Referee 1:

Comments to Authors:

In their manuscript **"Multi-decadal ozone air quality and the role of temperature in Switzerland during summertime"**, authors investigated changes in summertime ozone and its relationship with temperature from 12 national stations in Switzerland over the past two decades. Decreases in precursor levels have positively affected ozone in remote locations, while ozone is increasing close to busy roads. The ozone formation regime is becoming more NOx-sensitive, and high ozone is associated with hot days. The study is on a topic of relevance and general interest to the readers of ACP. Yet from an ozone chemistry perspective, the findings are not new. Methodologically, the exclusion of any VOC measurement, the omission of ozone production efficiency (OPE) discussion, the lack of quantitative analysis for NOx-temperature dependence discussion and titration discussion, plus correlating trends in ozone with only one key parameter (temperature) is insufficient to reveal photochemical mechanisms that control ozone levels (Line 100). Therefore, I recommend a major revision and am open to review the manuscript again if needed.

We thank Referee 1 for taking the time to review our manuscript and for their feedback. In light of the reviewer's comments, we see the need to further address the novelty of this work. Despite strong precursor reductions, ozone exceedances of healthy levels remain a matter of concern in Switzerland. The last study on ozone in Switzerland before this one addressed data up to 2014, which leaves more than a decade unaccounted for in the current literature. While most studies on ozone air quality investigate a specific city or urban area with concerning air quality levels, the excellent measurement infrastructure of the Swiss NABEL network allows the novel comparison of different site types representative of polluted conditions with large primary emissions (traffic), urban areas with reduced impact of direct emissions (suburban / urban), more pristine conditions with low local anthropogenic emissions (rural) and background conditions with negligible local pollution and some free tropospheric impact. The comparison of these different site types within the rather small geographic area of Switzerland and as part of one standardized network with consistent measurement and data analysis procedures provides a unique opportunity to characterize the dominating pathways of ozone formation, understand the underlying photochemistry and draw conclusions for required control mechanisms. Beyond that, we present evidence for decreasing O_3-temperature sensitivity over time. While this observation has been reported in the literature at various locations before, we show that the underlying mechanism is related to differences in decadal changes of O_3 in different temperature ranges (e.g. increases at traffic sites between 10 and 20°C and decreases above 30°C). Additionally, we suggest that this observation can be attributed to the decreasing impact of titration at polluted sites, which has not been reported before and could be a potential explanation for similar observations in other locations around the world. For these reasons, we believe our study provides new and valuable insights into surface ozone formation chemistry and related air quality impacts, that will be of wide interest to the ACP readership. We have revised the manuscript to clarify these aspects.

Lines 108 ff.: The NABEL network offers a unique framework for comparing O_3 formation mechanisms across a compact geographic region characterized by a high site diversity, including polluted conditions with large local anthropogenic emissions, urban conditions with less primary sources of O_3 precursors, more pristine conditions with low local emissions and background conditions with negligible local pollution and free tropospheric impact. Unlike the majority of the air quality literature focusing on a specific city or urban agglomeration, this study provides an overview of the mechanisms that control O_3 levels under these diverse conditions. Current literature on O_3 air quality in Switzerland (Boleti et al. (2018) and Boleti et al. (2019)) incorporates data through 2014. This study closes this decade-long gap and reveals unexpected increases of O_3 at polluted sites. Finally, this study provides evidence for titration as a driver of changing O_3-temperature sensitivity under polluted conditions, which has not been heretofore reported and may be an important consideration when unraveling photochemical processes in other regions.

Please find our detailed responses to the referees' remaining comments in the following.

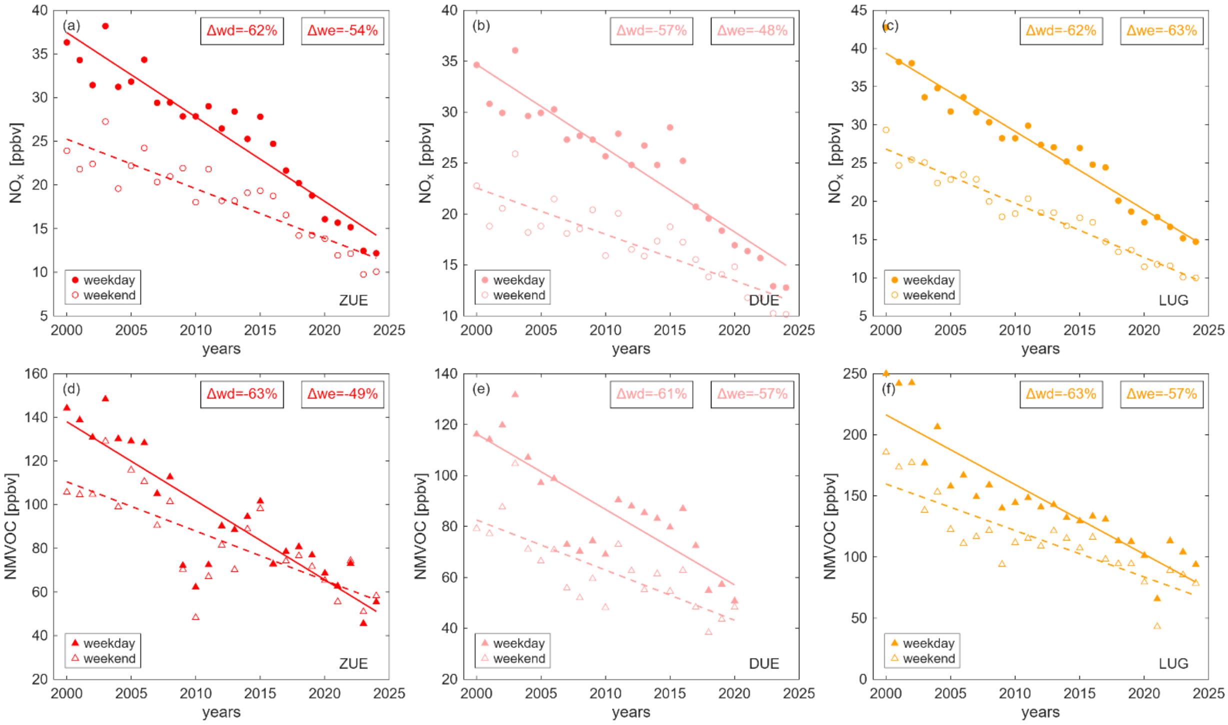
Specific comments:

1. Line 63-75, while I agree the "weekend effect" can be used to qualitatively speculate ozone formation regime, there exists varied quantitative methods to categorize the NOx-sensitive, VOC-sensitive, or transitional regimes based on direct measurement of radicals, or measurement of precursors gases + chemical modeling, or remote sensing of column HCHO/NO2. Given the NABEL network measures ozone, its precursors, and meteorology comprehensively, I am not sure why more quantitative metrics were not adopted in this study. From Line 124-125, it is understandable that VOCs are not always available for all 12 sites. But are they available in a few representative sites (at least one in each of your categories: traffic, (sub)urban, rural, and background) that could be utilized to validate your results/facilitate the discussions in Section 3.2? Diving into the mechanisms of ozone chemistry and its transitioning (Line 224-235) without the inclusion of any VOC measurement or estimation is concerning.

We thank the referee for allowing us to comment further on the ozone sensitivity and the role of VOCs.

We respectfully disagree with the referee that the presented methods are insufficient to characterize the ozone formation mechanism. The categorization of ozone formation into regimes is inherently qualitative (1. NO_x-sensitive, 2. transitional or 3. VOC-sensitive) and the weekend effect is therefore not inferior to other methods, such as the $HCHO/NO_2$ ratio. While the literature suggests qualitative cut-off values for each regime for the latter, e.g. Duncan et al. (2010): VOC-sensitivity for $HCHO/NO_2 < 1$ and NO_x-sensitivity for $HCHO/NO_2 > 2$, these are based on observational or modeling efforts which monitor the change in ozone formation with changes in NO_x or VOCs - similar to the weekend effect. The weekend effect is the only measurement-based approach to assessing the ozone formation regime across the NABEL network of sites.

We agree that the role of VOCs in O_3 formation was unclear in the first version of the manuscript. While VOC measurements are not available at traffic, rural and background sites for the full time period, continuous VOC measurements of the NABEL network are available at the urban sites in Zürich (ZUE), Dübendorf (DUE) and Lugano (LUG):



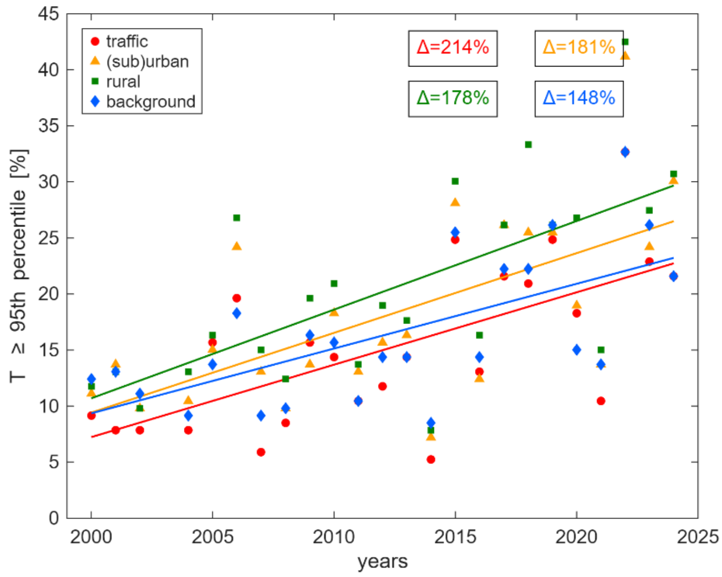
The top and bottom rows present decadal changes of NO_x and NMVOCs at these urban sites, respectively, for weekdays (filled symbols) and weekends (open symbols). These decadal trends show that the magnitude of precursor reductions has been similar for NO_x and VOCs. Therefore, we do not expect any changes in the ambient NO_x level of the transition from VOC- to NO_x-sensitive O_3 formation under urban conditions, and - assuming similar decadal VOC trends across all of Switzerland – at the remaining locations. For dominating NO_x-sensitivity, for example at background and rural sites today, changes in VOCs do not impact O_3 because the formation is limited by the availability of NO rather than peroxy radicals. For VOC-sensitive conditions, a decline in O_3 may result from VOC reductions. However, a precise quantification of the impact would require knowledge of the identity of these VOCs or the overall VOC reactivity, for which measurements are not available. We have added text to the manuscript to clarify the role of VOCs and have added the decadal trends of VOCs in Figure S5 of the Supplement.

Lines 143 ff.: Non-methane volatile organic compounds are measured via flame ionization detection, but the spatial and temporal availability of the measurements is limited. Continuous measurements are available at three urban sites (DUE, LUG and ZUE) only, which we use as an estimation for decadal VOC changes across Switzerland. Long-term speciated VOC or VOC reactivity measurements are not available at these sites.

Lines 277 ff.: Decadal changes of VOCs at ZUE, DUE and LUG (Figure S5 of the Supplement) highlight that the extent of VOC and NO_x reductions was similar over the past 20 years. Therefore, we do not expect any changes in the location of the transition point between VOC- and NO_x-sensitive O_3 formation over time. For sites characterized by NO_x-sensitive chemistry, changes in VOCs do not impact the abundance of O_3. Under VOC-sensitive conditions, a decline in O_3 may result from VOC reductions. However, a precise quantification of the impact would require knowledge of the identity of these VOCs or the overall VOC reactivity, for which additional measurements are needed at all sites.

2. Section 3.3.1 and Figure 5: I am not sure the uniform T>30C is a fair criteria for all sites. As you pointed out, the background sites are at higher elevation and therefore lower temperature and negligible exceedances. I may suggest considering using T>[a nominal percentile of the annual average T] as the more appropriate criteria to filter high temperature days for each site category. This may change your representation of Figure 5b and corresponding discussions. Currently, I am unable to retrieve much useful information from Figure 5b, because 1) the points are looking very scattered- I am not sure how robust is the r-square of that positive slope, 2) it is lumped from all sites, while all other discussions/figures throughout your manuscript are categorized into sites.

Thank you for this suggestion. We have investigated the decadal changes in peak temperatures considering daily maximum temperature exceedances of the 95th percentile of the dataset for each site type:



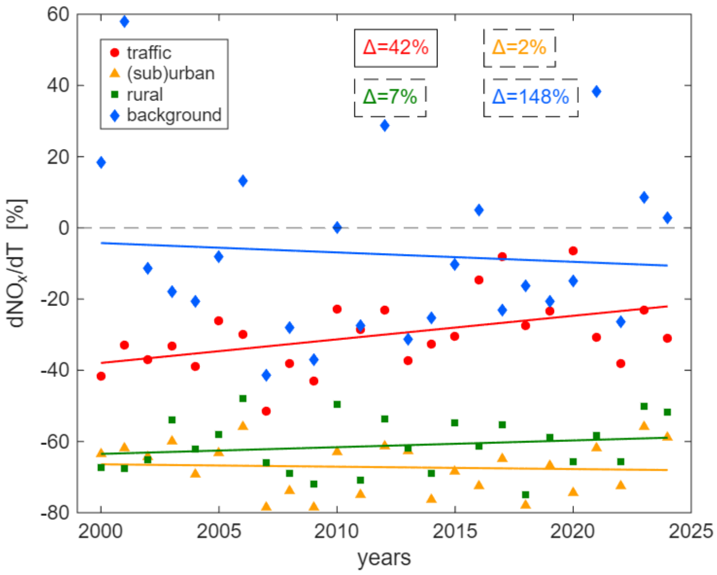
Similar to Figure 5(b) the results indicate that peak temperatures have increased by a factor of 2.5-3 at all sites. We agree with the referee that this illustration highlights the increase of peak temperatures at all sites well, which is particularly relevant when thinking about a climate penalty. However, we are discussing the impact of different temperature ranges on O_3 levels throughout the manuscript, for which fixed temperature values are beneficial: O_3 levels and exceedances are more dependent on the absolute temperature rather than the peak percentile of an individual site. We therefore keep Figure 5(b) in its current state but have added the figure showing the decadal peak temperature changes as 95th percentile exceedances to Figure S7 of the Supplement and added some additional explanation in the main text.

Lines 312 ff.: We present the share of daily maximum temperature exceedances of the 95th percentile of all measurements at individual site types in Figure S7 of the Supplement, which highlights that peak temperatures have increased similarly at all sites independent of the altitude.

While the referee is correct that Figure 5(b) shows a certain variability, highlighted by the moderate correlation of R^2=0.4, the decadal increase is statistically significant (p-value<0.05). The standard error of the calculated slope is around 25%, therefore the increase in the occurrence of peak temperatures would still be large at the lower uncertainty range. In comparison, the correlation coefficients for the linear fits of the decadal peak temperature changes as 95th percentile exceedances range between 0.40 and 0.44.

3. Section 3.3.2 and Figure 6: per Line 264-265, "Figure 6 presents the relationship between (a) NOx, (b) O3, (c) Ox and (d) the share of O3 in Ox with temperature, which all exhibit strong correlations." Would you please clarify if the data points here are from all years? If that is the case, NOx went down (due to combustion control) and temperature went up (due to climate change) over years. They will inherently show a negative correlation, right? This correlation could be entirely physical and has nothing to do with chemistry. Therefore, I am not convinced with the interpretation from Figure 6a that higher temperatures drive a lower NOx or the temperature dependence of NOx emissions (Line 268-289). I suggest additional analysis (e.g. multivariate regression) to really distinguish the temperature impact from other confounding factors to validate the corresponding conclusion. Moreover, for the BLH discussions around Line 300, it reads speculative. Even without direct BLH measurement, your dilution theory should be validated using any gas/particle species co-measured at the NABEL that has a lifetime >> NOx.

It is correct that Figure 6 includes data from all years. We agree with the referee that NO_x and temperature could be anti-correlated over long timescales (i.e. years) given independent trends in emissions and climate. However, this does not drive the correlations at an hourly time scale (as in our analysis). To rule out a mathematical reason for the negative temperature correlation, we have investigated the change of the NO_x-temperature correlation over time. This figure shows the relative change of NO_x between 10 and 30°C:



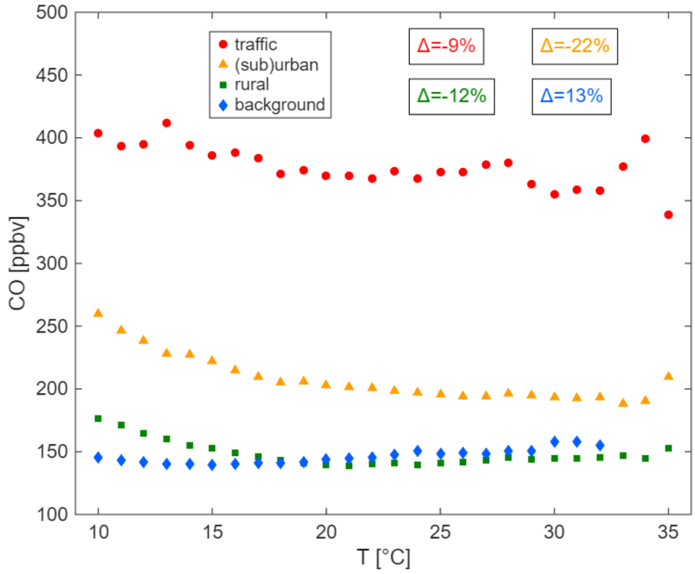
The NO_x-temperature correlation has remained constant at urban, rural and background sites. We observe a small positive change for traffic sites, but all values remain negative over the investigated time period. We can therefore exclude that simultaneous NO_x-decreases and T-increases over time affected the observed NO_x-temperature correlation. We present this finding in Figure S10 of the Supplement and have added text in the manuscript.

L. 350 ff.: The NO_x-temperature correlation (Figure S10 of the Supplement) exhibits little to no change from 2000 to 2024. This highlights that simultaneous NO_x

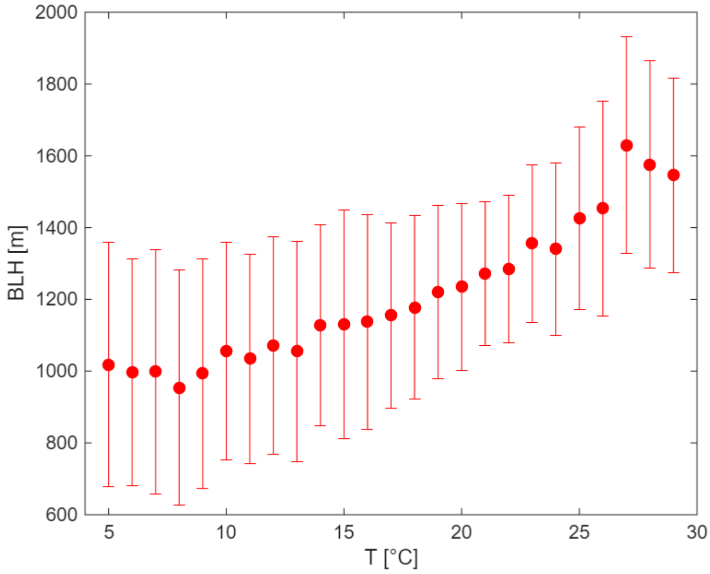
reductions and temperature increases can be ruled out as a reason for the observed NO_x-temperature correlation.

Caption Figure 6: The markers show the averages for all daytime (9-18) hourly data (2000-2024) for each temperature bin.

Carbon monoxide is co-measured at the NABEL sites and has a longer lifetime than NO_x. We also observe a negative CO-temperature correlation:



However, CO might also have temperature-dependent emission sources. An additional challenge is that due to the longer lifetime, emissions and meteorological conditions of previous days and weeks could distort the conclusions. We are therefore not confident that a longer-lived species could demonstrate the boundary layer impact convincingly. We have additionally investigated ERA5 reanalysis data across Switzerland:



This figure shows the daily boundary layer height at 2pm binned to temperature (average across Switzerland). The resolution of the dataset is 0.25°x0.25°, which cannot resolve the topography of Switzerland. However, it can provide an estimation of the BLH-temperature correlation, which is strongly positive and therefore in support of our dilution theory.

Without an extensive source apportionment of NO_x, a footprint analysis and a characterization of the temperature correlation of these sources, which is outside of the scope of this study, we are not able to draw final conclusions on the roots of the NO_x-temperature correlation. Instead, we have added the BLH-temperature correlation to Figure S11 of the Supplement in support of the dilution theory. The revised text further clarifies that while we explore potential mechanisms, a definitive explanation is not possible within the scope of this study.

Lines 358 ff.: Figure S11 presents the BLH-temperature correlation across Switzerland, based on ERA5 reanalysis data of the daily summertime BLH and the 2m-temperature at 14:00 local time. While the resolution of the ERA5 data (0.25°x0.25°) is not sufficient to resolve the topography of Switzerland, it provides an estimation of the BLH-temperature correlation. The positive correlation supports our theory that dilution effects could impact the temperature correlation of trace gases and that the day-to-day variability of the BLH is significant.

Lines 366 ff.: The hypotheses discussed above provide likely explanations for the observed NO_x-T correlation. However, the definitive driver(s) can only be identified through extensive source apportionment, footprint analysis and a precise characterization of the temperature behavior of these sources, which is outside the scope of this study.

4. Throughout the manuscript, ozone "titration" is repeatedly mentioned. However, in both the manuscript and SI, I am unable to identify any quantitative analysis that showed the actual titration effect. Given the main dataset for the analysis focused on summer daytime, I am not convinced of the importance of the titration effect on explaining many observations. From my limited experience, titration usually happens at night-time or early morning, when NO persists (no photolysis) and the boundary layer is shallow. But during daytime, especially between 9:00-18:00 local time in summer as the author stated, NO emitted to the troposphere should theoretically be rapidly converted to NO2 by ROx and HOx within minutes, orders of magnitude faster than NO + O3 titration reaction. But I am no expert on the atmospheric condition in Switzerland. Since you do have simultaneous NO, NO2, and O3 measurements, why not plot the diurnal average of O3 and NO at each site (or categories of lumped sites) to see if NO increases at traffic hours, does ozone drop? If that anti-correlation is obvious and universal across the traffic sites, then you can attribute some of your findings to titration. If O3 remains high even at rush hours, then titration is negligible. You could also try plotting the daytime vs. nighttime O3/NO ratio, as the nighttime titration should be obvious.

The referee is correct that the rate constant for NO + HOx/ROx is significantly larger than the rate constant for NO + O3:

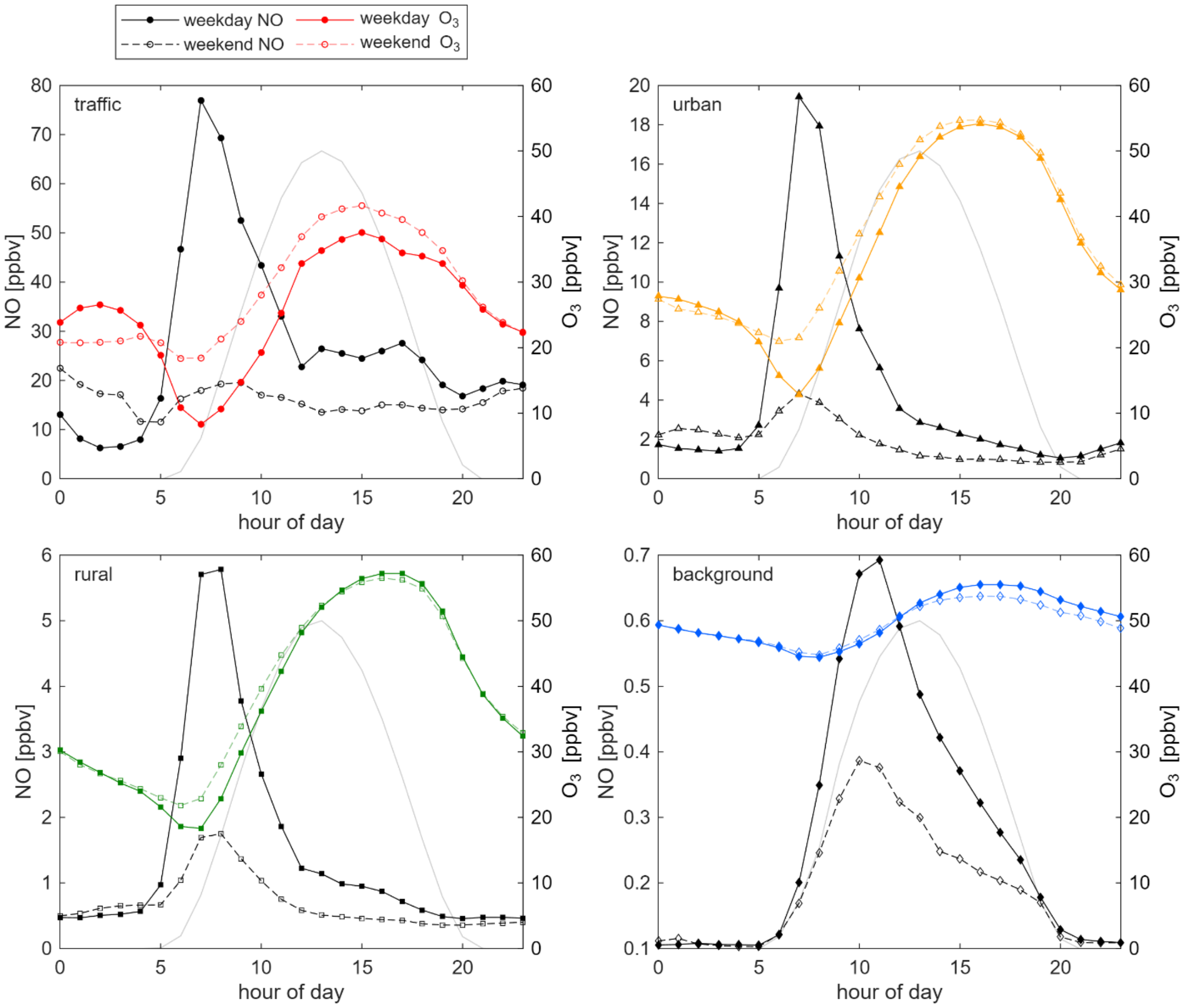
$k(NO+HO_2)$ = 8.5e-12 cm^3 $molecule^{-1}$ s^{-1} (at 298K, IUPAC)

$k(NO+CH_3O_2)$ = 7.7e-12 cm^3 $molecule^{-1}$ s^{-1} (at 298K, IUPAC)

$k(NO+O_3)$ = 1.9e-14 cm^3 $molecule^{-1}$ s^{-1} (at 298K, IUPAC)

At 25°C, the reaction of NO + O_3 is around 400-500 times slower than NO + peroxy radicals, however O_3 is much more abundant than HO_2 or CH_3O_2. While we do not have any measurements of peroxy radicals, levels are typically of the order of ppt compared to ppb levels of O_3. Therefore, the amount of NO that reacts with O_3 is much larger compared to the reaction with peroxy radicals.

We thank the referee for the suggestion to investigate diurnal cycle as an additional confirmation of the role of titration:



This figure shows the diurnal cycles of NO (black lines) and O_3 (coloured lines) at traffic, urban, rural and background sites, separated into weekdays (solid lined) and weekends (dashed lines). We show data for the beginning of the record (for summer days between 2000 and 2005), when titration was particularly important at polluted sites. The gray line indicates the diurnal cycle of radiation, normalized to the right y-axis to fit in the shown frame. Traffic sites are characterized by a large morning NO peak, which can be majorly attributed to the morning rush hour as demonstrated by the large weekday-weekend difference. It is accompanied by a distinct daily O_3 minimum, which supports the conclusion that titration plays a major role under these conditions. This effect is also visible for (sub)urban and rural sites but weakens when moving away from fresh vehicle NO emissions. Further away from primary sources, O_3 diurnal cycles are dominated by other mechanisms: O_3 levels decrease throughout the night due to nighttime loss mechanisms and are increasing after sunrise due to beginning photochemistry. Additionally at low elevation sites, the rise of the boundary layer in the morning mixes the O_3-poor stable nighttime boundary layer with O_3-richer air from the residual layer. The diurnal cycle of O_3 at background sites is less pronounced, which is likely due to the elevated altitude of the sites, which could be located outside of the boundary layer during the night and potentially parts of the day. We have added the diurnal cycles to Figure S6 of the Supplement.

Lines 295 ff.: This conclusion is additionally supported by the diurnal cycle of NO and O_3 (Figure S6 of the Supplement), which shows that weekday morning NO peaks at traffic sites are associated with distinct daily O_3 minima.

We have additionally clarified the term "titration" in the introduction.

Lines 59 ff.: The term titration refers to a temporary sink of O_3 through reaction with NO, which can dominate NOx cycling at night due to the absence of NO_2 photolysis or in proximity to large primary NO sources, which rapidly convert all or a part of O_3 to NO_2.

Technical corrections:

1. Around Line 50, discussion in Pusede et al. (2015) is an estimation of PO3 as a function of NOx and VOCR in a typical U.S. city under 2015 emissions, and the EKMA figure itself is a steady-state simplification. It is classic and important, but since then, more comprehensive field observation-based modeling studies in populated areas around the world have shown that peak ozone formation did not necessarily happen during the transitional regime but rather highly localized. They can occur well within the NOx- or VOC-sensitive regime:

- Peak PO3 in VOC-sensitive regime: Los Angeles, CA, U.S.: Stockwell et al., 2025 (https://doi.org/10.5194/acp-25-1121-2025).

- Peak PO3 in NOx-sensitive regime: San Antonio, TX, U.S.: Guo et al., 2020 (https://doi.org/10.1016/j.atmosenv.2021.118624).

- Peak PO3 in VOC- and NOx-sensitive regime in the morning and afternoon respectively, but not the transitional regime in between: Houston, TX, U.S.: Mazzuca et al., 2016 (https://doi.org/10.5194/acp-16-14463-2016).

- Peak PO3 in NOx-sensitive regime: North China Plain, China: Tan et al., 2024 (https://doi.org/10.1016/j.scib.2018.07.001).

I recommend incorporating these studies in the introduction and revise the corresponding lines.

We have revised the text to highlight that peak $P(O_3)$ occurs at the transition point in an idealized scenario but that depending on the spatial resolution and the meteorological conditions peak $P(O_3)$ can also occur in air masses that are characterized as NO_x- or VOC-sensitive. We have added the suggested citations to the manuscript.

Lines 49 ff.: The crossover between NO_x- and VOC-sensitive chemistry is described as a transitional regime (Pusede et al., 2015). While in theory O_3 formation peaks in this transition (given a large local, homogeneous air mass), depending on the spatial resolution and the meteorological conditions maximum O_3 production can also occur in air masses characterized as NO_x- or VOC-sensitive. Several studies have reported this observation in the U.S. and China (Mazzuca et al., 2016; Tan et al., 2018; Guo et al., 2021; Stockwell et al., 2025).

2. Section 3.3.2, Line 265, do you mean "data points above 10 and below 35C" instead?

Thank you, we have corrected the sentence.

3. I see no where that ozone production efficiency (OPE) was mentioned, which is modulated by both precursor levels and meteorological conditions (jNO2, temperature, cloud coverage, etc.). With NOx and O3 measurement but without NOy, it won't be easy to use the regression method to calculate OPE. But it should be considered in the discussion, because when the NOx went down and T went up, the OPE could increase and lead to more rapid ozone production (more propagation cycles) as shown in:

- Kleinman et al., 2002 (https://doi.org/10.1029/2002JD002529)

- Chace et al., 2025 (https://pubs.acs.org/doi/10.1021/acs.est.5c02073)

And also in those referred literature above. It won't change the main conclusion of your study so I listed it in technical corrections. But it is important to be incorporated and might facilitate some of your discussions (e.g. around Line 365, Line 380, Line 410, and else where).

As the referee points out correctly, unfortunately we do not have any NO_y measurements needed to calculate the OPE. However, we added the suggested citations and some text referring to the potential impact of OPE changes.

Lines 54 ff.: Various parameters, including ambient NO_x levels, VOC reactivity, temperature and photolysis rates, can additionally impact the number of O_3 molecules produced per NO_x, which is referred to as the ozone production efficiency (OPE) (Kleinman et al., 2002; Chace et al., 2025).

Lines 430 ff.: Changes in the ozone production efficiency over time could additionally impact the temperature sensitivity of O_3. Local measurements of NO_y would be required to investigate OPE changes.

Referee 2:

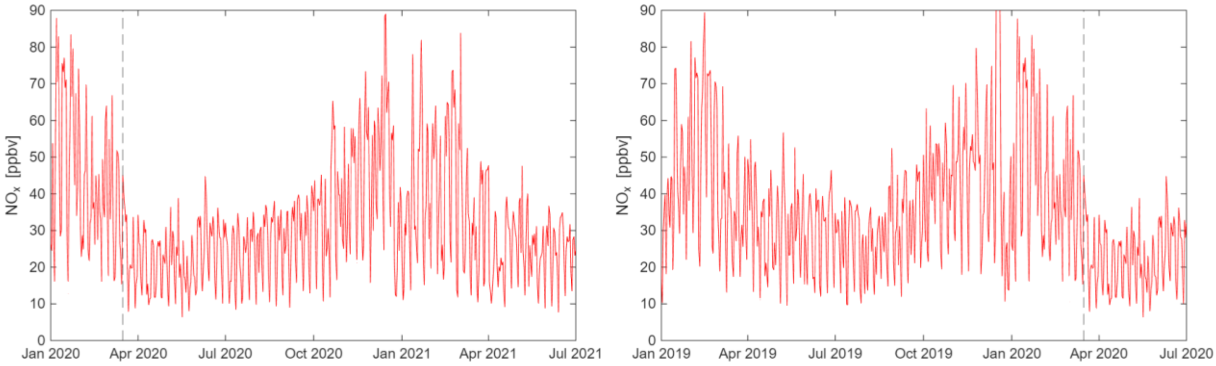
In this work, approximately two decades of oxidant measurements from a network of ground sites across Switzerland are analyzed to better understand the state of ozone formation chemistry across the region. Although the data streams are relatively simple, the analysis appears thorough and the analytical approach with respect to characterizing mechanistic sensitivities is appropriate for the target journal. This work is of clear importance in advancing the state of knowledge of air quality drivers in Switzerland and has broader implications for advancing the field of ozone and NOx chemistry characterization from ground-based measurements. I particularly commend the authors for a very clearly communicated and easy to follow manuscript which was a real pleasure to read. I believe that this work will be appropriate for publication once the following comments have been addressed:

We thank Referee 2 for taking the time to review our manuscript and for their positive feedback.

General Comments:

Some societal and historical phenomena which have great potential significance for analyses here described, specifically those related to traffic patterns, are not sufficiently addressed and should be considered (similarly to relevant phenomena around fuel standards and events such as Dieselgate, which are well discussed). Specifically, I request that the authors discuss changing work patterns (specifically increases in hybrid and remote work), which could be contributing significantly to changes in the weekend vs weekday phenomena here discussed. Is there available traffic data to demonstrate that the relative week day vs weekend number of vehicles on the road has remained constant? I would also request that the authors specifically address the COVID-19 pandemic and related changes in economic activity, which have been extensively studied as significant perturbations of normal traffic and emissions patterns across the world. I do not have sufficient local knowledge to comment on how policies may have impacted the regions in question, but the possibility of a perturbation should be acknowledged and assessed even if there is evidence that it was negligible and therefore ruled out.

We thank the referee for this suggestion. While the COVID-19 measures in Switzerland were generally more moderate compared to neighboring countries, changes in the mobility behavior can be observed in response to the pandemic. Due to the relatively short duration of the implemented measures, the response in precursor levels was rather small, particularly when investigating yearly summertime averages. According to the Swiss Federal Statistical Office the reduction in passenger transport was largest in aviation (~75% reduction in 2020) followed by railway transport (~35% reduction). Private motorized road transport declined by around 20%. All types of passenger transport increased again after 2020. Minor impacts in NO_x levels can be seen when comparing data from 2020 with 2019 (pre-COVID):



The left panel shows NO_x levels between January 2020 and July 2021; the right panel shows the previous year. The dashed lines indicate the legislative measure in Switzerland on 16th March 2020: Federal Council declares the ‘extraordinary situation’, followed by measures to decelerate the spread of the COVID-19 virus. Between the beginning of these measures and their relaxation approximately two months later (e.g. re-opening of schools on 11th May), NO_x mixing ratios were 22 ± 14 ppbv at traffic sites compared to 32 ± 21 ppbv in the previous year. In comparison, the difference was much smaller in the following three summer months with 24 ± 14 ppbv and 28 ± 16 ppbv, respectively. Overall, the NO_x reductions are visible both at traffic and (sub)urban stations with larger values in 2019 and 2021, than 2020. However, the difference is in the range of the observed year-to-year variability for other years.

According to a study of the Swiss Federal Statistical Office, the pre-pandemic share of hybrid or remote work was around 25%, peaked during the pandemic (~40%) and stabilized afterwards with around 37%. However, at the same time, the population in Switzerland is increasing by around 1% each year, and the number of private motorized vehicles has increased from 3.5 Mio. in 2000 to 4.8 Mio. today. The number of traffic congestion hours on national roads has continuously increased since the Covid pandemic and was approximately twice as high in 2024 compared to pre-Covid levels.

We have added a discussion of these aspects to the manuscript.

Lines 201 ff.: The COVID-19 pandemic led to governmental measures to decelerate the spread of the virus and resulted in decreases in primary pollutants in many countries (Gkatzelis et al., 2021). While the lockdown measures in Switzerland were overall more moderate in comparison to the European average, reductions of around 20 % in private motorized road transport in 2020 compared to 2019 were reported by the Swiss Federal Office of Statistics (Bundesamt für Statistik BFS, 2024, 2025b). On March 16, 2020 the Swiss federal council announced an "extraordinary situation" and introduced measures to contain the pandemic, which lasted approximately two months (Schweizerischer Bundesrat, 2020). For this time period, we observe reductions in NO_x mixing ratios by around 1/3 in comparison to the previous year. The reduction was only around 10-15 % in the following summer months. NO_x summertime averages were lower in 2020 compared to 2019 and 2021 at traffic and (sub)urban sites, which could be an outcome of the COVID-19 measures. However, the difference is in the range of the observed year-to-year variability for other years. Further aspects, which could impact the decadal NO_x trend are changes in hybrid and remote work, for which the Swiss Federal Statistical Office reported an increase from around 25 % for pre-COVID years to 37 % in recent years (Bundesamt für Statistik BFS, 2025a). However, the population in Switzerland is currently increasing by around 1 % per year and the number of private motorized vehicles has increased from around 3.5 million in 2000 to 4.8 million today (Bundesamt für Statistik BFS, 2024). The number of traffic congestion hours on national roads has continuously increased since the COVID-19 pandemic and was approximately twice as high in 2024 compared to pre-pandemic levels (Bundesamt für Strassen ASTRA, Fachbereich Verkehrsmanagement, 2025).

I would also request that the authors comment, even if briefly, on continuity and quality assurance related to drift corrections in the main text of the paper, as any potential instrument drifts or instrument changes over such a long stretch of time would have significant potential to bias results.

The instruments are calibrated and zero point corrected on a regular basis to ensure the quality of the long-time measurements. Drifts are monitored and corrected for. We have added some text in the manuscript to highlight this important point.

Lines 137 ff.: The instruments are zero point corrected every four weeks. The maximum four-week zero point drifts are ± 0.2 ppbv for NO_x and ± 0.3 ppbv for O_3. Span calibrations of the NO_x instruments are also performed every four weeks. For O_3, span calibration requires the use of a transfer photometer, which is deployed twice a year (in April and September) at each site. Drifts of the O_3 instruments are corrected when the response deviates more than ± 2% of the calibration gas concentration.

Line 112: please clarify what differentiates the background sites from the rural sites, as the current background definition states that the sites are rural? Is this low elevation rural sites vs high elevation rural sites? If so why was this differentiation selected?

Yes, the referee is correct that background sites are defined as rural sites above 1000m altitude (as stated on line 126). With a low elevation the rural sites are continuously located within the boundary layer and therefore represent localized photochemical processes at sites, which are negligibly impacted by anthropogenic pollution sources. Due to the elevation of the background sites, they capture local photochemistry but are also impacted by the free troposphere and can indicate if European background conditions have changed over time and might dominate the observed processes. We have revised the manuscript to clarify the differences between rural and background sites.

Lines 130 ff.: We differentiate between rural sites at low and high elevations to capture the local photochemistry at sites with negligible anthropogenic pollution versus conditions which are impacted both by local processes as well as free tropospheric impacts due to the influence of the residual layer.

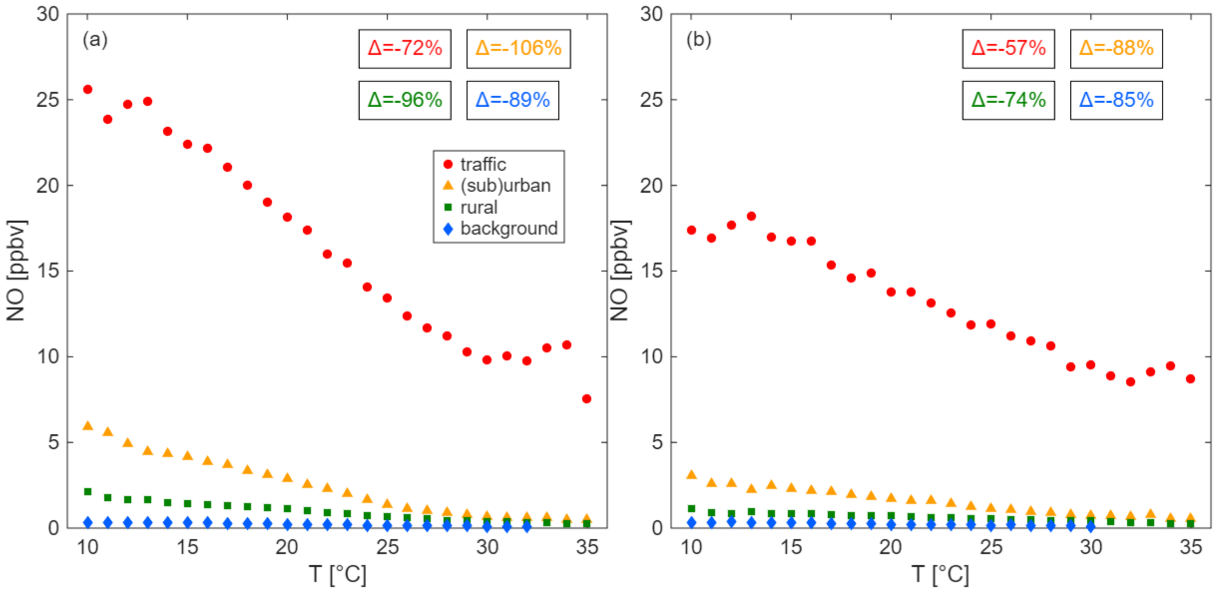
Line 118: Can you please comment on the potential interferences of VOC cross sensitivity for the UV-absorption method of ozone measurement? How were these either determined to be negligible or corrected for?

The interference from aromatic compounds, which possess absorption maxima near the 254 nm O_3 band, is known. The UV-absorption instruments used at the NABEL sites comply with the EN standard EN14625, which mandates a maximum interference of 1% for mixing ratios of 500 ppbv toluene and m-xylene. In Switzerland, total mixing ratios of BTEX (benzene, toluene, ethylbenzene and xylenes) and other aromatic compounds have remained well below 5 ppbv since

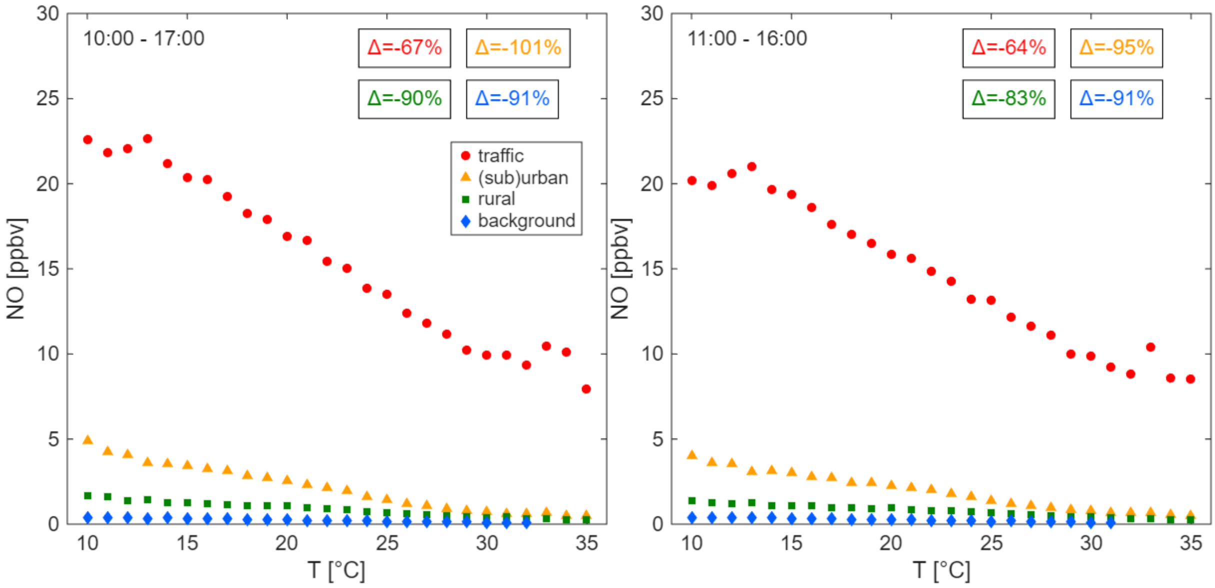
2000 (Le Bras et al., 2026, doi: 10.5194/acp-26-869-2026). Therefore, the interference from aromatic compounds is negligible.

Line 303: Please add additional evidence for the assertion that boundary layer height variability is minimized in the selected time area such that dilution is not the most likely explanation for the observed temperature effect on NOx. I understand the argument around hours after sunrise, but I would appreciate additional analysis to support ~2 hours after sunrise being sufficient to reduce the importance of this effect given the significant lag between light, and temperature and BLH and the importance of surface temperature in driving BLH dynamics.

We have investigated the NO-temperature correlation at 13:00-15:00 local time (radiation peaks at 14:00 local time), when we do not expect any significant diurnal changes in the boundary layer height, in comparison to the 09:00-18:00 data:

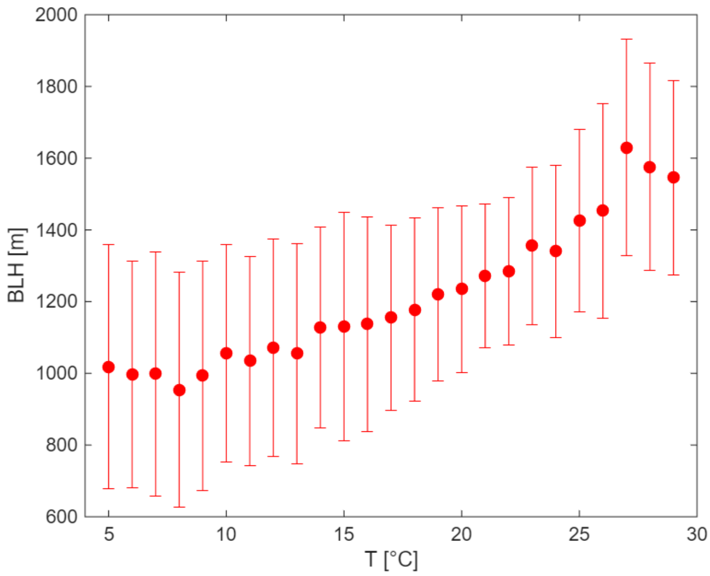


Panel (a) shows the 09:00-18:00 data and panel (b) presents the NO-temperature correlation for the 13:00-15:00 data. The figure shows that the temperature dependence of NO persists at all sites when eliminating the diurnal variability of the BLH. At the same time, the correlation decreases by around 15-20% (a bit less at background sites). While this could be an indication for the diurnal BLH variability affecting a part of the NO-temperature dependence, the 13:00-15:00 data eliminates NO emitted during the morning rush hour when temperatures are generally lower. Shortening the daily time window to ~3 and ~4 hours after sunrise and before sunset does not strongly affect the observed correlation (e.g. for traffic sites from -72% (09:00-18:00) to -67% (10:00-17:00) to -64% (11:00-16:00):



We therefore suggest that the chosen cut-off at 09:00 and 18:00 is a good trade-off between the diurnal BLH variability and the statistical robustness of our analysis.

Having said that, we will generally not be able to fully eliminate the impact of the BLH, as we expect a day-to-day BLH variability besides the diurnal BLH changes. We have investigated ERA5 reanalysis data of the BLH across Switzerland:



This figure shows the daily boundary layer height at 2pm binned to temperature (average across Switzerland). The resolution of the dataset is 0.25°x0.25°, which cannot resolve the topography of Switzerland. However, it can provide an estimation of the BLH-temperature correlation, which is strongly positive and solely results from the day-to-day variability.

We have added the NO-T correlation at 09:00-18:00 vs 13:00-15:00 and the BLH-T correlation to Figures S12 and S11 of the Supplement, respectively, and clarified these important aspects in the manuscript.

Lines 358 ff.: Figure S11 presents the BLH-temperature correlation across Switzerland, based on ERA5 reanalysis data of the daily summertime BLH and the 2m-temperature at 14:00 local time. While the resolution of the ERA5 data (0.25°x0.25°) is not sufficient to resolve the topography of Switzerland, it provides an estimation of the BLH-temperature correlation. The positive correlation supports our theory that dilution effects could impact the temperature correlation of trace gases and that the day-to-day variability of the BLH is significant.

and

Lines 362 ff.: We observe a 15-20 % decrease in the NO-temperature correlation when eliminating diurnal BLH changes (Figure S12). While this observation could highlight the impact of BLH variations throughout the day, it could also indicate that a part of the NO-temperature correlation results from NO emissions during the morning rush hour, which is usually accompanied by lower temperatures.