant. This can lead to the perception that it is regulators and policymaker who are biased. This view may be reinforced by the relative lack of scientific attention this issue receives relative to home heating emissions. These perceptions may present a barrier to support for policy measures to manage air quality in the region.

Uncertainty in the location, timing and intensity of outdoor burning sources, as well as the spatial extent of their impact, presents a dilemma for Otago Regional Council in terms of defining the boundaries of gazetted airsheds (how far boundaries extend from the urban edge) and what rules and policies should apply in which Air Zones.

This study was commissioned to attempt to fill some of these knowledge and evidence gaps and to help ORC improve its understanding of outdoor burning emissions, particularly in terms of evidence that would inform the definition of airshed boundaries and the suitability of different policy approaches. Specific questions that were addressed included:

- Where outdoor emissions occur, how often, when and for how long?
- How does outdoor burning smoke travel and disperse?
- What is the contribution of outdoor burning to local air quality and how does it vary in time and space? How does this compare to other sources, especially domestic home heating?

This is an under-studied topic and there are no established methods. It was understood that this study would therefore be somewhat experimental in nature.

The study was to focus on the Alexandra-Clyde basin, whilst testing methods that could be applied elsewhere.

1.2 Scope of the project

This project did not quantify the impact of emissions from natural, landscape or accidental fires.

Although we believe computer vision to be a promising method, due to its experimental and uncertain nature, and resource constraints, it was not pursued within this project. However, video images were captured which could support an independent analysis outside of this project.

2 Methods

2.1 Study design

The objective of monitoring and understanding outdoor burning sites and activity can, in principle, be met through self-reporting by those conducting the burns, through direct surveillance (e.g. mobile "spotter" teams, or using fixed cameras), or indirectly using fixed "smoke sensor" data supported by geospatial interpolation or inverse dispersion modelling. We take the view that compliance with, and accuracy of self-reporting is likely to be low, and mobile spotter teams are very labour intensive. Both methods are likely to yield mostly qualitative data and hence high uncertainties. Cameras and smoke sensors, however, are far more reliable and scalable. Smoke sensors can yield quantitative data, and camera images may provide quantitative data through computer vision analysis.

The objective of understanding the downwind impact of burning can, in principle, also be met through either a modelling or an observational approach, or a combination of the two. Modelling consists of two elements: emissions and dispersion. Emission modelling would be dependent on information that is largely (currently) unavailable (and are the amongst the goals of this project), such as the intensity of emissions, their timing, duration and location. Furthermore, dispersion modelling is also subject to significant uncertainty when local topographic features are present, such as in valleys and basins, or around complex variation in vegetation. For these reasons we believe that a model-only approach would present excessive uncertainty and was not pursued in this study.

A sensor/camera-based observational method was preferred, although resource constraints translated into a limit on spatial coverage/detail. Point-based air quality sensor monitoring was required in order to provide quantification that is comparable to the way air quality is monitored and evaluated generally. Given the uncertainty in modelling, resources were devoted to maximising the observational data that could be collected. This not only reduced uncertainty, but also provided the best resource to investigate the applicability of dispersion modelling in a future piece of research.

The recent rapid development of computer vision may provide an alternative method for monitoring outdoor burning. This method would involve training a machine-learning model to analyse video images to identify both emission sources and impacted downwind areas. However, the success of such a system is most likely going to rely on independent ground-based validation. Furthermore, the sensitivity of results to issues such as light levels, camera resolution and sensitivity, single versus multiple camera coverage, etc are unknown for this application at the time of writing. Given that computer vision has not (to our knowledge) previously been used in this way, and certainly not in New Zealand, we consider it to be quite uncertain and risky. However, we also think the potential is significant and the challenges are surmountable.

A consequence of the lack of existing data and large uncertainties around outdoor burning is the risk of selecting a sub-optimal study area and monitoring sites (i.e. missing the area most impacted), and sub-optimal balance between methods. We addressed this by conducting the study in at four stages. Stage one was a screening stage covering a wider spatial area with the goal of identifying the most impacted locations.

On three occasions (after one, two and four months) the data was reviewed and a decision taken to maintain or reduce the study area, thus forming Stages Two, Three and Four. Reducing the study area means monitoring resources can be concentrated where there is most value, for example due to stronger burning activity or more frequently impacted, better visual line-of-sight, or more instrument deployment options.

Outdoor burning emissions can be quite short-lived lasting minutes rather than hours. Similarly, any given downwind site might be impacted fleetingly with concentrations changing rapidly. For this reason, we selected fast-responding instrumentation that can capture these rapid changes.

2.2 Instruments

2.2.1 Ground-based monitoring

Fixed point monitoring was conducted using Clarity Node-S monitors.

The Clarity monitors report PM_1, $PM_{2.5}$, PM_{10}, temperature and humidity every 17 minutes, sending data to the cloud via the mobile phone network. We have successfully used these devices for ORC in Alexandra before. They are very easy to deploy and were chosen primarily because of their reliability.

2.2.2 Cameras

Due to the experimental nature of the project, 4 different camera models were trialled, specifically:

- ReoLink GO PT
- ReoLink GO PT Plus
- ReoLink GO PT Ultra
- ReoLink Argus Eco

All cameras recorded still images which were sent to a cloud server. Each day the images were compiled into a time-lapse video.

2.3 Siting

17 Clarity units were made available for the project. For Stage One 16 were deploy in a regular rectangular grid cross the basin (Figure 1), with sites designated a row-and-column ID. One additional unit co-located with the ORC regulatory air monitoring station on Ventry Street in Alexandra.

Our original intention was to place the cameras to the north, west , south and east sides of the basin to facilitate triangulation of fire position. The Alexandra Basin is characterised by extensive use of tall trees for wind breaks. These create visual blockages and require cameras for our purposes to be at least 40 m above the basin floor. That largely limited suitable sites to the hillsides surrounding the basin. Furthermore, logistically practical sites (accessible by vehicle, open views and where permission could be gained) were limited

Figure 1 indicates the locations chosen.

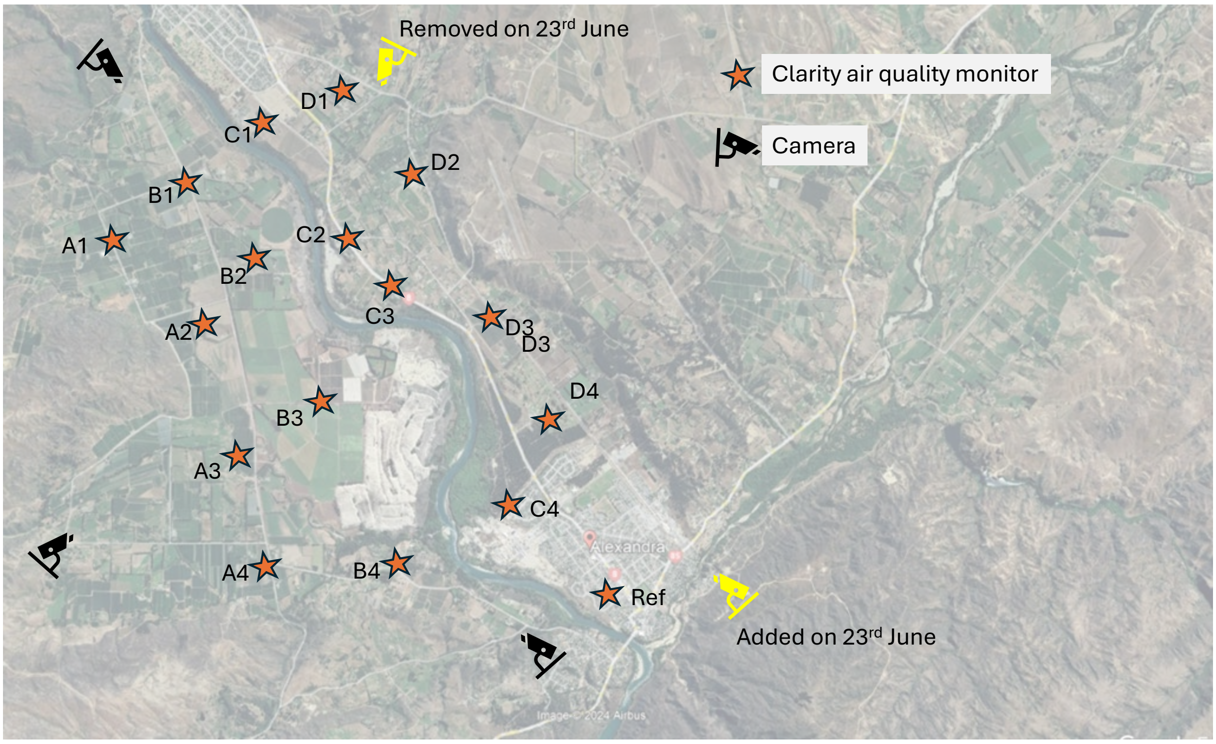

Figure 1: Instrument locations.

2.4 Time periods and Stage reviews

The Clarity monitors were installed in two stages on 20th March and 10th April 2025.

All monitors were removed on 17th September 2025.

At each Stage Review we found that smoke was being detected by every monitor. Consequently, the monitoring locations were kept unchanged throughout the study. On the other hand, we found that the cameras to the north side of the basin were detecting smoke far less frequently than those to the south. Consequently, the camera initially installed with a view over Springvale was moved to a site close to the Alexandra clock on 23rd June (Figure 1).

2.5 Data processing methods

2.5.1 Instrument inter-comparability

2.5.2 Data adjustment

The Clarity monitor is not an approved method for regulatory purposes. It is based on a different sensor technology (optical scattering) from the approved methods. Otago

Regional Council operate a Beta Attenuation Monitor at their Ventry Street site in Alexandra. This is a high-quality instrument that reports PM_{10}. To allow for our data and ORC data to be compared and combined, one Clarity monitor was placed at the ORC site at Ventry Street for the entirety of the observational campaign.

ORC provided with the data from their Alexandra site for the campaign period. The instrument at Alexandra was temporarily replaced from 1st July 2025 leading to a step-change in the concentrations being recorded. We therefore base the adjustment equation on data collected before 1st July, i.e. using the data from the instrument which has been used to conduct regulatory compliance monitoring for many years.

Figure 2 shows the relationship between hourly PM_{10} measured by the Clarity and the ORC monitor (BAM). The relationship is very nearly linear with a slope of approximately 0.25 and a y-intercept of 8.4 $\mu g\ m^{-3}$. A similar co-location exercise was conducted at the same site in 2023 yielding a very similar relationship (Longley, 2023).

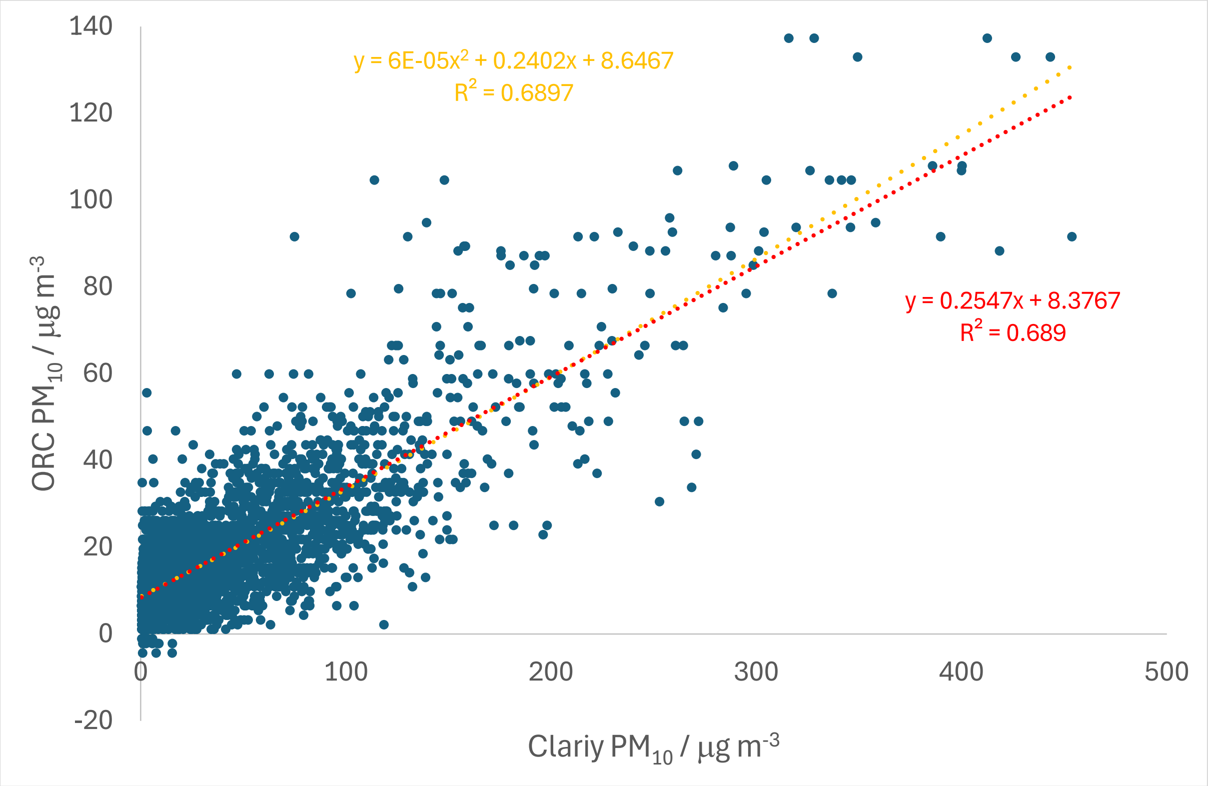

Figure 2: Relationship between hourly PM_{10} measured by ORC regulatory monitor and co-located Clarity monitor

Broadly speaking the slope represents the differential sensitivity of the instruments to the particles being measured and the y-intercept represents regional background particles to which the Clarity is insensitive. We would expect both of these values to vary over time. We explored empirically correcting for this time variation, first variation in the y-intercept (background) and secondly variation in the slope or curvature (sensitivity).

To investigate variation in the y-intercept we estimated its value for every record by

PM_{10} [background] = PM_{10} [ORC] – 0.2547 (PM_{10} [Clarity])

There were several time periods during which regulatory PM_{10} data was not available. For these periods we assigned a default estimate of the background of 8.4 µg m^{-3} (i.e. the y-intercept in Figure 2 and average value).

We then smoothed the result over 12 hours to generate an estimated time series of background PM_{10}. This had an interquartile range of 5.3 – 10.0 µg m^{-3}.

Finally, an adjusted PM_{10} value is calculated by:

PM_{10} [Clarity-adjusted] = 0.2547 (PM_{10} [Clarity]) + estimated background PM_{10}

We found that a 2nd order polynomial equation gave a slightly better model fit than a linear equation (i.e. the value 0.2547 was replaced with the expression 6.44×10^{-5} (PM_{10} [Clarity]) + 0.2402).

The residual error in this estimate was found to have a diurnal variation with under-estimation during the evening (from approximately 4 to 10 pm) and to a lesser extent in the morning (from 7 to 10 am). This corresponds to the time of fresh combustion emissions and a potential change in the optical properties of the particles. Humidity is also suspected to be a modifying factor.

A correction factor as function of time of day (ranging from -2 µg m^{-3} to +4 µg m^{-3}) was empirically derived and added to all data.

The result of the fit adjustment for raw 20-minute data is depicted in Figure 3. This shows relatively good agreement, with a slight tendency for adjusted concentrations to under-estimate reference concentrations.

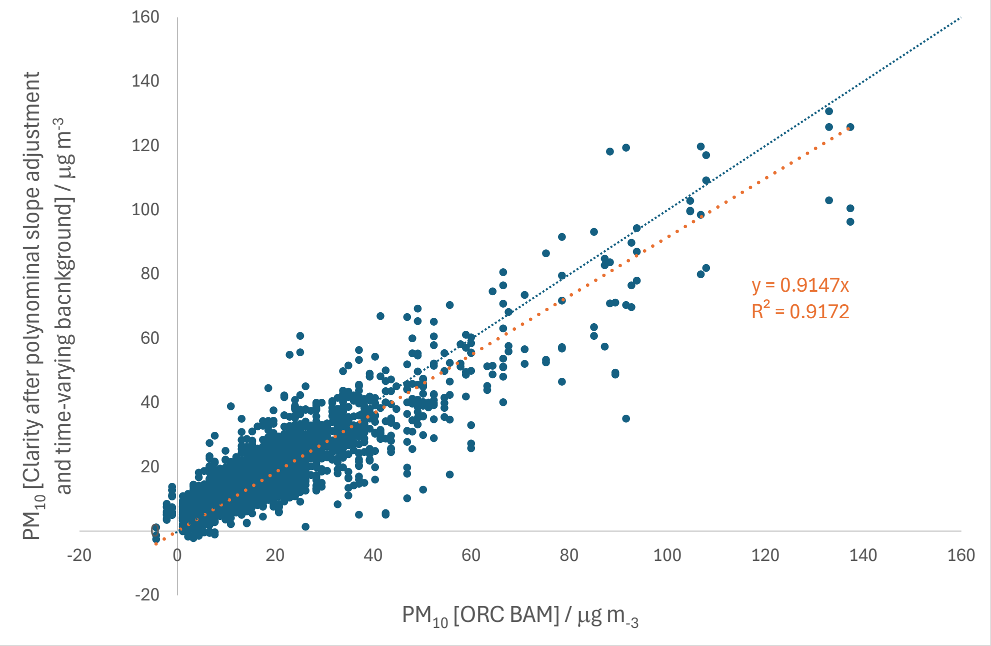

Figure 3: Plot of 20-minute PM_{10} data from the reference instrument and the Clarity node, after adjustment

Further improvements to the adjustment method were investigated but found to yield no significant improvement. Specifically, a time-varying slope function was attempted but introduced as much additional uncertainty as it removed.

We have therefore applied the equation below to all recorded PM_{10} data from all monitors:

Adjusted PM10 = $(6.44 \times 10^{-5} (PM_{10}\,[\text{Clarity}])^2) + (0.2402\, PM_{10}\,[\text{Clarity}]) + PM_{10}\,[\text{calculated background}] + D$

Where D varies from -2 to +4 $\mu g\ m^{-3}$ as a function of time of day.

2.6 Survey

A simple survey was created, firstly as a trial to test response rates, and secondly to get some basic data on the perceptions of outdoor burning, consisting of subjective observations, perceived impacts and self-reported annoyance.

The survey was entirely online and delivered to the community via the ORC web site and social media channels.

3 Results

3.1 Insight into processes controlling air quality in the basin

The library of images captured using the cameras, supported by basin-wide ground-based air quality measurements, have provided an unusually rich basis for understanding the processes that control air quality in the basin. Resource constraints have prevented a full analysis of this extensive dataset. However, this section presents some results, key insights and hypotheses based on an initial analysis.

3.1.1 Emissions

Emissions are covered in more detail in the sections below. However, in brief, the observations collected in this study confirm that smoke from home heating is a common and significant source throughout the winter, typically beginning at 3 to 4 pm, although some homes may be burning wood right through the day.

We also found that outdoor burning was common throughout the study in semi-rural locations across the basin. Cameras were not placed to capture outdoor burning on the Manuherekia valley to the north and north-east of Alexandra, yet we did partially observe fires, or smoke from off-camera fires coming towards the basin from that direction. Outdoor burning fires varies hugely in their duration, intensity and impact, as discussed further below.

3.1.2 Formation/removal: Condensation and evaporation

Many of the video images collected showed a high prevalence of ground-level cloud and/or fog in the basin during winter mornings. This is caused by surface cooling and is reversed by solar heating beginning at sunrise, and particularly around 9 to 11 am. Furthermore, many of the images collected appear to show a noticeable haze left behind once the cloud or fog evaporates. This haze appears to linger for approximately one hour. We have found some evidence of this haze corresponding to elevated PM_{10} concentrations measured in the basin.

This observation is consistent with research that has shown that particles that are scavenged by fog droplets can be re-released into the aerosol phase when the fog evaporates (e.g. Mazoyer et al., 2019). This is more likely in low wind conditions, and where there are abundant organic aerosols (such as those arising from wood burning).

3.1.3 Vertical dispersion: Inversions

Previous studies have indicated the importance of atmospheric inversions in controlling dispersion in the basin and thereby exerting a strong influence on air quality, and that Alexandra's inversions are particularly frequent and persistent (Tate, 2010, Price, 2015). An inversion is a natural phenomenon by which strong cooling at the Earth's surface cools the air at the base of the atmosphere, "inverts" the normal thermal gradients in the atmosphere and suppresses the normal thermally driven vertical mixing of the air. This means that pollution emitted into the lower layers of the atmosphere will experience reduced vertical dispersion. This phenomenon is more common on winter evenings due to rapid cooling. It is also exacerbated in inland valleys and basins where sheltering of surrounding higher land reduces mechanical turbulence and cold air flows downstream off cold mountain slopes and collects in the valley reinforcing the reversed vertical temperature gradient.

The particularly high prevalence and importance of inversions was strongly confirmed by this study. The effect of inversions was observable in two ways:

- Visible changes in haze thickness with height seen in the video images (especially views looking north from the south)
- An often repeated “sawtooth” pattern in the time series of pan-basin PM_{10} concentrations.

To illustrate this second observation, Figure 4 depicts part of the time series of the 25th percentile of PM_{10} concentrations across the basin (excluding site C4 and the reference site, both of which lie in the Alexandra urban area). This provides a reasonable estimate of the degree to which the basin as a whole is “filling up” with accumulating pollutant below the inversion. It shows a moderately consistent accumulation beginning around 5pm and continuing until 9:30 to 10 am after which there is a rapid fall in concentrations reaching the baseline level usually within around 1 - 3 hours.

This “sawtooth” pattern is distinctly different to what is observed within Alexandra (and Clyde) where there is usually a strong concentration peak between sunset and midnight followed by a fall in concentrations, and often a second, smaller rise after sunrise. Two distinctive features of the sawtooth pattern are that concentrations continue to rise all night and are still rising (or at least not falling) after 9am.

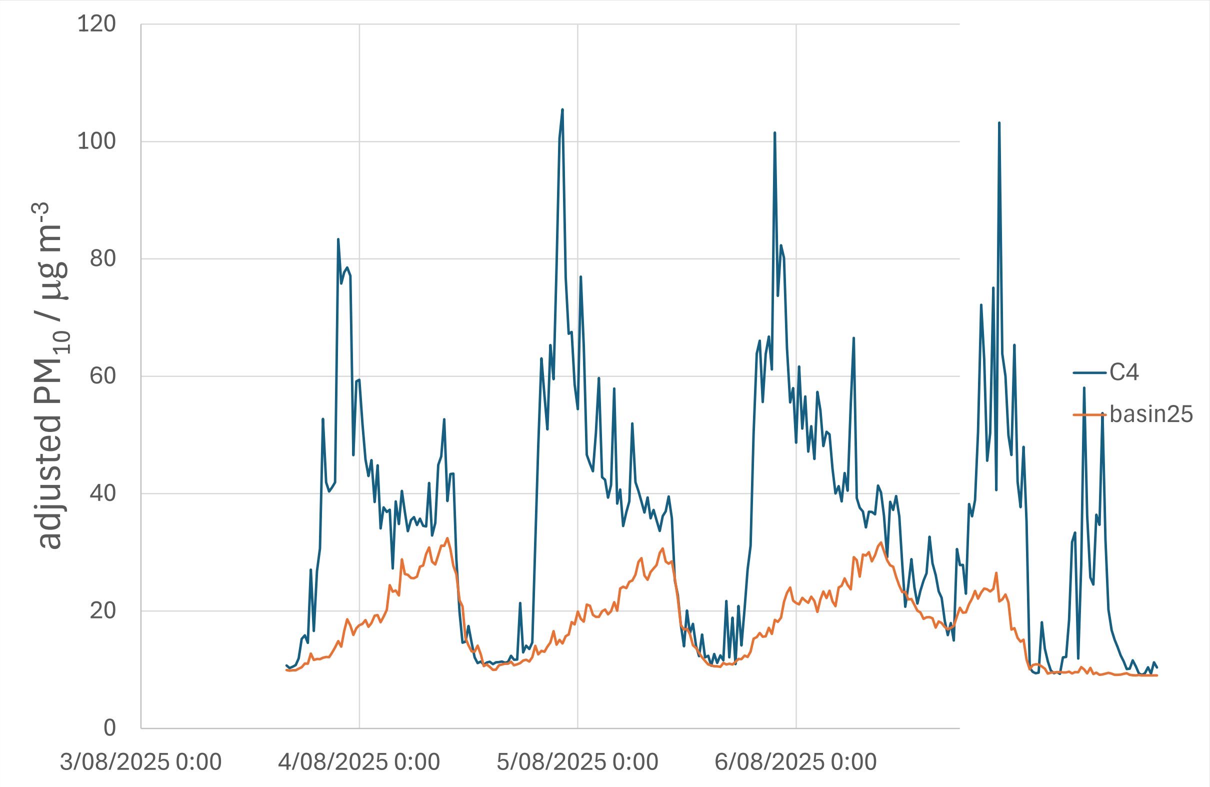

Figure 4: Example of the "Sawtooth" pattern in basin 25th percentile PM_{10} (and PM_{10} at urban site C4) indicating the repeated formation and collapse of an inversion

We attempted to manually classify each day (from 4pm to 4pm) as being an "inversion-controlled day" or not. We found 63 inversion-controlled days between 17th April and 10th September (146 study days), with the most being in July and August (38 out of 62 days). There were 12 consecutive days in late July with full or partial inversions.

The impact of the inversions on air quality (indicated by the 25th percentile concentration across the basin at the time just before the inversion dissipated) rose steadily through April, May and June, stabilising in July before becoming much more variable in August. The typical mid-winter inversion raised basin-wide 25th percentile adjusted PM_{10} concentrations by 16 $\mu g\ m^{-3}$ and a maximum of 34 $\mu g\ m^{-3}$.

Inversions generally began forming around 3:30 pm. Inversions end when heating (from above or below) re-establishes vertical mixing. In this study we found inversions typically began to collapse between 10 am and mid-day in midwinter, and slightly earlier in early and late winter. Generally, it took 2 - 3 hours for the inversion to dissipate, but occasionally took up to 6 hours and on some occasions did not fully dissipate before beginning to re-form.

3.1.4 Lateral dispersion

Images from the video cameras made it clear that smoke was readily transported laterally across the basin, but that the impact of this on dispersion (the reduction in concentrations) was limited because of limited removal. Rapid and regular changes in wind direction at the surface were observed (inferred from the movement of smoke), often apparently unconnected to more consistent winds aloft (inferred form the movement of clouds). Smoke was regularly seen to move in one direction across the basin but soon after move in another direction, often in the reverse direction with the effect of transporting smoke back towards the emission source. That the returning smoke remained visible indicates a relative lack of dilution. These phenomena were mostly observed under inversion conditions.

We infer from this that the combination of the shape of the basin and the formation of inversions allowed smoke to be mixed within the basin, but far less likely to escape the basin. This implies that any smoke emitted within the basin can impact the whole of the basin in the right conditions.

There were occasions when we observed what appeared to be smoke flowing off the Manuherekia plains into the basin. This is consistent with our expectations regarding the effect of "katabatic flow", i.e. cold air sinking down the slopes of the surrounding uplands and flowing down into the basin, transporting any smoke emitted on the plains with it. Although harder to determine from the camera angles selected, we did not find evidence of smoke flowing "uphill" out of the basin during inversion conditions. As inversions lifted the natural relative buoyancy of hot smoke would tend to lift it to a height greater than the basin's north-east rim allowing smoke to exit the basin.

3.2 Visible fires (image analysis)

This project was resourced to deploy cameras to collect video images, but not to conduct analysis of those images. We believe that recent advances in computer vision may be able to yield a substantial amount of information from the images collected.

However, every video collected has been watched at least once and we offer this high-level subjective analysis on that basis.

Results in this section cover images from 24th April to 12th June.

3.2.1 Suitability of the method for identifying fires

Fires were inferred from the presence and direction of motion of visible smoke plumes.

A smoke plume from at least one fire was visible in the images captured by at least one camera nearly every day of the campaign. However, the appearance of the smoke was quite variable. In all cases the seat of the fire or any flames were not visible but in most cases its presence could be inferred. Some of the smoke plumes may have originated from domestic fireplaces. A computer vision method that can pinpoint locations may help in separating these sources from open burning sources.

Smoke varied substantially in terms of thickness, height and extent. Using relatively high viewpoints for the cameras significantly helped in being able to distinguish plumes with significant vertical rise from those that remained close to the surface. The high viewpoints also enabled the boundary layer (cap of the inversion) to be clearly visible at some times.

Smoke plumes may have been obscured in the morning when low cloud or fog was present. This may lead to some under-estimation of the prevalence of open burning smoke in the morning.

In many cases a view of the seat of the fire was probably obscured behind vegetation. This is influenced by the prevalent use of vegetation barriers as wind breaks on the farming properties around the basin.

Subjectively we found that the angle and intensity of sunlight played a significant role in how distinct the smoke plumes were. Frequently smoke that was very distinct in images from the cameras at the southern edge of the basin were barely visible or not visible in images from the other cameras. In other words, cameras on the southern edge looking north were generally more sensitive as they were looking into the sun making it easier to detect the light scattering caused by airborne particulate.

3.2.2 Subjective quantitative analysis

Each video has been reviewed. Any visible plumes have been recorded in terms of their start time, end time and direction (relative to camera view). Where it is suspected the same plume has been detected by more than one camera this has been recorded as one plume.

3.2.3 Temporal variation

The median number of smoke plumes (fires) detected per day was 3. There was only one day when zero plumes were detected. The maximum number of plumes detected in one day was 11.

3.2.4 Diurnal variation

With two exceptions smoke plumes were not observed before 8 am. The most common time for smoke plumes being first detected was between 8 am and 2 pm. After 2 pm the appearance of new plumes gradually reduced until sunset.

The median duration of smoke generation from a source was 70 minutes, the 75th percentile was 2 hours 20 minutes, and the 95th percentile was 4 hours and 52 minutes.

Combining these two pieces of information we found that active fires were most likely to be creating smoke plumes between 11 am and 3 pm.

3.2.5 Fire source locations

A limited manual triangulation was attempted for a selection of fires. A manual method is subject to uncertainty. However, this method suggested that open burning emission sources were spread across the basin, but potentially with a higher concentration in mid-basin locations in Earnscleugh. A computer vision method should be able to pinpoint sources far more accurately.

In some cases, smoke appeared to be entering the basin from the Manuherekia plain, even the though the actual seat of the plume could not be observed (was out of view). An additional camera with a view of the Manuherekia valley would have been advantageous.

3.2.6 Plume dynamics

The movement of a plume represents the interaction of the thermodynamic properties of the plume and the surrounding atmosphere. We observed many of the “classic textbook” plume behaviours. When emitted into an inversion the vertical rise of plumes was suppressed and restricted, sometimes leading to the phenomenon of “fumigation” when pollutants are drawn back down to the surface leading to elevated ground-level concentrations. Conversely, we also observed “lofting” where smoke penetrates the inversion (due to high heat and vigorous vertical motion) and then is dispersed above the inversion without returning to the surface. This leads to a highly visible plume with little or no ground-level impact. We also observed “looping” which occurs in the absence of an inversion and consists of vigorous upward and downward motion of the plume leading to momentary “plume strikes” when the plume briefly touches the surface.

A computer-vision based classification of the dynamics of each plume could enable their relative prevalence and impact to be distinguished.

3.3 Air quality (all sources) in the basin

The following section presents results for “adjusted PM_{10}” (to enable comparison with National Environmental Standards) from 11th April to 16th September 2025 (inclusive), unless otherwise stated.

3.3.1 Long-term mean spatial variation in PM

The mean PM_{10} concentrations across the whole campaign are depicted in Figure 5. Site C4 was located just within the urban area of Alexandra and experienced the highest concentrations overall. Along with the reference site, it is excluded from our calculation of the "basin" average, which was 17.4 μg m^{-3}. This was 73 % of the average at the reference site (23.9 μg m^{-3}) and 66 % of the average at site C4 (26.2 μg m^{-3}).

Away from the edge of Alexandra, the spatial variation of average concentrations was relatively weak with concentrations gradually decreasing with distance from Alexandra and decreasing from the centre towards the edges of the basin.

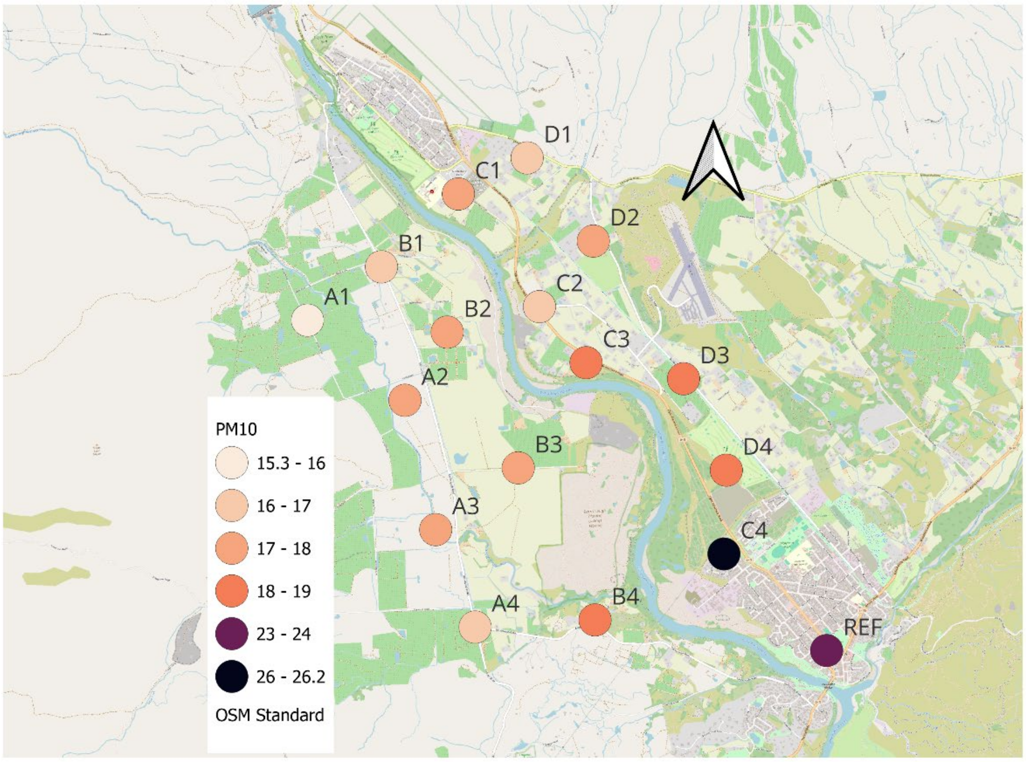

Figure 5: Campaign-mean PM$_{10}$ for all sites

At 10am, the average spatial pattern was somewhat reversed with a slight tendency to higher concentrations towards the basin edges and lower in the centre.

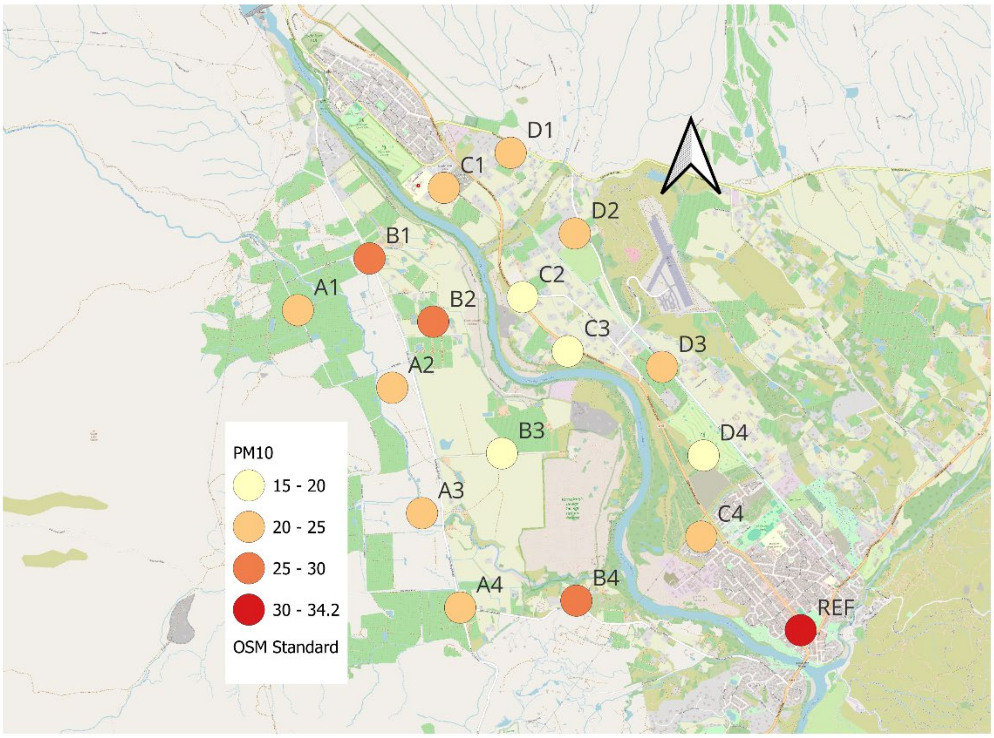

Figure 6: Campaign-mean PM₁₀ for all sites at 10 am

Figure 7 shows how the range of concentrations across the basin varied from day to day.

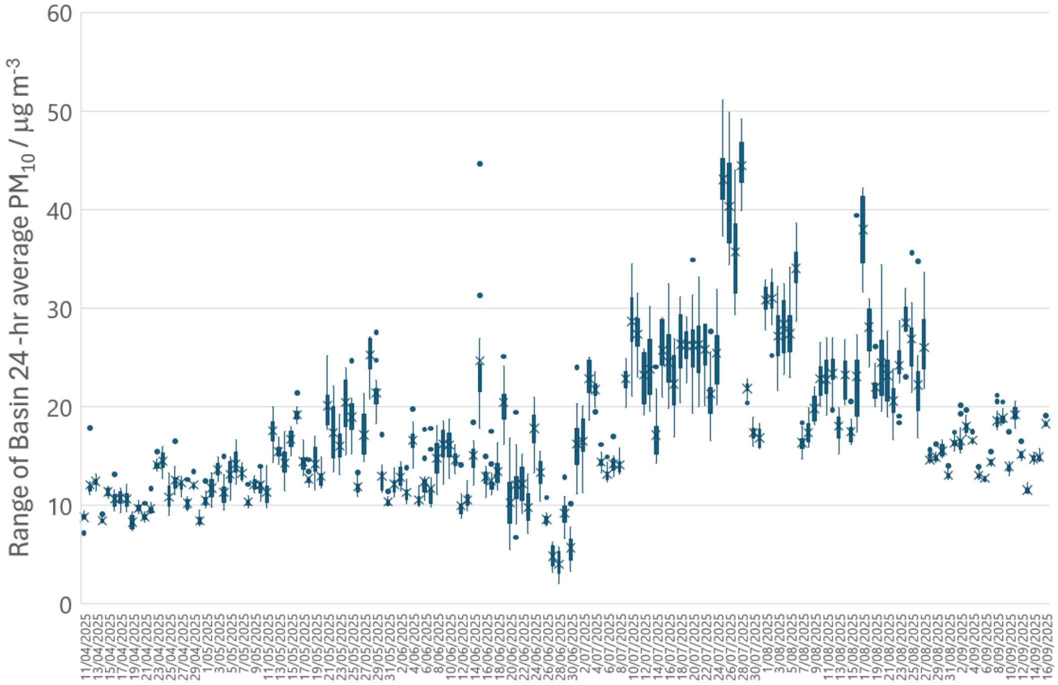

Figure 7: Distribution of 24-hr average PM₁₀ across the 15 basin sites

3.3.2 Relationship between air quality in Alexandra and across the basin

Figure 8 shows that the average PM₁₀ (24-hour) across the basin was slightly lower than that at the Alexandra reference site, but broadly correlated, albeit with the relative difference growing at higher concentrations. This indicates that although PM is moderately well mixed across the basin on average, on high concentration days there is relatively less mixing. This may well be due to high concentrations being associated with reduced wind speeds with emissions into Alexandra's atmosphere occurring at a faster rate than they are being dispersed across the basin.

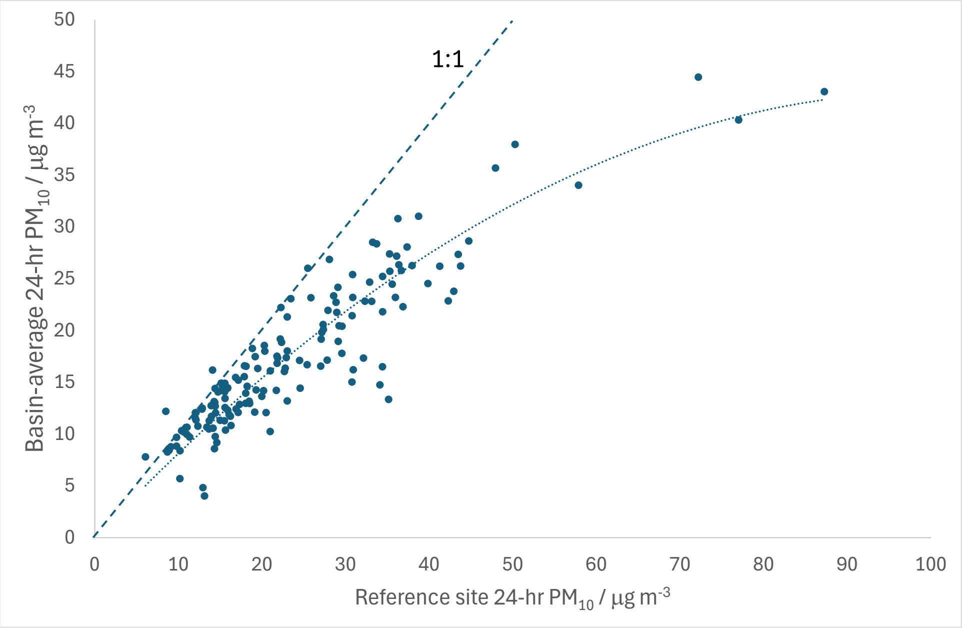

Figure 8: Relationship between 24-hour average PM_{10} at the reference site, and averaged across the basin

3.3.3 Compliance with PM_{10} standard

During this study there were 6 exceedances of the PM_{10} National Environmental Standard, as recorded at the regulatory ORC site. Our monitor placed at the same site (after adjustment, which introduces some uncertainty) recorded 5 (we recorded different results on 28th May).

We also recorded concentrations that would constitute exceedances 10 time at site C4 on Larch Crescent, and once each at site D3 (off Dunstan Rd north of the Golf Course) and D4 (at the Golf Course). Maximum 24-hour PM_{10} concentrations across the remaining sites ranged from 40 $\mu g\ m^{-3}$ (A1) to 49 $\mu g\ m^{-3}$ (B4).

We estimate that were a 24-hour $PM_{2.5}$ standard of 25 $\mu g\ m^{-3}$ in force (as proposed by MfE) then **all sites** across the basin would have experienced approximately 4 exceedances during this study. In other words, the extent of non-compliance with the NESAQ would expand from urban Alexandra to the whole basin.

Furthermore, we estimate that were a 24-hour $PM_{2.5}$ standard of 15 $\mu g\ m^{-3}$ in force (i.e. the World Health Organisation Guideline) then **all sites** across the basin would have experienced approximately 30 – 50 exceedances during this study (approximately 80 in urban Alexandra), i.e. most days in July and August.

3.3.4 Discussion

Determining whether these PM_{10} concentrations are abnormal or typical for a semi-rural area is difficult, mainly due to the extreme scarcity of measurement data from such locations. In an unpublished research study undertaken in 2022, we found a broadly similar quantitative relationship between PM_{10} measurements in Ashburton and at sites ~4 km outside the town although that was restricted to meteorological conditions in which an urban plume was likely to be advected in that direction. Without a more comprehensive comparable dataset from a more open location (like Ashburton) to back it up, we hypothesise that concentrations outside Alexandra are relatively elevated more often (i.e. in a wider range of meteorological conditions) than in places like Ashburton due to the nature of the basin which suppresses dispersion.

3.4 Contribution of non-combustion sources to air quality in the basin

Clues to particles sources can be gained from the relationship between PM_{10} and PM_1 measured by the sensors. This method can be used to distinguish combustion from non-combustion particles. However, it is unlikely to be able to distinguish outdoor burning from home heating sources as they generate aerosols with very similar particle size distributions.

Figure 9 plots PM_{10} against PM_1 for the 20-minute average data from every site over the whole campaign. This figure shows that the vast majority of the data fits an approximate straight line with a slope in the range 2 – 3, indicating an aerosol with a relatively consistent composition which straddles the fine and coarse fraction. This is indicative of particles formed form the combustion of solid fuel.

There are three other clusters.

Points above the main cluster (red oval) represents aerosol with a high coarse particle composition, probably indicating mineral dusts. These are predominantly observed at two sites (B2 and C1) straddling the Clutha River just south of Clyde, and less so at four other sites (A3, B3, C2 and C3) a little further to the south. Either wind-blown or mechanically disturbed dust emissions could explain this.

Points below the main cluster (green oval) represents aerosol with little to no coarse particle composition, possibly representing a very clean-burning fuel. These are predominantly observed at one site (C3) with only 3 other such measurements elsewhere.

The third cluster (blue oval) is the very high concentration cluster where a curvature is seen. These data come entirely from the Ventry Street reference site in Alexandra. We speculate that this might representing particle growth from condensation and adsorption at high concentrations.

In conclusion, this analysis shows that particulate matter in the basin was strongly dominated by combustion sources.

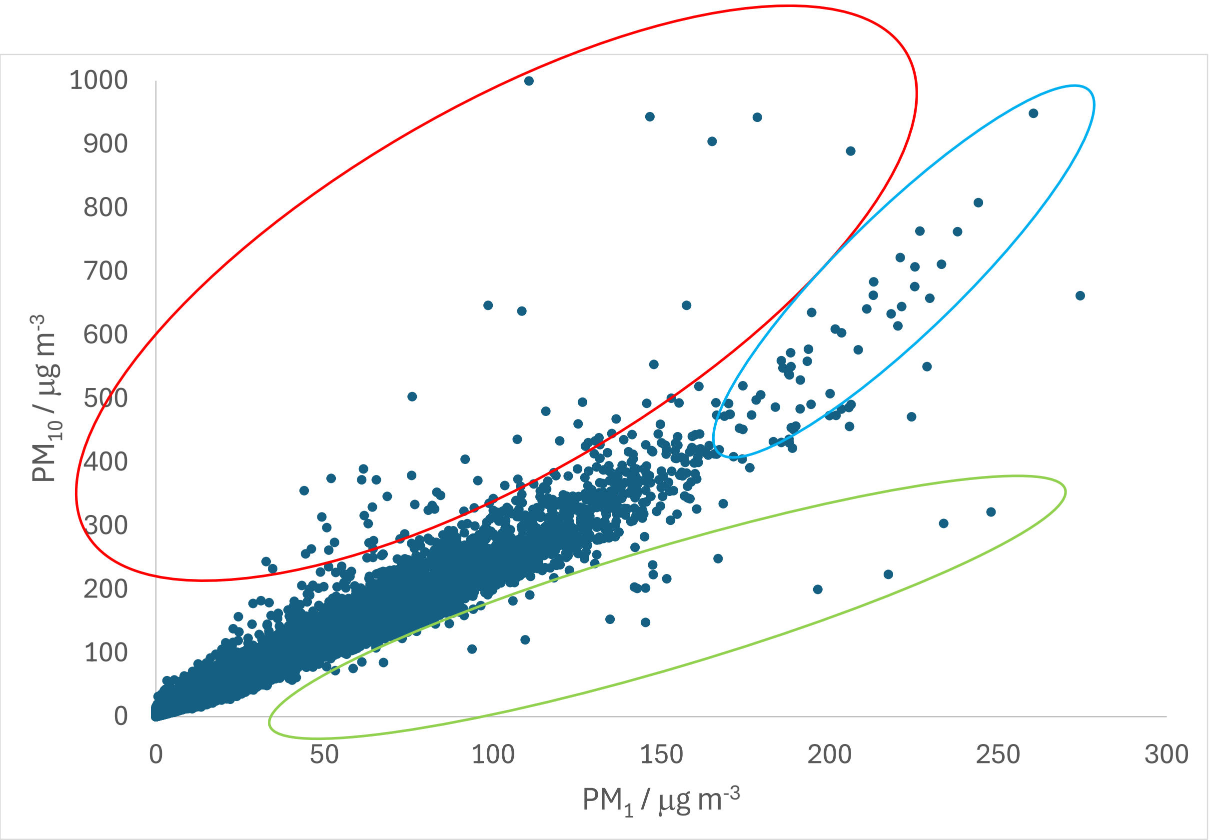

Figure 9: 20-minute average raw (unadjusted) PM_{10} against PM_1 for all sensors for the whole campaign. Coloured ovals indicate clusters described in the text.

3.5 Contribution of outdoor burning to air quality in the basin

3.5.1 Discussion

Distinguishing the relative contribution to air quality of outdoor burning and solid fuel home heating in an observational study is very challenging. The major challenge is that it is largely the same fuel that is being burned (wood and related materials) generating very similar airborne particles of a similar size. Clues can potentially be gained from differences in timing and location of the emission sources.

However, this study has shown how the particular nature of the basin leads to the rapid mixing and accumulation (rather than dispersion and removal) of emissions. This will tend to mix particles from open burning and home heating sources quite rapidly.

This analysis is therefore partial and uncertain and may be considered as providing pieces of incomplete evidence.

To gather these pieces of evidence we conceptually split the data into periods when mixing is likely to be minimised (no inversion, outside the typical hours of home heating) and when mixing is present, consider whether observed air quality can be explained by plausible assumptions about each source.

As noted above, inversions generally collapsed between 10 am and 12 noon. This was accompanied by a rapid reduction in concentrations at most sites across the basin (as indicated by the 25th percentile or basin average). Data from the Alexandra urban sites indicates home heating emissions generally began around 4 pm. Smoke plumes from fires were seen on most days in this time gap. We consider two analyses:

- Was there any evidence of the reduction in basin-wide concentrations being influenced by the presence of fires in the basin?
- What was the impact of fires on concentrations at individual sites?

Both of these analyses are limited in that we have not been able to quantify the smoke plumes (in terms of their magnitude, height, dynamics, etc).

3.5.2 Impact of post-inversion fires on basin-wide air quality

As noted above, inversions generally started collapsing around 10 am, evidenced by a rapid fall in basin-wide concentrations. On some days concentrations fell to a baseline level within approximately one hour. On other days the fall was more gradually, taking approximately 4 hours. There were a few cases in which basin-wide concentrations rose after 10 am, which are discussed separately below.

We could find no qualitative evidence that the rate at which concentrations fell after 10 am was related to fire activity. Indeed, this variation in the rate of concentration reduction could be explained solely by atmospheric conditions. However, if we make the assumption that a reduced rate of concentration reduction is entirely due to local fire activity, then we estimate that the maximum contribution of those additional fires would be of the order of 1 – 1.5 $\mu g\ m^{-3}$ averaged over 24-hours, or less than 10 % overall.

We found a few cases where it appeared that the fall in concentrations was potentially interrupted. This potentially corresponded to periods of vigorous fire activity. However, the effect was to raise basin-wide concentrations by only 1 or 2 $\mu g\ m^{-3}$ for no more than 3 hours. The impact on 24-hour average concentrations would therefore be smaller than 0.5 $\mu g\ m^{-3}$, or 1 or 2 %.

3.5.3 Impact of "events" on localised air quality

Open burning gives rise to intense smoke locally which can, in principle, give rise to short-lived bursts of high concentrations nearby. To investigate this effect the basin-wide 75th percentile concentration was calculated for each 20-minute period. Sites reporting adjusted PM_{10} concentrations more than 10 $\mu g\ m^{-3}$[1] higher than the 75th percentile are assumed to be experiencing a localised impact "event". Concentrations in excess of 50 $\mu g\ m^{-3}$ above the 75th percentile were deemed to be a "serious event".

In total 441 events and 23 serious events between the hours of 10 am and 4 pm were identified.

A selection of these events was assessed by reviewing the corresponding video images before, during and after the event. In many cases no fire could be identified. Instead, haze was seen to emerge following the evaporation of localised cloud or fog, indicating that the

---

[1] The thresholds of 10 mg -3 was chosen arbitrarily.

event was unrelated to fires. A fuller computer vision analysis could help distinguish fog-driven events from fire-driven events.

An example of local impacts from 14th August is illustrated in Figure 10, Figure 11 and Figure 12. Figure 10 shows the interquartile range of concentrations across the basin as an orange band. Concentrations above the 75th percentile are shown as points. The collapse of the inversion (as described above) is indicated by the decay of the central band of concentrations from around 9:30 to 13:00. A very high concentration (109 μg m^{-3}) is seen at a single site (A3) at 9:40. High concentrations are soon after observed at first A1 and A2, then B1 and B2, and then sporadically through the afternoon at A1, A2 and A3.

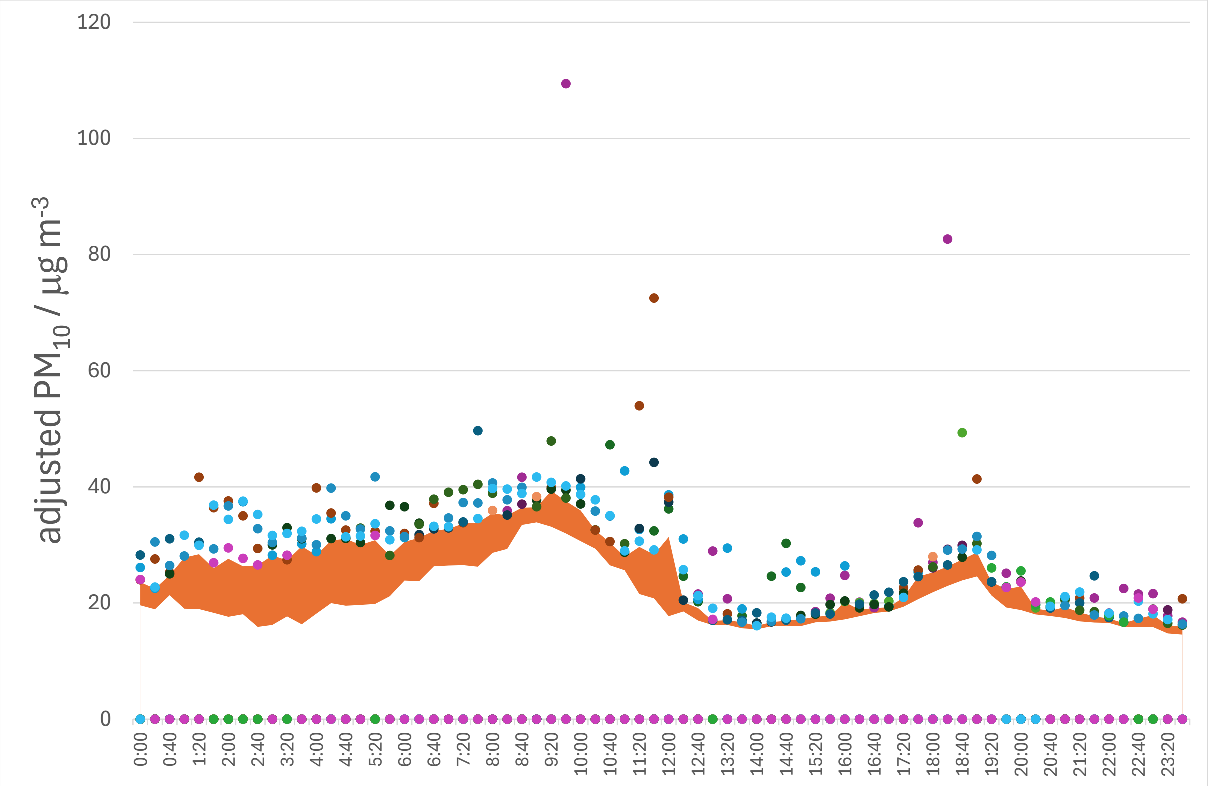

Figure 10: Details of basin interquartile range (orange) band) and locally high concentrations (points) on 14th August.

Figure 11 and Figure 12 show images from one of the cameras through the same day. Smoke can be seen emanating from a point which (through triangulation) we estimate to be near the corner of Earnscleugh Rd and Boulton Rd, which places it between measurement sites A1, A2, B1 and B2. Figure 11 shows how smoke is being emitted under an inversion and (initially) advected to the left, which is towards A1 and A2. By mid-day Figure 10 shows that the inter-quartile range has broadened indicating the smoke is impacting an increasing number of sites. However, in Figure 12 it can be seen that by the afternoon the inversion has substantially weakened, and looping flow has developed. Around 3 pm we see a momentary change in wind direction and possibly an inversion starting to re-establish.

Figure 11: Camera images of smoke being emitted in Earnscleugh on the morning of 14th August showing a gradual collapse of an inversion

Figure 12: camera images from the afternoon of 14th August showing a brief change in wind direction and possible re-establishment of an inversion

As noted above, we have currently been unable to quantify how many of these events are related to fires and how many to other processes (such as particle condensation as fog evaporates). However, we can assess the maximum possible impact of outdoor burning if we assume all events are related to fires.

On average, events added 0.5 μg m^{-3} to the 24-hour average PM_{10} concentration. The maximum contribution was 16.3 μg m^{-3} which was added to 24-hour concentrations at site B3 on 16th August. Thus, the impact of events followed a highly skewed distribution.

There do, however, appear to be some situations where the impact of fires was more significant. We assume that this is indicated by a significant rise in concentrations across the basin between 2 pm and 4 pm which cannot be explained by urban home heating emissions.

We found 6 strong candidates.

For example, on the afternoon of 28th May a significant smoke source was seen to be emitting large volumes of smoke from a site in the vicinity of Hillview Road, just north of the Alexandra urban area and roughly between sites D3 and D4. Emission began around 12 noon and ended around 4 pm, with the most intense emissions between 12 and 2 pm. The video shows a plume initially rising upwards until abruptly hitting an inversion leading the plume to spread laterally at height. The plume then fumigates (is drawn back towards the surface) around 1 pm. The nearest monitor (D4) detects an "event" of ~12 μg m^{-3} (excess) at 1:40 pm as the wind appears to advect the plume to the east and/or south. similar sized events are detected at D3 an hour later. Around 1:30 pm a change in wind direction pushes the plume west or north across the basin reaching the far side around 2:30 pm. An event of 15 μg m^{-3} is observed at D3 at 2:40 pm. The camera images show the smoke to be travelling at low altitude, presumably trapped below an inversion which is unusual for this time of day but does occur on occasion. Although not conclusive, the cameras do not appear to capture any emissions from homes before the sun sets at around 5:20 pm. On this day an official exceedance of the PM_{10} standard was recorded at the ORC Ventry Street site.

Data from the 26th July suggests that the inversion persisted all day as the normal collapse in concentrations around 10 am to 2 pm did not occur. Unfortunately, video coverage for this day was lost. There was an official exceedance of the PM_{10} standard at the ORC Ventry Street site on 26th July.

On 10th July there are at least 2 fires on the east side of the basin, and possibly more on the Manuherekia plains. From the video a shallow inversion is clearly visible. Smoke appears to initially be "fanning" i.e. dispersing laterally at a fixed height, but this later changes to fumigation (smoke brought down to ground level). Earlier fires appear to inject smoke to a greater height, reducing the impact of fumigation. There was no exceedance but PM_{10} at the Ventry Street site was still high at 42 μg m^{-3}`.

On 24th July, there was a visible source of smoke on the Manuherekia plain that was drifting into the basin. At 13:45 a fire begins on the basin floor close to the Airport. By 14:05 the plume was fumigating (Figure 13).

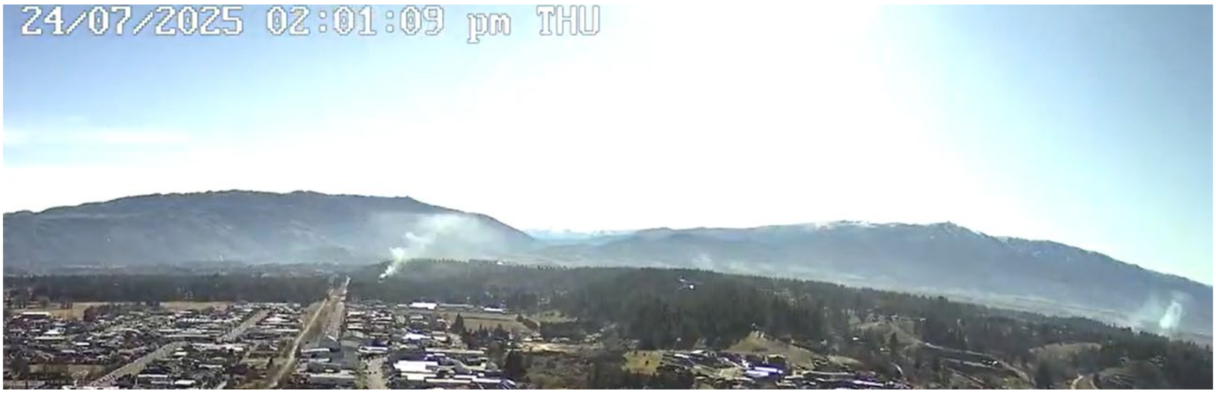

Figure 13: A still from 24th July showing 2 or 3 smoke sources on the plains and one in the basin

On the days highlighted in this section, the basin 25th percentile PM_{10} between 2 pm and 4 pm was 2 – 5 $\mu g\ m^{-3}$ higher than the norm.

Two larger rises of 10 $\mu g\ m^{-3}$ were observed on 22nd May and 27th July. On 22nd May concentrations also rose across the basin from 2 pm. Localised events are not observed. Furthermore, clear smoke sources are difficult to make out on the video images. Despite this, the presence of at least one smoke source on the Manuherekia plains to the north of Alexandra can be inferred, as can the lack of vertical dispersion implying a persistent inversion. Unfortunately, video was not captured on 27th July. There was an exceedance of the PM_{10} standard at Ventry Street on 27th July.

Determining the ultimate impact of these cases requires some assumptions to be made. In the worst-case we can assume that emissions continue until 5 pm, and concentrations continue to rise in that time at the same rate. We can also assume that the smoke generated was trapped within the basin underneath the inversion until 10 am the next morning. So, in the worst case of PM_{10} rising by 10 $\mu g\ m^{-3}$ by 4 pm, we assume this rises to 15 $\mu g\ m^{-3}$ by 5 pm and stays at that level until 10 am. In this case the estimated contribution of these fires to 24-hour PM_{10} will be approximately 12 $\mu g\ m^{-3}$.

We estimate that 24-hr average PM_{10} concentrations in the basin were raised by 5 – 10 $\mu g\ m^{-3}$ on 4 days out of the 181 covered in the study (20th March – 17th September) due to afternoon fires injecting smoke under an inversion.

3.6 Survey results

There were 245 responses to the survey. Approximately half of respondents reported living in Alexandra, 21 % in Clyde, 10 % in Springvale, 8 % in Earnscleugh and 14 % elsewhere.

54 % reported seeing smoke weekly or more often (75 % in Springvale and Letts Gully) and 46 % reported smelling smoke weekly or more often.

The most common locations where smoke was seen were Alexandra (29 %) and Earnscleugh (28 %). Note that Earnscleugh was the area where our analysis found that most smoke plumes originated from.

46 % of respondents found smoke “slight, quite or very annoying”, with 18 % reporting it to be “very annoying”.

32 % of respondents said they believed that smoke from outdoor burning was negatively affecting the health of someone in their household. 21 % of these said this occurred every time there is smoke. There was a strong relationship between respondents seeing or smelling smoke and believing it impacted their health (Figure 14, Figure 15), although this simple survey is unable to indicate if noticing smoke were influencing opinions, or whether a pre-existing opinion regarding smoke made people more likely to notice or recall noticing smoke.

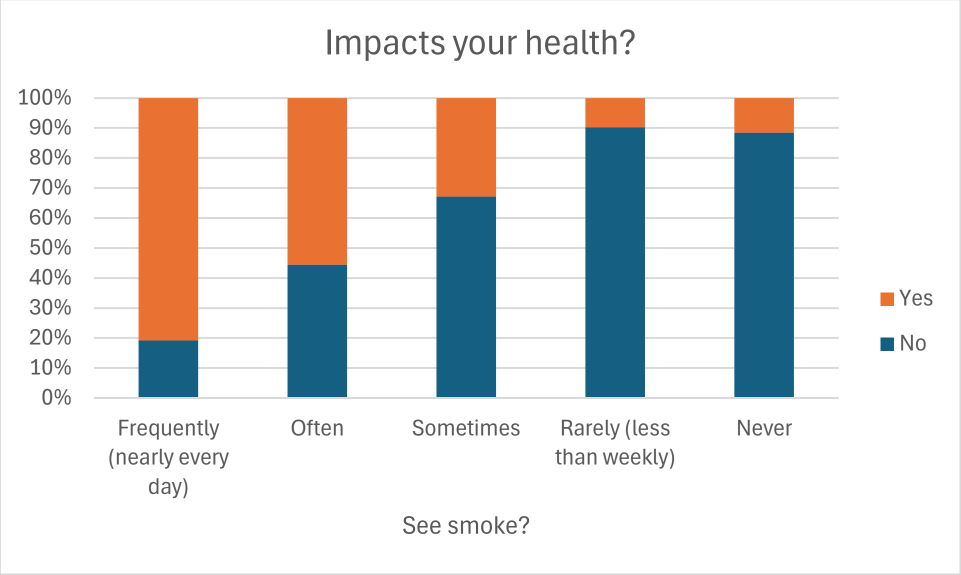

Figure 14: Proportion of respondents reporting smoke impacted the health of a household member as a function of how often they reported seeing smoke

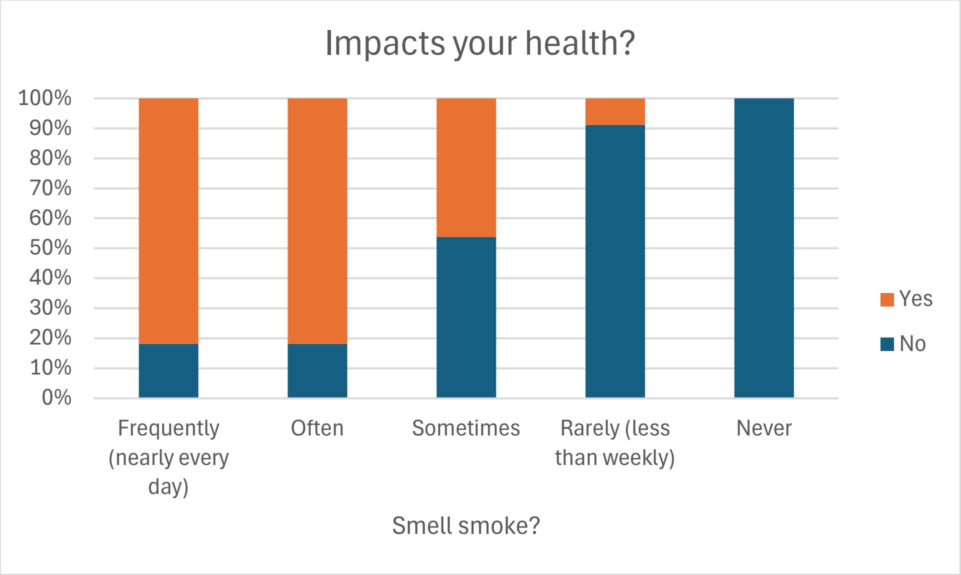

Figure 15: Proportion of respondents reporting smoke impacted the health of a household member as a function of how often they reported smelling smoke outdoors

18 % reported an interest in using an app to report smoke incidents and impacts.

4 Discussion

4.1 Limitations to analysis

4.1.1 Cloud and fog

The regular presence of cloud and/or fog obscured the view from our cameras. This has led to the potential under-counting of outdoor burning events in the mornings. On several images it was possible to make out what appeared to be smoke billowing up within a cloud layer suggesting a possible fire below the cloud.

We noted above the appearance of a distinct haze following the clearance of cloud and fog. Whether or not this haze is related to outdoor burning events is unknown, other than it being feasible.

4.1.2 Boundary layer depth (inversion intensity) not measured

The boundary layer depth (i.e. height of the inversion above the surface) is an important determinant of ground-level air quality as it determines the volume into which pollution is being mixed. This value was not measured in this study, although it may be estimated in some cases from the video images.

4.1.3 Persistence (age of aerosol) unknown

A key unknown in this study is the time period over which particles emitted into the basin atmosphere remain in the basin atmosphere, as this is a major determinant of their impact (for example, their contribution to 24-hour averages). In other words, smoke from a fire burning at 2 pm when there is no inversion may exit the basin within the hour. Conversely, smoke from a fire lit at 5 pm under a developing inversion may feasibly remain trapped within the basin until the inversion lifts, perhaps at mid-day the next day. However, this study is unable to verify this.

4.1.4 Impacts outside the basin

No measurements were made outside of the basin. However, the cameras did clearly record images of significant volumes of smoke on the Manuherekia plains, as well as several significant fires. Video images also captured the movement of smoke in both directions between the plains and the basin. The contribution of sources in the basin to air quality in the plains, and vice versa, remains unknown but is very unlikely to be zero.

4.2 Opportunities for extended analysis

4.2.1 Computer vision and image analysis

We believe there are substantial opportunities for extracting further insight through analysis of the video images. We suspect this would be most efficient and effective using computer vision methods. This holds out the potential for a robust and objective analysis that could include the location, intensity, height and spread of smoke plumes.

The sensitivity of the analysis to the quality and properties of the images is currently unknown and worthy of further research. This should yield important information for future applications of this method – in particular the specifications for the cameras (and their locations with regards to distance from subject, altitude, sun angle, etc) and images collected.

4.2.2 Perception analysis

We hypothesise that attitudes regarding outdoor burning are strongly influenced by the perceptions developed from subjective observation. More specifically, the highly visible nature of some outdoor burning smoke, and the odour than can be associated with it, may drive the perception that the smoke is more impactful in terms of perceived annoyance and distress, than is represented in objective measures like ground-level concentrations. In fact, the general relationships between perceived air quality and measured air quality are poorly understood. Resource constraints, and lack of pre-project data on the appearance and prevalence of smoke in the basin prevented us from exploring these issues within this study. However, the library of images now collected, when coupled with the monitor data, could, in principle, be used as the basis of a research study on how air quality perceived from images aligns with measurements, or otherwise.

5 Conclusions

To the best of our knowledge, this study was unique in that it was the first to use a combination of ground-level grid monitoring and multiple cameras to observe outdoor burning over a 6-month period.

We found that air quality was relatively consistent across the basin, albeit with a weak gradient of concentrations falling with further distance from Alexandra. Average concentrations across the basin were 73 % of the average at the Ventry Street reference site. We estimate one exceedance of the NESAQ for PM_{10} occurred at two of the basin sites closest to Alexandra. We estimate that the proposed $PM_{2.5}$ daily Standard would have been exceeded 4 times at every site across the basin.

A small and brief contribution of dust was seen at some sites, mainly near the Clutha River. Otherwise, the data indicates that PM_{10} was strongly dominated by combustion particles. Distinguishing the contribution of home heating versus open burning is very challenging due to these emissions being mostly chemically and physically identical. We did also observe the formation of significant haze in the hour following the evaporation of cloud or fog, which could briefly elevate PM_{10} levels. This is likely to be the condensation of combustion particles previously scavenged when the cloud or fog formed.

Video images showed outdoor burning to be occurring in the Alexandra-Clyde basin almost every day during the study, with a median of three events per day and a maximum of 11. Burns were most common between 11 am and 3 pm and lasted an average (median) of 70 minutes with some lasting up to 5 hours.

The combination of video images and ground-level monitoring revealed that the impact of smoke from these fires on air quality was highly sensitive to the presence or absence of atmospheric inversions in the basin. An inversion is a natural phenomenon by which vertical dispersion is suppressed leading to smoke emitted below the inversion to be trapped below it, leading to much higher ground-level concentrations than would otherwise be the case.

The Alexandra-Clyde basin is especially prone to the formation of inversions. We inferred the presence of inversions on 63 days out of 146. Inversions tended to form around 4 to 5 pm and persist until 10 am the next day. On some occasions inversions persisted beyond 10 am or re-established earlier than usual.

This means that the basin frequently experiences poor dispersion conditions for 18 hours a day, including the hours when home heating emissions are active. Outdoor burning emissions most commonly occurs during the hours without inversions, i.e. with good dispersion conditions. During these occasions smoke plumes could be quite visible across the basin but have little or no impact on ground-level concentrations due to effective vertical dispersion.

We did observe some outdoor burning smoke plumes emitted into inversions, either before 10 am, after 4 pm, or during those days when inversions occurred between these hours. We estimate that outdoor burning emissions into partial inversions raised basin-wide 24-hour PM_{10} by less than 0.5 $\mu g\ m^{-3}$, or 1 or 2 %, and local concentrations at single sites by an average of 0.5 $\mu g\ m^{-3}$, although the maximum was 16.3 $\mu g\ m^{-3}$.

The worst-case scenario was when inversions persisted or formed earlier in the afternoon. Making the worst-case assumption that smoke emitted into such inversions remains trapped until the inversion lifted the next day, we estimate that 24-hr average PM_{10} concentrations across the basin were raised by 5 – 10 $\mu g\ m^{-3}$ due to afternoon fires injecting smoke under an inversion. This occurred on 6 days out of 181. On three of these days this additional smoke appears to have been sufficient to cause an exceedance of the PM_{10} standard at the Alexandra Ventry Street site.

Figure 11 illustrates many of the key features discussed in this report. To the left of the image smoke rises from a fire in the mid-basin area. A thin vertical plume is topped by a bulbous mass of smoke where the rising smoke has hit the inversion and accumulated. Some lateral dispersion can be seen spreading the smoke across the basin, but at a significant height above the ground (clear air can be seen below it). To the right well-mixed smoke from urban home heating emissions is thick near ground-level (the forest behind the town is well visible). The smoke is clearly drifting away from the edges of Alexandra into the basin.

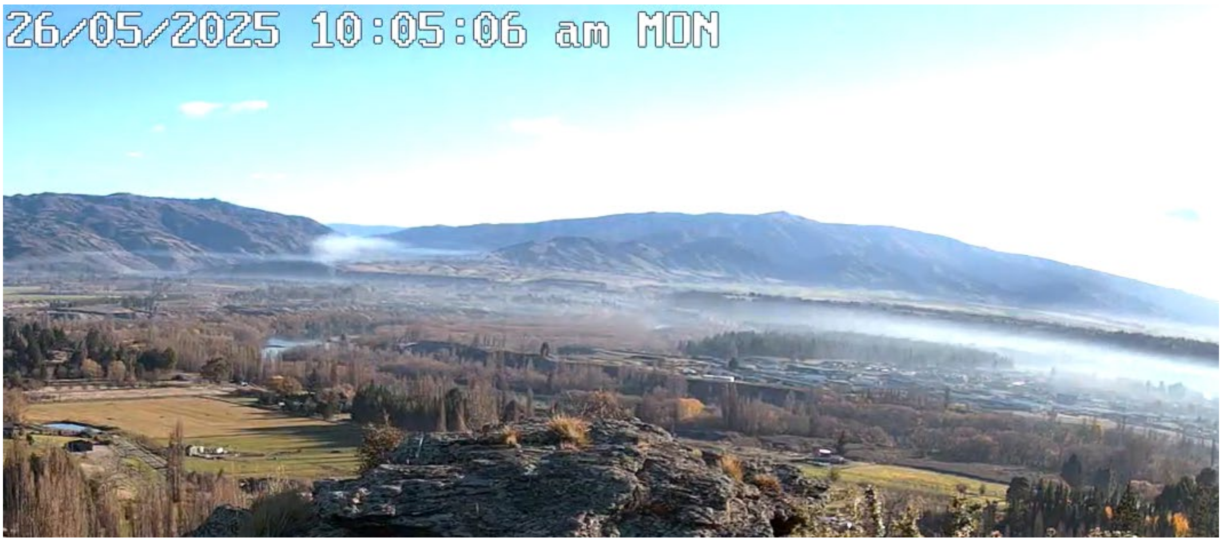

Figure 16: An example of smoke emission from a fire under an inversion (left) and urban home heating emissions being advected out of Alexandra (right)

Overall, we find that smoke from the majority of outdoor burns is well dispersed and makes a minor or negligible impact on air quality measured at ground-level in the basin, due to its buoyancy and effective vertical dispersion. However, the impact on visibility may still be significant.

Conversely, smoke emitted into an inversion can become trapped and have a significant impact across the basin. This is most likely before 10 am or after 4 pm. Emissions in the afternoon are likely to have a greater impact due to the trapping and accumulation of the smoke through the night until the following morning. On a few days per year inversions appear to persist all day. On these occasions smoke form outdoor burning may have a very significant impact on local air quality.

6 Policy implications and recommendations for further work

In our opinion, this work provides a rationale for the regulation of outdoor burning. We find that whereas domestic home heating is the dominant contributor to air pollution and the risk of exceedance of standards, for a few days per year outdoor burning makes a significant contribution to that risk. The question then becomes whether policy action might be able to reduce that risk, and at what cost?

In no particular order, and without regard to their feasibility, we speculate that the following actions might significantly mitigate the risk:

- Reducing/eliminating outdoor burning after 3pm
- Reducing/eliminating outdoor burning on days when inversions are present or have a high risk of occurring (which implies that this can be either predicted or measured, and reliably communicated)

These options are based on our hypothesis that outdoor burning under afternoon inversion conditions generates smoke that accumulates within the basin for 18 hours or more without significant dilution. Confirming this hypothesis to the degree required to justify policy action would probably require additional evidence.

We propose that there are three main approaches to acquiring this additional evidence:

- Extended re-analysis of the dataset from this project
- A dispersion or numerical modelling study
- An extended observational study

An extended observational study could consist of a "tracer" study in which inert tracers are released that simulate emissions from outdoor burns but are chemically distinguishable from all other emissions. Another approach is to study a "rural analogue", i.e. a basin of similar shape to the Alexandra-Clyde basin, but without any urban emissions. This kind of study could measure the dispersion of emissions from existing outdoor burning (a natural experiment), or smoke could be released specifically for the purposes of the study (simulation experiment), subject to ethical and regulatory consent.

A re-analysis is limited by the fact that outdoor burning and home heating emissions were mixed in this study and cannot be separated. An extended observational study would likely be relatively expensive whereas a modelling study may introduce more uncertainty than it reduces. We are happy to advise further on these options.

Any further study, especially observational study, should include consideration of:

- emissions and concentrations on the Manuherekia plains, and the exchange of smoke between the plains and the basin,
- methods to quantify the emissions strength of outdoor burning sources (e.g. through vision analysis)
- methods to quantify the boundary layer depth in the basin.

7 Acknowledgements

With thanks to ORC staff for assistance with deploying and maintaining some instruments. Permission to deploy monitors and cameras was kindly provided by multiple private landlords, and Central Otago District Council.

8 References

Mazoyer, M, Burnet, F, Denjean, C, Roberts, GC, Haeffelin, M, Dupont, JC, Elias, T, 2019. Experimental study of the aerosol impact on fog microphysics. *Atmospheric Chemistry and Physics* 19, 4323-4344.

Price, N, J, 2015. A multi-scale assessment of the air pollution climatology of Alexandra, New Zealand using a combination of field observations and statistical and physical based modelling. MSc thesis. University of Otago

Tate, A, 2010. Wintertime PM10 measurements and modelling in Alexandra and Mosgiel, Otago, New Zealand. MSc thesis. University of Otago