

JGR Atmospheres

RESEARCH ARTICLE

10.1029/2025JD045353

Key Points:

- Regional transport has a large impact on Fairbanks NO_x due to dominant role of ozone titration in loss of NO
- PM_{2.5} is affected by regional transport of existing particulate matter, but not precursors
- Chemical boundary conditions produce significant model-model differences, with implications for air quality attainment modeling

Supporting Information:

Supporting Information may be found in the online version of this article.

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Citation:

Hoffman, A., Arnold, S. R., Heard, D., Whalley, L., Brett, N., Law, K. S., et al. (2026). Regional transport in the Arctic: Sensitivity of NO_x and particulate matter in wintertime urban Alaska to background air. *Journal of Geophysical Research: Atmospheres*, 131, e2025JD045353. <https://doi.org/10.1029/2025JD045353>

Received 16 SEP 2025

Accepted 14 MAR 2026

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Regional Transport in the Arctic: Sensitivity of NO_x and Particulate Matter in Wintertime Urban Alaska to Background Air

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Abstract Fairbanks, Alaska is an example of a subarctic city that experiences severe wintertime air quality episodes characterized by high concentrations of fine particulate matter (PM_{2.5}) and nitrogen oxides (NO and NO₂ = NO_x). Previous research into these air quality episodes has focused on the impact of surface-based inversions that trap local emissions. However, the availability of ozone is a limiting factor in local wintertime pollutant transformation, and sensitivity of Fairbanks air quality to ozone availability has not been investigated on a regional scale. Here, we assess the impact of regional transport on ozone, NO_x, and particulate matter concentrations in Fairbanks, Alaska using three boundary conditions in the Community Multiscale Air Quality (CMAQ) model v5.4. We compared model results with measurements made during the ALPACA field campaign (model-observation comparison), and we assessed the differences between pairs of models (model-model comparison) using Euclidean distances of NO₂, SO₂, PM_{2.5}, and O₃. We find that modeling of NO_x in the Fairbanks area depends on, amongst other factors, accurate boundary conditions of background ozone because NO + O₃ titration is the dominant loss reaction of NO. PM_{2.5} in Fairbanks is affected by transport of existing background particulate matter rather than the transport of precursors. The boundary condition transport impacts metrics of air pollutants used for regulatory purposes throughout the model domain. Our results have important implications for accurate modeling of air quality in Arctic regions subject to strong local emissions and surface-based inversions, and have implications for assessment of air quality attainment status.

Plain Language Summary Air quality in the subarctic city of Fairbanks, Alaska is often unhealthy during the winter due to extreme cold weather causing the trapping of local pollution close to breathing level. Despite the dominance of local emissions in degrading air quality, we show that atmospheric transport of ozone and particulate matter over long distances can affect air pollution in Fairbanks. We use different model representations of background air to understand how much Fairbanks air quality is impacted by regional transport. We show that modeled air quality in downtown Fairbanks is sensitive to abundances of background pollutants that can be transported into the city. This is due to the dominant atmospheric chemical reactions that occur in the extreme cold and dark conditions of Arctic winter which is sensitive to background ozone. These findings have implications for air quality attainment modeling.

1. Introduction

Fairbanks, Alaska, is an Arctic urban region where wintertime meteorology and emissions combine to produce substantial enhancements in air pollutants. For example, wintertime fine particulate matter (PM_{2.5}) concentrations in Fairbanks regularly exceed the US Environmental Protection Agency (EPA) air quality standards (ADEC, 2022). In 2009, Fairbanks and surrounding communities were designated a non-attainment area due to frequent 24-hr PM_{2.5} exceedances (ADEC, 2022). The air quality events that lead to the non-attainment status in Fairbanks are mainly attributed to high local heating emissions and surface-based temperature inversions, which

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trap the local emissions close to the surface for extended periods (Cesler-Maloney et al., 2022; Tran & Mölders, 2011; Ye & Wang, 2020).

Surface-based inversions (SBIs) in Fairbanks are characterized by a stable surface layer caused by cold temperatures at the surface and warmer temperatures aloft, low wind speeds preventing mixing, high concentrations of NO_x (NO and NO_2) and particulate matter, and low concentrations of ozone relative to background levels (Cesler-Maloney et al., 2022; Malingowski et al., 2014; Mayfield & Fochesatto, 2013). The breakup of surface-based inversions results in decreased near-surface PM and NO_x but higher ozone, facilitated by mixing of background air from aloft (Simpson et al., 2024). During some inversion events, a shallow layer of background ozone can flow down from the surrounding hills and increase the concentration of ozone within the surface layer in the outlying areas of Fairbanks (Maillard et al., 2022; Pohorsky et al., 2025). Ozone during inversions is rapidly titrated by NO and contributes to high nitrate radical (NO_3) concentrations. In the low radiation winter conditions of Fairbanks, NO_3 may be important in driving oxidation of primary pollutants (Ketcherside et al., 2025). Thus, understanding of winter atmospheric chemistry in Fairbanks and other polluted Arctic urban regions depends on accurately accounting for background concentrations.

While previous analysis of air pollution events in the Arctic has focused on local emissions and temperature inversions, less work has been done to assess the impacts of regional transport and ambient background concentrations (Ioannidis et al., 2023, 2025). The air quality conditions in the Arctic are expected to experience rapid change over the coming decades with increased shipping and resource extraction, resulting in population growth to support these activities and new patterns of emissions (Aliabadi et al., 2015; Corbett et al., 2010; Peters et al., 2011; Schmale et al., 2018). It is therefore important to understand the contribution of both local emissions and regional transport of ambient concentrations in controlling air quality variability in the Arctic.

Chemical transport models are powerful tools for understanding the drivers of air quality conditions. Such models are often run on a limited domain (e.g., regional models) where only production and loss terms within the domain area are defined. Transport of pollutant concentrations beyond the domain edges must be provided to the model as boundary condition inputs (BCons) (Curci, 2012; Jiménez et al., 2007; Liu et al., 2001). In areas where transport of background air has a large impact on air quality, using representative and accurate BCons is important as different representations of transported concentrations into the model domain will affect predictions of pollution (Tong & Mauzerall, 2006; Tang et al., 2009; Song et al., 2008; Appel et al., 2007; D.-R. Choi & Koo, 2013). Previous analyses of BCons in models have focused on ozone and particulate matter because these species have longer lifetimes and tend to be impacted by regional transport (Curci, 2012; Samaali et al., 2009; Song et al., 2008). The transport of NO_x reservoir species, including peroxyacetyl nitrate (PAN), has also been assessed in modeling studies, although these studies have evaluated the impact on ozone rather than NO_x (e.g., Fischer et al., 2014; M. Choi & Ying, 2024). Less focus has been given to the impact of BCons on NO_x under the assumption that species with short lifetimes are minimally affected by boundary conditions in the inner model domain (Brost, 1988; Im et al., 2018; Liu et al., 2001).

Here, we evaluate the impact of regional transport on near-surface pollution in the Fairbanks urban area by comparing chemical transport model simulations using three different BCon representations of background concentrations. The goal of this work is to assess the impact that different regional background concentrations have on model simulations of wintertime air quality in Fairbanks, both for species with longer lifetimes that can be transported regionally and for species with short lifetimes that are primarily sourced from local emissions. The secondary goal of this work is to identify the model BCons that best capture pollutant transport. We perform model-model comparison (Section 3.1) to quantify the impact of the different BCons on air quality modeling. We then evaluate the model performance with each BCon by comparison with observations from the Alaskan Layered Pollution and Chemical Analysis (ALPACA) field campaign, which took place in January–February 2022 (Simpson et al., 2024). We compare the models with observations made at a sub-urban site (Section 3.2) to understand the influence of BCons on background concentrations. We compare the model against observations at sites within the city center (Section 3.3) to understand how the BCons impact model performance in polluted conditions. Finally, we assess the chemical and physical processes driving the differences in the BCon simulations to better understand drivers of pollution in Fairbanks (Section 4). Our results highlight the impact that regional transport can have even in remote areas in the Arctic and how boundary condition choice will affect the modeling of air quality in such locations.

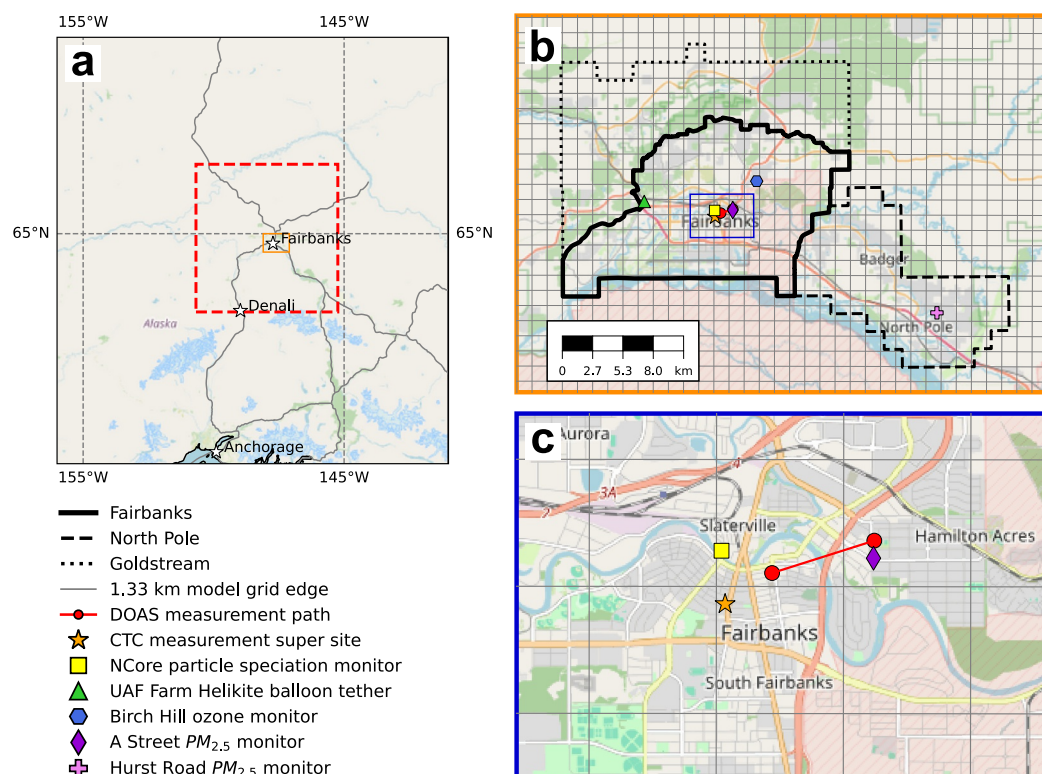


Figure 1. Study area over Alaska. Panel (a) the model domain is the outer red dashed line and the Fairbanks inset is marked with orange line. Panel (b) the Fairbanks North Star Borough (FNSB) non-attainment areas with the ALPACA measurement locations referenced in this study. The CMAQ 1.33 km horizontal grid is marked in gray lines. Panel (c) inset of downtown Fairbanks to show the distribution of ALPACA measurements in the city center. Map data from OpenStreetMap (openstreetmap.org/copyright). Distributed under the Open Data Commons Open Database License (ODbL) v1.0.

2. Methods

2.1. Study Area

We modeled air quality over Alaska during the 2022 winter using the U.S. EPA Community Multiscale Air Quality (CMAQ) model version 5.4. We ran the model with a 1.33 km horizontal resolution over central Alaska (Figure 1a) on a 199 by 199 horizontal grid and 38 vertical layers with a top of approximately 50 hPa. This model domain is the same as the U.S. EPA and Alaska Department of Environmental Conservation (ADEC) regulatory model set up over the Fairbanks North Star Borough (FNSB) non-attainment area (ADEC, 2022). The FNSB non-attainment area is in the center of the model domain (orange box in panel a, and panel b inset). There are few other regional sources of anthropogenic pollution nearby; there is a limited road network and no other major urban centers within the modeled area. Denali National Park and the associated Interagency Monitoring of PROtected Visual Environments (IMPROVE) monitoring station (U.S. EPA, 2024a) are on the edge of the model domain.

2.2. Representation of Background Concentrations

We used three BCon inputs to evaluate the sensitivity of air quality to the regional transport of background concentrations. In selecting the simulation outputs to derive our BCons, we wanted to use accessible and publicly-available datasets to be consistent with a typical research approach to generating BCons. Previous analyses of Fairbanks on the regulatory domain described in Section 2.1 have used northern hemisphere simulations from the U.S. EPA EQUATES project to derive BCons (Farrell et al., 2025; Huff et al., 2025). We downscaled two northern hemispheric (NHEMI) and one global simulation listed in Table 1 to generate BCons for the nested domain around central Alaska. The CMAQ 2019 NHEMI simulation came from the U.S. EPA EQUATES project database and was the latest year of data available in the database (Foley et al., 2023; U.S. EPA, 2021). The CMAQ 2022 NHEMI and GEOS-Chem 2022 global simulations are available from the U.S. EPA Amazon Web Services

Table 1
Parent Model Simulations Used to Derive Boundary Condition Inputs for Nested Fairbanks Domain

Model configuration						
Nested simulation	Base model	Domain	Horizontal resolution	Vertical layers (Top pressure)	Chemistry scheme	Time step
CMAQ19	CMAQ v5.3.2	Northern hemisphere (polar stereographic)	108 km × 108 km	44 (50 hPa)	cb6r3m_ae7_kmtbr	1 day
CMAQ22	CMAQ v5.4	Northern hemisphere (polar stereographic)	108 km × 108 km	44 (50 hPa)	cb6r5_ae7	1 hr
GC22	GEOS-Chem v14.0.1	Global	2° × 2.5°	72 (0.01 hPa)	Standard fullchem	3 hr
Inputs						
Nested simulation	Biogenic emissions	Fires	Anthropogenic emissions	Other emissions	Meteorology	Year
CMAQ19	CAMS v2.1 extension to MEGAN v2.1	SMARTFIRE2 and BlueSky Pipeline (US), FINN v1.5 (non-US)	2017 NEI (US), 2010 HTAP scaled with CEDS v2021.04.21 (non-US)	online dust and sea spray	WRF v4.1.1 nudged with 1° GFS	2019
CMAQ22	CAMS v2.1 extension to MEGAN v2.1	FINN v1.5	HTAP Phase 3	online dust and sea spray	WRF v4.4.2 nudged with 1° GFS	2022
GC22	MEGAN v2.1	monthly GFED	CEDS v2	online dust and sea spray	MERRA2	2022

(AWS) repository (U.S. EPA, 2024b). We used these simulations to derive our BCons because they represent the study year of interest. Our Fairbanks nested domain was covered by a subset of 3×2 grid cells in the GEOS-Chem simulation and a subset of 4×4 CMAQ NHEMI grid cells (Figure S1 in Supporting Information S1). Because a limited number of grid cells are used to generate the BCons, there is little horizontal variability along the domain edges from the boundary condition inputs. However, considering the remote nature of Fairbanks, there are few emission sources immediately outside the domain that would contribute to sharp gradients in pollutant concentrations along the boundaries. The CMAQ 2022 NHEMI simulation had an hourly temporal resolution, while the temporal resolution of GEOS-Chem data was every 3 hr and CMAQ 2019 NHEMI was daily. The hourly BCons for these two simulations were interpolated from this coarser temporal resolution. The CMAQ19 BCons therefore lack the typical diurnal cycle of some species, such as ozone, that will be present in the CMAQ 2022 BCons (Figure S2 in Supporting Information S1).

Hourly BCons from the CMAQ NHEMI simulations were calculated using the CMAQ preprocessing tool BCON. Initial conditions were calculated using the CMAQ preprocessing tool ICON. For the GEOS-Chem global simulation, the initial condition input and hourly BCons were calculated using the aqmbc Python package which maps GEOS-Chem version 14 chemical species to CMAQ chemical species in addition to regridding the larger model onto the nested domain (Henderson et al., 2014). We compared the 2019 and 2022 CMAQ NHEMI simulations to assess the impact of using different years for BCon inputs, and we compared the 2022 CMAQ NHEMI and the 2022 GEOS-Chem global to assess the impact of using different models for BCon inputs.

2.3. Model Configuration

The modeling period ran from 17 January to 24 February 2022, including 4 days of spin-up. We used aero7 aerosol chemistry (Appel et al., 2021), the Carbon Bond 6 revision 5 chemical mechanism (cb6r5_aero7) (Luecken et al., 2019; Yarwood et al., 2020), and in-line photolysis. Meteorological conditions in all simulations are the same. We ran CMAQ using offline meteorology, meaning that meteorology was prepared separately and there is no feedback between chemistry and meteorology. Meteorological inputs came from the Weather Research and Forecasting model (WRF) version 4.3 (Skamarock et al., 2008), which were developed for the ALPACA campaign period by the U.S. EPA (Brett, Law, et al., 2025; Gilliam et al., 2023). The WRF simulations were nudged with surface measurements of temperature, humidity, and wind made during the ALPACA campaign as well as local U.S. weather station sites, radiosonde and LiDAR measurements, and reanalysis data above 300 m from the National Centers for Environmental Prediction (NCEP) Global Forecast. Detailed information on the WRF physics schemes and observational nudging are presented in Brett, Law, et al. (2025) and Gilliam et al. (2023). Hourly anthropogenic emissions inputs included power plants, on-road and non-road vehicles, airports, and commercial and residential heating. These inputs were provided by the Alaska Department of Environmental Conservation (ADEC) and the power plant companies and processed for CMAQ as part of the ALPACA project (ADEC, 2019; Farrell et al., 2025; Huff et al., 2025). Detailed breakdown of the emissions by sector in the surface layer and aloft are presented in Brett, Law, et al. (2025); Brett, Arnold, et al. (2025); Albertin et al. (2024). Sea spray, windblown dust, and biogenic emissions were all assumed to be negligible during winter in central Alaska so these emission streams were turned off.

We conducted one sensitivity study with vehicle NO_x emissions enhanced based on temperature. Brett, Law, et al. (2025) suggested that a logarithmic dependence on temperature was needed to adjust the diesel vehicle emission inputs because the default emission was insufficient to explain observed NO_x concentrations. We applied the suggested temperature-dependent enhancement factor (EF) to the vehicle NO_x emissions (Equation 1) when temperature (T) was between -35°C and 0°C . Vehicle emissions in our inputs are split between on-road and non-road emissions with NO_x emission ratios from the MOVES2014b model (ADEC, 2019; Huff et al., 2025). We maintain the existing NO_x emission split from MOVES2014b (90% NO , 0.8% HONO , and 9.2% NO_2) (Kurtenbach et al., 2001) and increase each hourly emission by the temperature-based enhancement factor (Figures S3 and S4 in Supporting Information S1). We used the enhanced vehicle emissions with the CMAQ22 boundary condition inputs to test the impact of enhanced vehicle NO_x on model performance, called Vehicle NO_x adj..

$$EF = 10 - 8.9 \times (1 - e^{-0.1(T+36)}) \quad (1)$$

2.4. Observational Data

We compared output from the Fairbanks nested simulations against observations made during the ALPACA campaign to determine which BCon resulted in better model-observation agreement. The ALPACA campaign took place from 17 January to 25 February 2022 in Fairbanks, Alaska. Figure 1b shows the locations of several ALPACA sampling locations relative to the CMAQ grid. We compare model output of ozone, NO, NO₂, and PM₁ to surface measurements made at the CTC site (orange star); ozone and NO₂ measured along the long path differential optical absorption spectrometer (LP-DOAS, hereafter DOAS) path (red line); and ozone measured at Birch Hill (blue hexagon) (Cesler-Maloney et al., 2022; Kuhn et al., 2025; Simpson et al., 2023, 2024). We compare against daily PM_{2.5} measurements at the NCore site (yellow square), the A Street monitor (purple diamond), and the Hurst Road monitor in North Pole (pink cross), which are nearby permanent particle speciation monitors operated by the ADEC for the EPA. In addition to surface measurements, we compared with vertical profiles of O₃ and NO₂ made at the University of Alaska—Fairbanks (UAF) Farm site (green triangle) by the MICROMEAS instrument onboard the tethered balloon (helikite) (Barret et al., 2024, 2025; Pohorsky et al., 2025). The measurements of ozone, NO, and NO₂ at the CTC site and the ozone at the Birch Hill site are all made on a minute basis, while the PM₁ measurements at the CTC site were made every 2 minutes. These observations were averaged hourly to compare against CMAQ hourly average (ACONC) output. The LP-DOAS measurements had an approximately 20–30 min sampling frequency and the MICROMEAS data was reported on a 15 min frequency, so these data were also averaged hourly to compare with CMAQ ACONC. The PM_{2.5} measurements were reported as daily averages, so we calculated the daily average of the CMAQ hourly output. A full list of all measurement locations and instruments from the ALPACA project is available in Simpson et al. (2024) and performance in the unique, cold environment of Fairbanks for many of the instruments has been assessed previously (Barret et al., 2025; Cesler-Maloney et al., 2022; Kuhn et al., 2025).

2.5. Data Analysis

To evaluate the model-observation agreement, we used the standard performance metrics of normalized mean error (NME), normalized mean bias (NMB), coefficient of determination (R^2), and root mean square error (RMSE) (Equations S1–S4 in Supporting Information S1) (Simon et al., 2012). We made model-model comparisons of relevant pollutants aligned at the measurement locations ($N = 816$ for each simulation) using an independent t -test to compare means of the model data. The model samples at each measurement location are not normally distributed (Figure S5 in Supporting Information S1) but the samples sizes are large enough that an independent t -test is valid (Kim & Park, 2019; Kwak & Kim, 2017). Because of the nonparametric distribution of the data, we also use a Mann-Whitney U test to compare the rank distributions of the data. Both tests were two-tailed with a confidence level of 95%. We only report two models as being statistically significantly different if the null hypothesis is rejected for both the independent t -test and the Mann-Whitney U test. Both statistical tests are calculated using the scipy.stats v1.14 Python package (Virtanen et al., 2020).

To quantify model-model differences, we calculated Euclidean distances between modeled SO₂, NO₂, PM_{2.5}, and O₃. Euclidean distance calculates the shortest distance between two points in a multi-dimensional space, so it is a minimum of the difference between model simulations. We calculated the daily maximum hourly average SO₂, NO₂, and PM_{2.5} and the daily maximum 8-hr rolling average ozone for the entire model period, including the spin-up time. These pollutant statistics are the U.S. EPA National Ambient Air Quality Standards (NAAQS) primary standards used in determining the air quality of a location. We used the NAAQS standards to demonstrate how BCons could impact the model's ability to simulate air quality as it would be quantified for determining attainment status of each pollutant (U.S. EPA, 2014).

For each pair of simulations, we converted the daily NAAQS value of a pollutant to a Z-score (Equation S5 in Supporting Information S1). The Z-score normalizes the population of each set of pollutant concentrations, which prevents weighting the distance calculation by pollutants with higher ambient concentrations and to avoid comparing pollutants of different units (ppb vs. $\mu\text{g}/\text{m}^3$). We calculated the Euclidean distance at each grid point and each day following Equation 2, where p is the location in Euclidean space of the Z-score at each grid cell and day of simulation 1, and q is the same for simulation 2.

$$d(p, q) = \sqrt{(p_{\text{SO}_2} - q_{\text{SO}_2})^2 + (p_{\text{NO}_2} - q_{\text{NO}_2})^2 + (p_{\text{O}_3} - q_{\text{O}_3})^2 + (p_{\text{PM}_{2.5}} - q_{\text{PM}_{2.5}})^2} \quad (2)$$

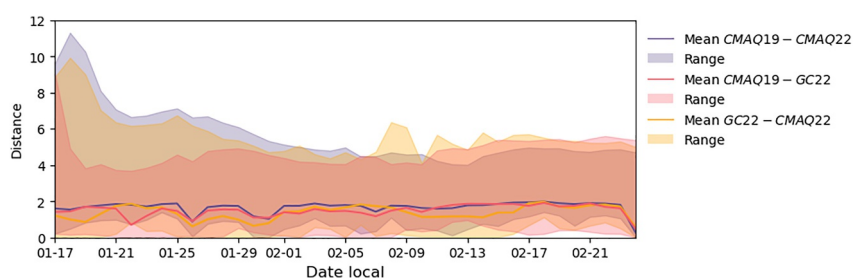


Figure 2. Time series of Euclidean distances between each pair of model simulations. The solid lines are the average distances between each pair of simulations and the shaded areas are the range (minimum and maximum distances of the entire domain). Larger distances indicate that the combined NAAQS distributions are more different between model simulations, and smaller distances mean that the combined NAAQS distributions are more similar in the two model simulations.

3. Results and Discussion

3.1. Assessment of BCon Impact Based on Model-Model Comparison

Previous research on BCons has focused on long-lived species such as ozone under the assumption that short lived species will not be impacted except at the edges of the model domain (e.g., Liu et al., 2001). We address this assumption by comparing NO_x in addition to ozone and PM at ALPACA measurement sites around FNSB, which is in the center of the model domain.

Model-model comparisons reveal some pollutants are more impacted by the BCons than other pollutants (Tables S1–S5 in Supporting Information S1). Ozone concentrations at the CTC and Birch Hill sites and along the LP-DOAS path are statistically significantly different between all three simulations at a 95% confidence level. NO at the CTC site is also statistically significantly different between all three simulations at a 95% confidence level. Only the CMAQ19-CMAQ22 NO_2 concentrations at the CTC site are statistically significantly different. The comparisons of CMAQ19-GC22 and CMAQ22-GC22 failed to reject the null hypothesis, as do all model-model NO_2 comparisons along the LP-DOAS path. GC22 $\text{PM}_{2.5}$ is statistically significantly different from both CMAQ simulations at NCore, Hurst Road, and A-Street, but the two CMAQ simulations are not statistically different from one another at these sites. Only the CMAQ19 and GC22 simulations PM_{10} at CTC are statistically significantly different from one another.

To evaluate the magnitude of BCon influence across the entire domain, we calculated Euclidean distances between pairs of model Z-scores for the daily U.S. EPA NAAQS metrics, which are relevant for air quality regulation and attainment. The three distances for each pair of model simulations in Figure 2 follow a similar temporal pattern. The greatest maximum distance between each pair occurs near the start of the model simulation, when concentrations are based on distinct initial conditions calculated from the same model outputs used to derive the BCons. The maximum distance between each pair of simulations decreases over the course of the first three to four model days and is then stable throughout the remainder of the simulation period. This demonstrates that our 4 day spin-up is adequate for allowing the model processes to overcome biases introduced from the initial conditions. The domain-wide average distance between each pair of simulations is stable, fluctuating between 1 and 2 throughout the simulation period. There are several local minima in average distance, but average distance never goes to zero except at the end of the simulation. Some pairs of simulations have a minimum distance of zero at points during the simulation period, with CMAQ19-GC22 minimum distances being the lowest on average and CMAQ19-CMAQ22 minimum distances being the highest on average.

The impact of BCons on the model simulation depends on location within the domain, local topography, and meteorology (Liu et al., 2001; Yan et al., 2021). It is often assumed that areas far from the boundary are less impacted except during transport of exceptional pollution events such as dust or wildfire smoke (Curci, 2012; Jiménez et al., 2007; Samaali et al., 2009). Yet our statistics show the BCons are affecting each simulation across the entire domain, even grid cells far from the boundaries. Given the small size of the domain, it is expected that there would be transport from the boundaries to the center of the domain; what is surprising is the extent to which short lived species such as NO in the center of the domain are impacted. NO has a short atmospheric lifetime on the order of minutes to hours even in cold and dark wintertime conditions (Kenagy et al., 2018; Seinfeld &

Pandis, 2016). Given the short lifetime, it is unlikely that transport of NO from the boundary is causing the difference in NO concentration at the CTC site. Instead, we hypothesize that differences between the model simulations of NO concentration are due to the differences between BCon inputs of ozone because of the ozone titration reaction. This will be explored in Section 4.1.

There are several possible reasons for the difference between GC22 PM_{2.5} and the two CMAQ BCon simulations, including the existing PM_{2.5} concentrations in the global GEOS-Chem simulation, the concentrations of PM_{2.5} precursor species such as SO₂ and VOCs in the global GEOS-Chem simulation, a discrepancy caused by mapping of GEOS-Chem particle species and gas-phase aerosol precursors to the analogous species in the CMAQ boundary condition files (Henderson et al., 2014), or different transport events simulated in the GEOS-Chem base simulation due to the difference in resolution between global and hemispheric. The processes affecting PM in will be explored in Section 4.2.

3.2. Comparison Against Background Measurements

Here, we assess the ability of the three BCon simulations to capture background concentrations away from the localized emission sources in FNSB. We compare our three nested BCon model simulations with measurements from the UAF Farm site. The UAF Farm is a sub-urban site outside of downtown Fairbanks. The Farm measurements are cleaner and more representative of regional background concentrations (Barret et al., 2025; Pohorsky et al., 2025) because the site is further from emission sources and experiences valley flow which introduces surface-level background air to the site (Brett, Law, et al., 2025; Fochesatto et al., 2026; Maillard et al., 2022; Pohorsky et al., 2025). Comparison of the model vertical profiles with the tethered balloon (helikite) measurements during weakly stable or unstable events demonstrates how well the model captures background conditions.

We focus on two balloon flights conducted during periods without a surface-based inversion (Figure 3). Measurements for Profile 1 (top row) started on 8 February, 2022 at 22:00 local time (AKST) and ended on 9 February, at 03:00 AKST and measurements for Profile 2 (bottom row) occurred on 21 February, from 13:00–15:00 AKST. Analysis of the temperature profiles measured onboard the helicite indicate that there were no surface-based inversions during the flights (Pohorsky et al., 2025). Temperature profiles from radiosondes launched at the Fairbanks International Airport during or shortly after the flights reveal elevated inversions between 500 and 900 m (Figure S7 in Supporting Information S1).

In both profiles, CMAQ22 has the highest background ozone concentration and CMAQ19 has the lowest. CMAQ22 has the best agreement with observations during Profile 1, although all simulations underpredicted ozone concentrations in this profile. During this balloon launch, the Farm site experienced increased ozone at the surface due to valley flow (Pohorsky et al., 2025). The ALPACA WRF meteorology used in our simulations underestimate the valley flow of background air to the Farm site (Brett, Law, et al., 2025), which partially explains the underestimated ozone in the model simulations at the surface. In Profile 2, observed ozone concentration is lower at the surface (26.7 ppb) and CMAQ19 has better agreement in the lowest layer. Moving aloft, the observed average concentration increased to 30 ppb and GC22 had better agreement until the highest layer in the profile. For NO₂, there is little difference between the simulations within a profile and all three simulations demonstrate a greater difference in NO₂ concentration with altitude in Profile 1 than Profile 2. The elevated levels of modeled NO₂ during both flights indicate that there is influence from anthropogenic emissions despite the distance from the city center. The NO_x concentrations in all three simulations are driven by the same emission inputs, resulting in the minimal variability between simulations in Figure 3.

To account for any loss in the model of background O₃ due to the titration reaction with excess NO, we examine the sum of O₃ and NO₂, termed “O_x.” Average observed and modeled O_x is stable with height in both flights. In Profile 1, the observed average O_x is 41 ppb throughout the altitude range, and in Profile 2 the observed average is 39 ppb. CMAQ22 has the highest O_x and CMAQ19 has the lowest, which is the same distribution of the models for O₃ concentrations. Comparison of model O_x with observations reveals that the underestimates in model O₃ are accounted for by excess NO₂, likely formed through the titration reaction with NO. For Profile 1, CMAQ22 has the best agreement with observed O_x. For Profile 2, GC22 has better agreement with the observations, although CMAQ22 overestimates by only 1–2 ppb. Based on the Farm analysis, BCons of the correct year (i.e., 2022) capture the background concentrations of O₃ better than 2019.

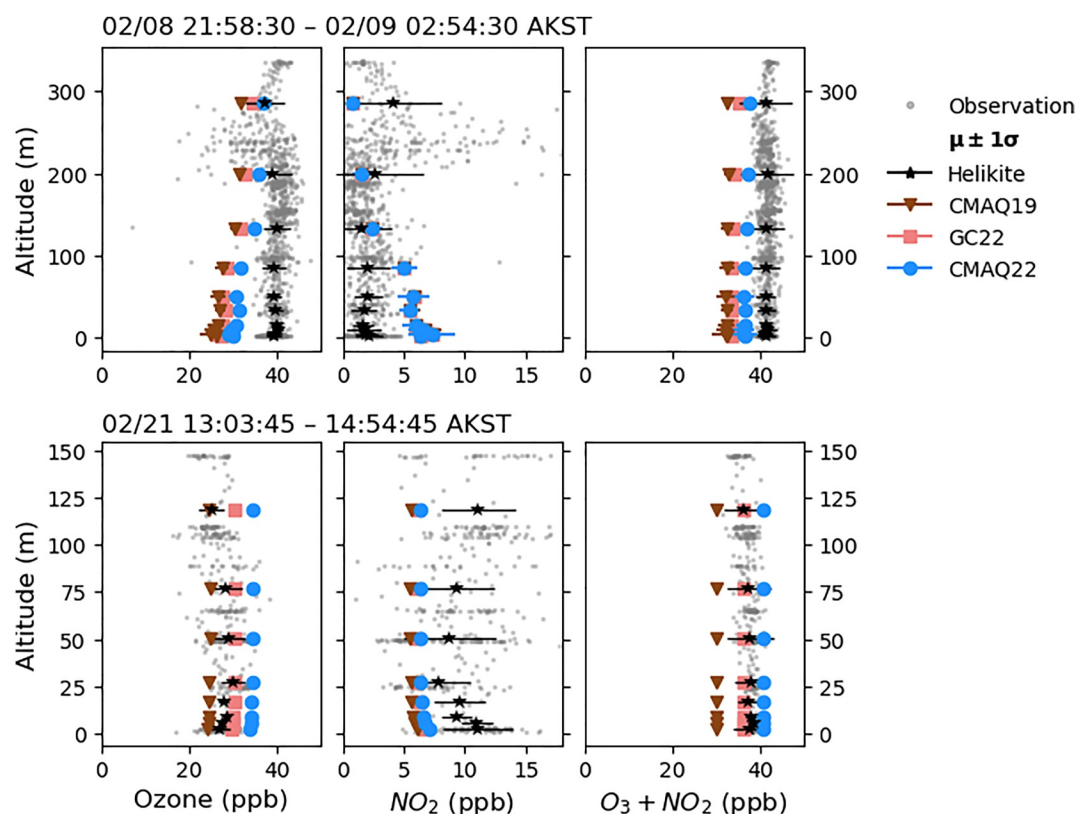


Figure 3. Vertical profiles of ozone (left), NO_2 (middle), and O_3 (right) from model simulations and the MICROMEAS instrument onboard the helikite. The top row is for the balloon flight starting on 8 Feb. at 22:00 (Profile 1) and the bottom row is for the balloon flight starting on 9 February at 03:00 (Profile 2). Individual balloon measurements made at 15 min intervals are marked in gray circles. The model layer mean of the observations is a black star, CMAQ19 simulation is marked with brown inverted triangle, GC22 simulation is marked with pink square, and CMAQ22 is marked with blue circle. The horizontal lines represent one standard deviation in each layer.

3.3. Assessment of BCon Impact on Model Performance

The statistics in Tables 2 and 3 show model performance compared to observations for the three BCon simulations. Model performance for ozone varies by measurement location and is overall moderate. The CMAQ22 ozone has the worst performance metrics (largest NME and RMSE) at all three sites with a tendency for the greatest overestimation compared to observations (highest NMB). Model performance for NO is poor at the CTC site, with all three simulations underestimating NO concentration (negative NMB). There is also a lack of variability in the simulated NO concentrations, with standard deviations less than 5.5 ppb. Observed NO at the CTC site has a large standard deviation ($\sigma_{\text{obs}} = 38.90 \text{ ppb}$), meaning that there is wide range of NO concentrations in observations, which is not captured in the models. The models also perform poorly for NO_2 at the CTC site and along the DOAS path, with a tendency toward underprediction (negative NMB) and large error statistics.

The tendency toward ozone overprediction and NO_x underprediction in the models is evident in Figure 4. All simulations overpredict ozone concentrations when observed concentrations are the lowest, for example, from 1 to 3 February at the CTC site and along the DOAS path, both of which are downtown locations. The CMAQ22 simulation has the largest overprediction during these times. During periods of high observed ozone, such as the two spikes on 24 and 25 January, CMAQ22 has the best agreement with observations. The observed ozone peaks occur during clean air events, when background air moves through Fairbanks (Simpson et al., 2024), which indicates that the CMAQ22 boundary conditions are best able to represent background ozone concentrations. In contrast, CMAQ19 has the lowest ozone during these periods and GC22 is between the two CMAQ simulations, consistent with the order of simulated ozone concentration magnitude at the Farm site shown in Figure 3. The CMAQ19 BCons are from 2019 and therefore do not capture background ozone concentrations for the specific period of interest. In addition, the BCons for CMAQ19 were derived from daily average output rather than hourly

Table 2
Performance Metrics of Model Hourly O_3 , NO , and NO_2

Pollutant	Model	σ (ppb)	R^2	NMB (%)	NME (%)	RMSE (ppb)
$O_{3,CTC}$ ($\sigma_{obs} = 12.14ppb$)	CMAQ22	9.33	0.21	82.9	103.0	11.5
	CMAQ19	7.93	0.26	32.6	81.7	10.6
	GC22	8.48	0.25	53.2	88.4	10.8
$O_{3,BirchHill}$ ($\sigma_{obs} = 6.01ppb$)	CMAQ22	6.91	0.14	15.3	23.5	8.46
	CMAQ19	5.62	0.16	−9.90	18.6	6.95
	GC22	5.62	0.18	0.31	15.8	6.28
$O_{3,DOAS}$ ($\sigma_{obs} = 10.0ppb$)	CMAQ22	10.2	0.20	39.1	57.7	12.6
	CMAQ19	8.69	0.22	3.26	45.3	9.73
	GC22	9.06	0.23	17.8	47.0	10.3
NO_{CTC} ($\sigma_{obs} = 38.90ppb$)	CMAQ22	4.51	0.12	−84.6	89.2	37.6
	CMAQ19	5.51	0.10	−80.2	87.6	37.5
	GC22	4.92	0.11	−82.3	88.3	37.6
$NO_{2,CTC}$ ($\sigma_{obs} = 14.83ppb$)	CMAQ22	8.27	0.19	−36.4	49.4	13.5
	CMAQ19	7.31	0.20	−40.5	51.3	13.3
	GC22	7.82	0.19	−38.4	50.2	13.4
$NO_{2,DOAS}$ ($\sigma_{obs} = 15.4ppb$)	CMAQ22	9.12	0.12	−47.7	58.8	19.9
	CMAQ19	8.20	0.12	−50.3	59.9	20.1
	GC22	8.73	0.12	−48.8	59.2	20.0

and GC22 were every three hours, so concentrations during peak times may be further underestimated by using these daily or three hourly averages.

When simulated ozone concentrations are overestimated compared to observations, the underestimation in simulated NO_x concentrations increases. The period of greatest model NO_x underprediction occurs between 1 February and 4 February, when modeled ozone is elevated compared to observations at all sites. This period is during a strong surface-based inversion event with the coldest surface temperatures observed during ALPACA (Brett, Law, et al., 2025; Fochesatto et al., 2026; Simpson et al., 2024). During this inversion, and other SBI events, build-up of NO_x emissions means observed NO concentration reaches over 100 ppb. In comparison, BCon simulation concentrations do not exceed 35 ppb during the same period. Robinson et al. (2023) used mobile sampling to show that pollutants in Fairbanks had high spatial and temporal variability, sometimes on the sub-

Table 3
Performance Metrics of Model Hourly PM_{10} and Daily $PM_{2.5}$

Pollutant	Model	σ ($\mu g/m^3$)	R^2	NMB (%)	NME (%)	RMSE ($\mu g/m^3$)
$PM_{10,CTC}$ ($\sigma_{obs} = 11.4 \mu g/m^3$)	CMAQ22	6.09	0.26	−20.2	53.0	10.1
	CMAQ19	5.67	0.26	−23.7	53.5	10.3
	GC22	6.43	0.24	−17.1	54.5	10.2
$PM_{2.5,NCore}$ ($\sigma_{obs} = 8.57 \mu g/m^3$)	CMAQ22	5.61	0.41	7.4	43.2	6.5
	CMAQ19	5.29	0.39	5.1	43.5	6.6
	GC22	7.97	0.20	44.5	66.8	8.6
$PM_{2.5,HurstRd}$ ($\sigma_{obs} = 19.8 \mu g/m^3$)	CMAQ22	10.1	0.34	−35.4	67.9	17.3
	CMAQ19	9.77	0.34	−36.8	67.6	17.3
	GC22	11.5	0.40	−22.6	65.0	15.9
$PM_{2.5,AStreet}$ ($\sigma_{obs} = 11.7 \mu g/m^3$)	CMAQ22	6.16	0.24	−41.9	56.4	12.3
	CMAQ19	5.71	0.23	−43.3	57.4	12.4
	GC22	8.94	0.33	−18.3	46.7	10.3

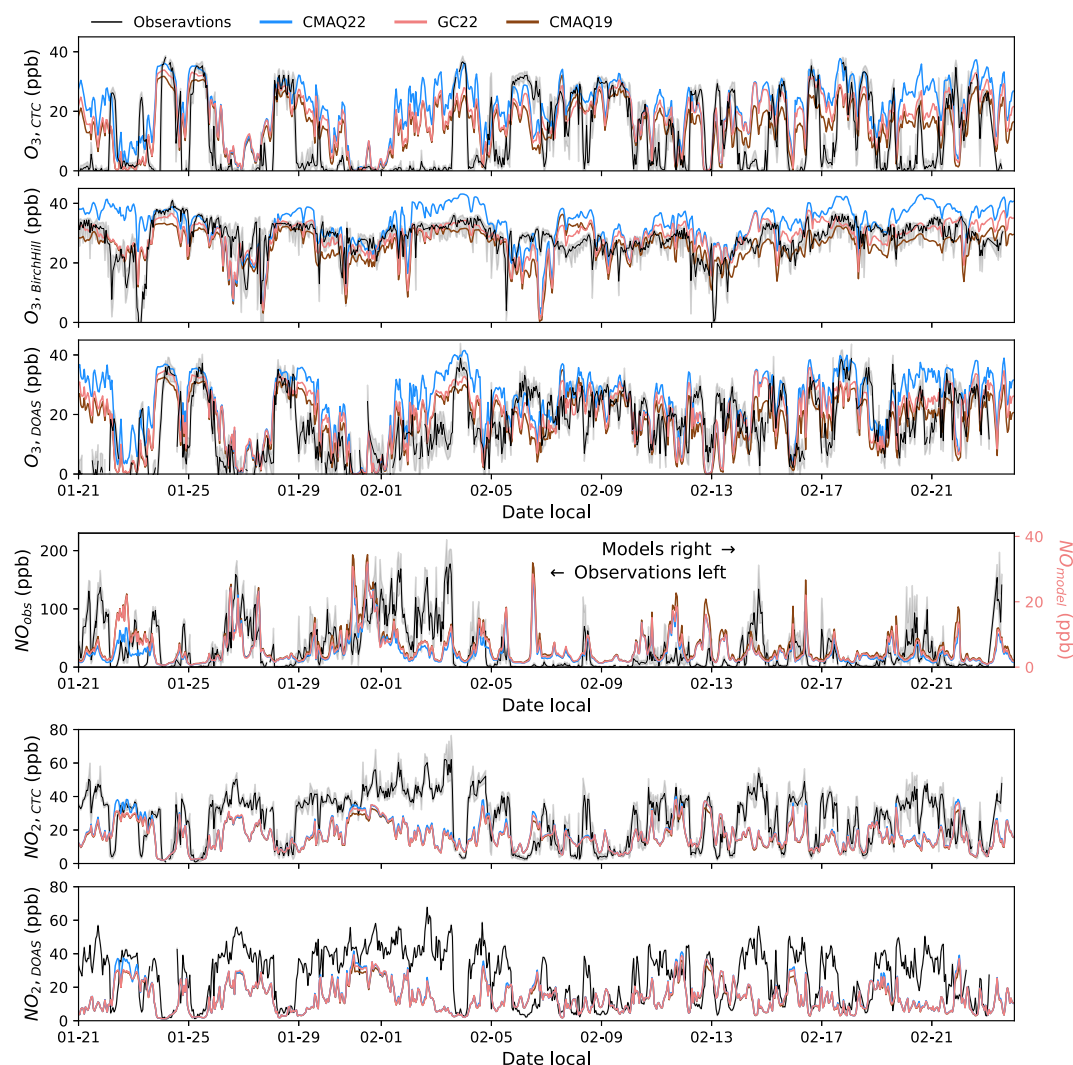


Figure 4. Time series of hourly BCon simulations and observations of ozone, NO, and NO₂. The location of each measurement is included in the y-axis label. Observations are the black line in the time series. CMAQ22 model results are the blue lines, CMAQ19 are brown lines, and GC22 are pink lines. The NO model results are plotted on the right-hand y-axis because the concentrations are much lower than observations. The gray shading around observations is the 10th and 90th percentile of the hourly observations for the CTC and Birch Hill measurements and the hourly instrument error or the DOAS measurements.

neighborhood scale, and that the presence of strong temperature inversions enhance this spatial variability. The underestimation in concentration and small standard deviation in modeled NO_x concentrations may be in part due to the difficulty in capturing such fine-scale variability in factors such as emissions and meteorology within the 1.33 km horizontal resolution of the model.

The performance metrics in Table 3 show that the three model simulations perform moderately well at capturing the temporal trends in PM₁ concentration at the CTC site, although the models are all biased low ($-23.7 \leq NMB \leq -17.1$). In comparison, the simulations tend to overpredict PM_{2.5} concentration, especially GC22. GC22 has a higher NME, NMB, and RMSE than CMAQ22 or CMAQ19 at NCore, but similar or better performance at Hurst Road and A Street monitors. GC22 simulation has a tendency toward higher PM_{2.5} concentrations, which results in better agreement at sites with high pollution levels. At A Street, located in downtown Fairbanks, GC22 has drastically better performance than the CMAQ simulations, as evidenced by the much smaller NMB and RMSE. At Hurst Road, GC22 is less different from the other two simulations. The Hurst Road side is far from downtown Fairbanks, but is within the North Pole non-attainment zone which also experiences

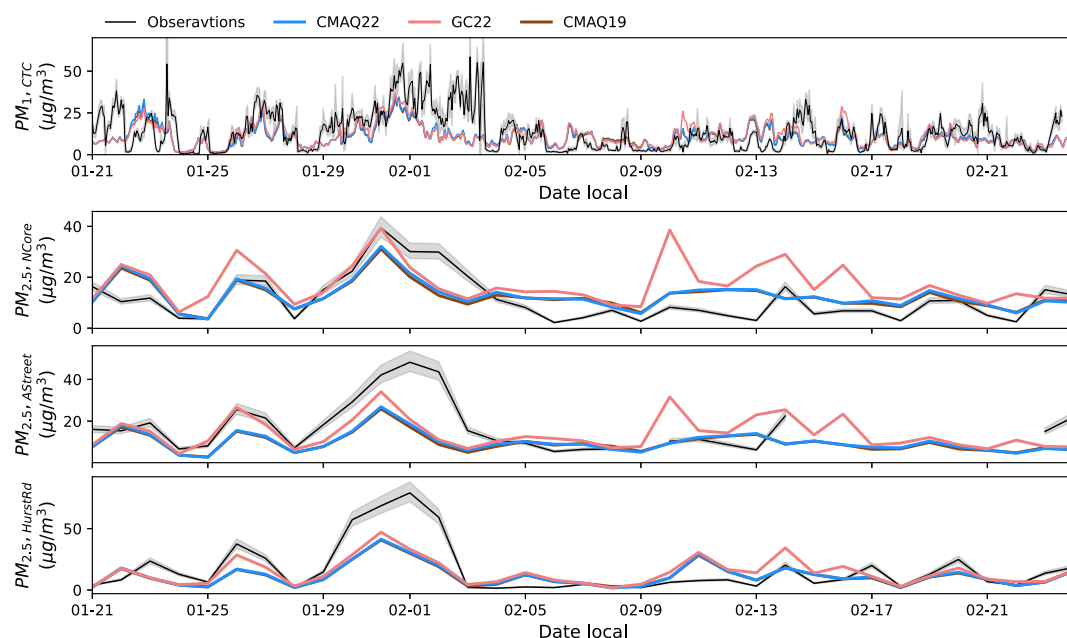


Figure 5. Time series of hourly PM_1 and daily $PM_{2.5}$ mass concentrations. Observations are the black line, CMAQ22 model results are the blue lines, CMAQ19 are brown lines, and GC22 are pink lines. The gray shading around observations is the 10th and 90th percentile of the hourly observations for the CTC PM_1 measurements, and $\pm 10\%$ of $PM_{2.5}$ measurements for EPA particle monitors following the Federal Equivalence Method bias tolerance (U.S. EPA, 2020). The A Street particle monitor is missing several days of data in mid to late February.

high $PM_{2.5}$ pollution events during SBIs due to local emissions. Because of the high local emission in North Pole, there may be a smaller relative impact of the different boundary conditions in the models.

GC22 $PM_{2.5}$ concentration in Figure 5 does not follow the temporal pattern of the CMAQ simulations later in the campaign period at the NCore or A Street monitors. From 9 February to 17 February, GC22 $PM_{2.5}$ is 10–20 $\mu g/m^3$ higher than observations with daily peaks not seen in CMAQ19 or CMAQ22. Brett, Law, et al. (2025) showed that the period of 9–17 February had mixed cyclonic and anticyclonic meteorological conditions, resulting in short-lived stable conditions mixed with weakly stable or unstable conditions that allowed for some transport of background air. The differences in PM may therefore come in the transport of background air. This will be explored more in Section 4.2.

3.4. Influence of Vehicle NO_x Emissions

To examine the impact of the underestimated NO_x emissions on model performance, we conducted a sensitivity study using the CMAQ22 BCon and the temperature-enhanced vehicle emissions described in Equation 1. We used the CMAQ22 BCon because these inputs had the best agreement with background ozone concentration at UAF Farm site. We compare CMAQ22 and Vehicle NO_x adj. with measurements from the CTC site, as this is the only site with co-located ozone, NO , and NO_2 . We focused the analysis during the strong surface-based inversion event in Figure 6, when the models had the greatest underestimate in NO_x concentrations at the CTC site. The event started on 29 January and lasted through midday on 3 February; observed ozone is close to zero and NO_x is elevated (Simpson et al., 2024). In addition to NO_x and ozone, we compare with observed O_3 . In the clean air at the UAF Farm site, comparison of O_3 revealed that underestimates in ozone were accounted for by elevated NO_2 in the models.

When more NO_x is added to vehicle emissions at low temperatures, model-observation agreement of NO and NO_2 improves (Table S8 in Supporting Information S1). Model NO concentration in the Vehicle NO_x adj. simulation peaks during this inversion at 147 ppb, and NO_2 has a higher and more sustained concentration. Modeled O_3 also improves. The periods of highest observed O_3 occur during the inversion event when ozone is lost through titration with NO and there is a build-up of NO_x (Brett, Law, et al., 2025; Pohorsky et al., 2025; Simpson et al., 2024). The Vehicle NO_x adj. simulation has better agreement with the observed O_3 peaks when O_3

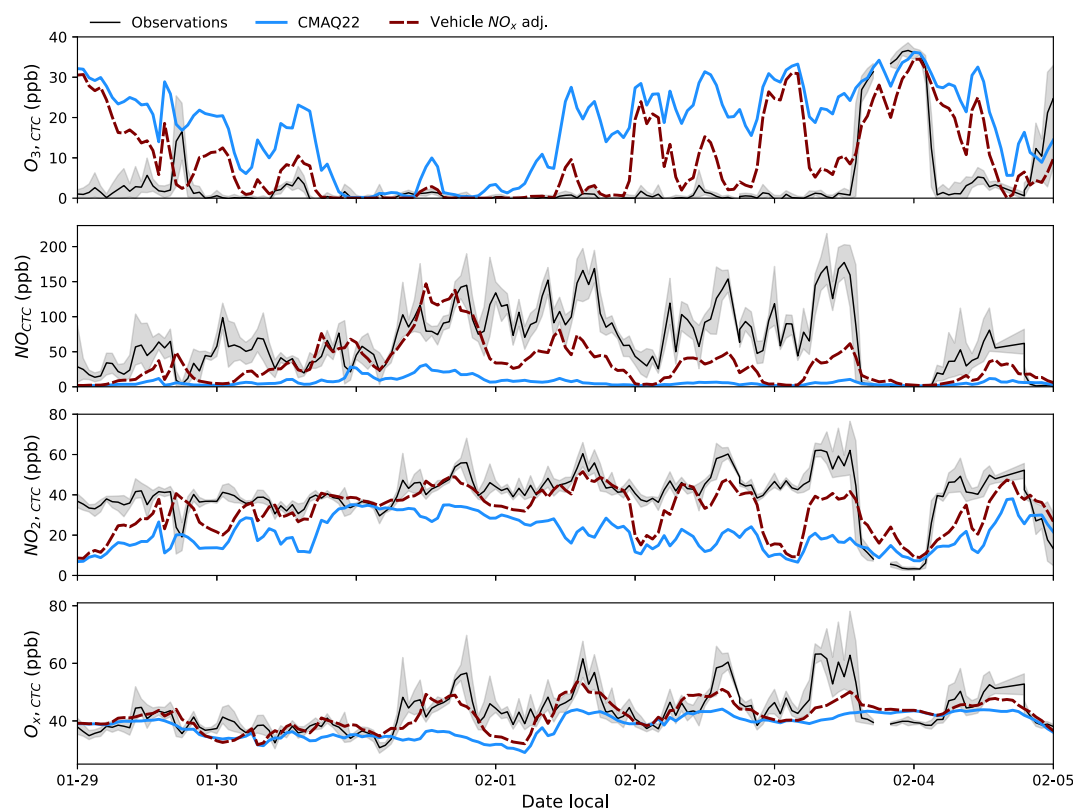


Figure 6. Time series of hourly observed and modeled O_3 (top), NO (second row), NO_2 third row, and O_x (bottom) concentrations at the CTC site from 29 January to 5 February, during which a strong surface-based inversion occurred. The CMAQ22 base simulation is in blue and the Vehicle NO_x adj. simulation is the red dashed line. Observations are the black line, and the gray shading is the 10th and 90th percentile of the observations each hour.

concentration is over 45 ppb, well above the highest ozone concentrations experienced in the clean, background air. However, starting on 1 February, NO and NO_2 concentrations in the Vehicle NO_x adj. simulation decrease drastically at night and O_3 simultaneously increases. The WRF meteorology simulated weaker vertical temperature gradients than seen in observations at the end of the SBI event in early February (Brett, Law, et al., 2025), so increased mixing and weaker stability may be impacting the model's ability to capture trends after 1 February. In addition, the peak O_x concentrations near the end of the inversion are still underestimated in the Vehicle NO_x adj. simulation, indicating there are additional missing emissions. Vehicle emissions are highest during the day, so these dips in NO_x concentrations could represent periods with an underestimated or missing nighttime emission source. Albertin et al. (2024) showed that burning of home heating oil is also a large contributor to surface NO_x emissions in Fairbanks during ALPACA. This emission source would be prevalent at night, so more work should be done to understand whether home heating oil is accurately represented in the model emission inputs.

4. Processes Affecting Pollutant Concentrations

CMAQ has an optional tool called “process analysis” that can be used to calculate the model-simulated change in concentration due to physical processes or chemical reactions (Gibson, 1999). The CMAQ process analysis has two components, both of which are calculated on each internal model time step and for each individual grid cell but summed over the output time step and spatial distribution selected by the user. The integrated process rate analysis (IPR) calculates the change in concentration due to physical processes and the net of chemical reactions; the integrated reaction rate analysis (IRR) calculates the change in concentration due to user-specified reactions in the chemical mechanism (Gibson, 1999). Both methods work by splitting the continuity equation that drives the model into a series of partial or ordinary differential equations for the user-selected processes or reactions and calculating the individual change in concentration (Jang et al., 1995; Jeffries & Tonnesen, 1994). We used the IPR for ozone, NO, NO_2 , and particulate matter as well as the IRR for NO loss reactions to better understand the

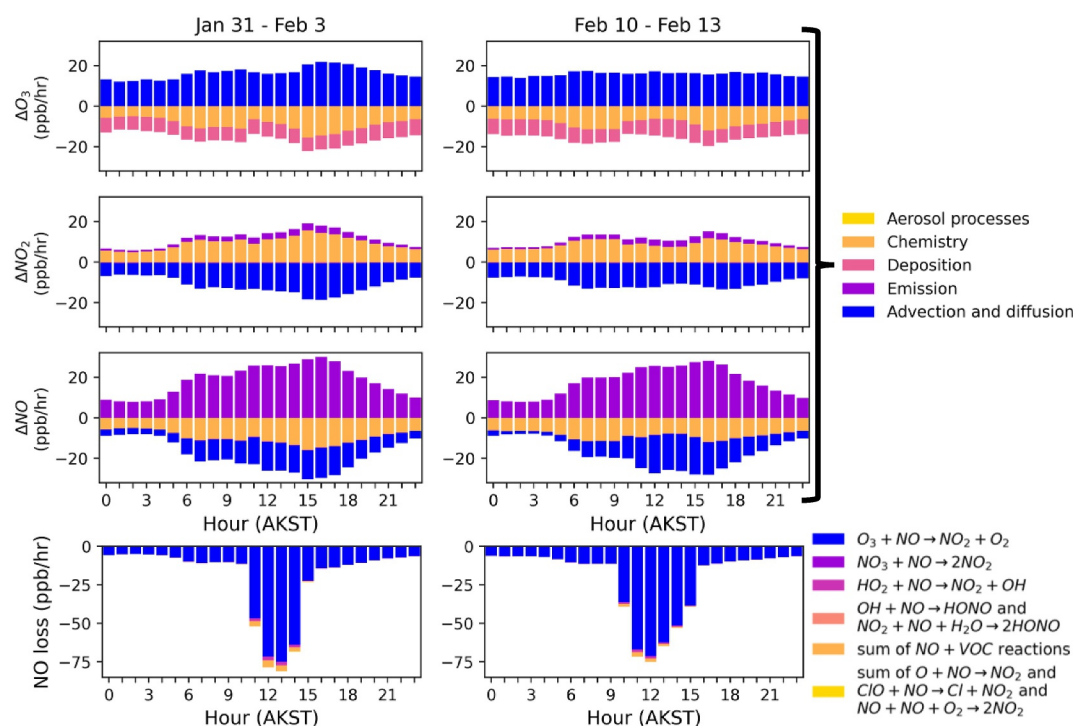


Figure 7. Model process analysis of ozone (top), NO_2 (second row), and NO (third row) for the surface layer over the entire FNSB non-attainment area for two periods. The first period (left side) is from 31 January to 3 February, during which the strong surface-based inversion event occurred resulting in high pollution. The second period (right side) is from 10 February to 13 February, which is a period of mixed atmospheric stability. Average hourly NO loss from all reactions in the cb6r5 mechanism for each period is plotted along the bottom row. Loss is plotted as a negative along the y-axis.

relative impact of different processes affecting the pollutant concentrations in Fairbanks and how these processes may be influenced by the changing BCons.

4.1. Gas-Phase Chemistry in Downtown Fairbanks

Figure 7 shows the hourly average contributions of emissions, chemistry, total transport (sum of advection and diffusion), deposition, and aerosol processes to surface ozone, NO , and NO_2 over the entire FNSB non-attainment area (Figure 1b) for two periods with different pollution and meteorological conditions. The first period, from 31 January to 3 February, is during the strong SBI event when ozone concentrations are minimal and NO_x concentrations are elevated. The second period, from 10 to 13 February, is during the weakly stable and unstable meteorological conditions that coincided with the divergent model $\text{PM}_{2.5}$ concentrations in Figure 5.

Transport is the primary source of ozone in the FNSB non-attainment area, while ozone loss is driven by chemistry and deposition. The typical formation of ozone through chemical reactions in polluted environments is correlated with sunlight and temperature (Jacob & Winner, 2009). Because of the cold and dark conditions in Fairbanks during winter, gas-phase chemistry does not result in increased ozone. Instead ozone is lost through reactions with VOCs and NO_x (Cesler-Maloney et al., 2022). The dominance of transport as a source of ozone in Figure 7 supports our hypothesis that the advection of differing background ozone concentrations between the model simulations is the driver of the differences in ozone concentrations seen in downtown Fairbanks.

Gas-phase chemistry and emissions are the major sources of NO_2 while transport decreases NO_2 in the FNSB non-attainment zone. NO loss is also impacted by transport, but during some periods of the day gas-phase chemistry is an equal or greater driver of NO loss. NO production is driven by emissions. The contribution of chemistry to NO loss has a similar diurnal trend as the contribution of chemistry to NO_2 production: chemistry is highest during the day, and there are two periods of local maxima for NO_2 (minima for NO) around 7a.m. and 4p.m. The chemistry process in Figure 7 is the sum of all production and loss reactions affecting the gas

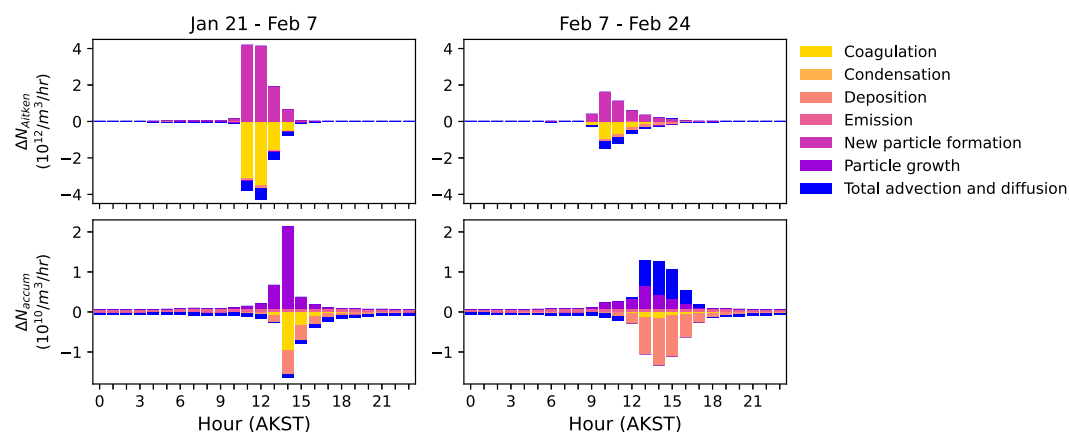


Figure 8. Stacked bar charts of budget analysis of Aitken mode (top) and accumulation mode (bottom) number concentrations in the FNSB non-attainment area. The left column is Period 1, from 21 January to 7 February, and the right column is Period 2 from 8 February to 24 February. The seven processes included are deposition, emissions, condensation, coagulation, new particle formation, particle growth, and total transport (total horizontal and vertical advection and diffusion). Note the different y-axis ranges for Aitken and accumulation mode number concentrations.

concentrations. There are NO production reactions and NO₂ loss reactions affecting concentrations, but the sum of all chemistry results in a net loss of NO and a net production of NO₂.

The diel cycle of NO loss in Figure 7 demonstrates that most of the NO lost in both periods is through ozone titration to form NO₂, hence the similar trends between chemical NO loss and chemical NO₂ production in Figure 7. Even though the total contribution and diel pattern varies during the two periods, the campaign average contribution to NO loss by this reaction is −351.5 ppb NO/day. The sum of all NO reactions with peroxy radicals derived from VOCs (labeled sum of NO + VOC reactions in Figure 7) is the second largest contributor during the day, followed by the NO + HO₂ reaction. All other reactions average a few ppt or less of NO loss per hour and less than 1.5 ppb loss per day. Although HONO and N₂O₅ heterogeneous chemistry is included in the cb6r5_aero7 chemical mechanism in CMAQv5.4 (Sarwar et al., 2008, 2012), this chemistry has minimal impact on NO_x concentrations in the FNSB non-attainment area during the ALPACA campaign as evidenced by the dominance of ozone titration driving both NO loss and NO₂ production.

The dominance of ozone titration in NO loss demonstrates how BCon inputs with differing background ozone concentrations can result in statistically significantly different NO concentrations, even in areas with high NO emissions such as downtown Fairbanks. Because chemistry is such a large driver of NO loss in the model simulations, and reactions with ozone are the greatest contributor to this chemistry, the differing ozone concentrations in the BCon inputs resulted in statistically significantly different NO concentrations. Furthermore, since differences in NO_x concentration are due to ozone titration and average model NO concentration is much lower than model NO₂, the same absolute difference from NO loss to NO₂ production is a higher fraction of NO concentration than NO₂. Thus, the change in concentration due to titration will have a greater impact on mean NO than NO₂.

4.2. PM_{2.5} Production and Loss Mechanisms

We assessed the budget of the two fine particle modes in CMAQ22, the Aitken ($d_g < 0.1 \mu\text{m}$) and accumulation ($0.1 \leq d_g < 2.5 \mu\text{m}$) modes (Binkowski & Roselle, 2003). CMAQ modes do not have fixed size distributions or a strict size cutoff at 1 μm , thus there is overlap between the different modes and the PM₁ and PM_{2.5} size designations (Binkowski & Roselle, 2003). Figure 8 shows the hourly production and loss process contributions for Aitken mode and accumulation mode particles averaged over the entire FNSB non-attainment zone in two time periods: Period 1 from the start of the simulation until 7 February, and Period 2 from 7 February until the end of the simulation. Period 1 coincides with the peak in PM₁ concentration in Figure 5, which is due to a strong SBI event which trapped heating and vehicle exhaust emissions (Simpson et al., 2024). There is good agreement between all three models for PM_{2.5}. Period 2 has mixed meteorological conditions leading to some transport of

pollution from outside the Fairbanks area (Brett, Law, et al., 2025; Simpson et al., 2024). GC22 in Figure 5 during this period shows the decoupling from the other two model simulations of $PM_{2.5}$.

New particle formation is the dominant driver of Aitken mode production in the CMAQ22 simulation, accounting for 98.7% of production in Period 1 and 96.9% in Period 2. There is minimal influence of emissions on production, only 1.32% of production in Period 1 and 2.98% in Period 2. Coagulation drives Aitken mode loss in Period 1 (76.3% of total loss) but is a smaller contributor in Period 2 (48.9%). Loss of Aitken mode particles by transport is a greater fraction of loss in Period 2 (25.7%) compared to Period 1 (18.3%). Total change in accumulation mode number concentration in Figure 8 is two orders of magnitude smaller than Aitken mode. Highest accumulation mode production occurs later in the afternoon than the peak in Aitken mode production. Production is driven by particle growth (particles in the Aitken mode grow large enough to become accumulation mode) in Period 1 (68.4% of total production) and a combination of total transport (25.5%) and particle growth (33.4%) in Period 2. Although particle growth is a large fraction of accumulation mode production, it does not represent a large fraction of Aitken mode loss as changes in accumulation mode number concentrations are two orders of magnitude smaller than Aitken mode. Accumulation mode loss is primarily driven by transport in the evenings and early morning and by dry deposition and coagulation in the afternoon.

In Period 2, transport is the dominant factor affecting accumulation mode particle number production. New particle formation is a small source of Aitken mode particles in Period 2, and particle growth has less contribution to accumulation mode production in Period 2 than Period 1. Together, this suggests that transport of existing accumulation mode particles is the driver of production in Period 2. The GC22 BCons have higher mass concentrations of several key $PM_{2.5}$ species than either CMAQ BCon (Figure S2 in Supporting Information S1), which may be transported into Fairbanks during this period. However, all three simulations overestimate $PM_{2.5}$ during Period 2, which indicates that transport into the domain is not the only factor controlling the high $PM_{2.5}$. Deposition or growth to coarse mode particles may be underestimated during this period. Although much previous research has found that daily $PM_{2.5}$ in Fairbanks is highest during long SBI events that trap local pollution (Cesler-Maloney et al., 2022; Dingilian et al., 2024; Pohorsky et al., 2025; Tran & Mölders, 2011), our work indicates that regional transport of particulate matter, for example, from sea spray aerosol or black carbon from combustion (Ioannidis et al., 2023, 2025), may be pollution sources during weakly stable periods in the winter.

5. Conclusions

We have analyzed gas and particle pollutants in Fairbanks to evaluate the impact of regional transport, as represented by sensitivity to advection from different boundary condition (BCon) datasets in the WRF-CMAQ regional model. Euclidean distances show that background concentrations affect the NAAQS values of ozone, $PM_{2.5}$, SO_2 , and NO_2 for the entire simulation period and across the entire model domain. The Z-score distances do not allow inferences of absolute differences in concentration, but show how disparate the concentration distributions are. Changing model BCons result in different concentration distributions of air quality pollutants relevant to the NAAQS attainment levels. The pollutants most affected by the regional transport from the model boundaries are ozone, NO, and $PM_{2.5}$. NO_2 is only moderately affected by the different BCons.

Analysis of processes affecting gas concentrations revealed that ozone in the FNSB non-attainment area primarily originates from transport of background air and is lost by titration with NO. The CMAQ22 simulation had the highest background ozone concentrations, which resulted in the best, though slightly underestimated, model-observation comparison during clean air events both in downtown Fairbanks (CTC site) and at the western edge of the city (UAF Farm site). All simulations overestimate ozone during inversions. NO concentration is affected by transport of ozone because the $NO + O_3$ reaction is the dominant loss mechanism of NO, while NO production is dominated by from direct emissions. However, NO concentrations are too low compared to observations because of missing or underestimated emission sources such as vehicle exhaust. If model emissions of NO were increased, the NO lost to ozone titration may represent a smaller fraction of total NO and thus advection of ozone from different BCons may no longer result in statistically significantly different NO concentrations. In addition, higher ambient NO concentrations because of increased emissions in the model would also increase the amount of ozone that could be titrated, potentially decreasing the extent of model overestimates of ozone during polluted SBI events and improving model performance.

Advection significantly impacted the transport of existing background $PM_{2.5}$ from the model boundaries. GC22 had the worst agreement with the observations during $PM_{2.5}$ transport events because of higher background $PM_{2.5}$.

concentrations. New particle formation was not a significant source of PM during transport events. When new particle formation was high, model-model agreement of PM_1 and $PM_{2.5}$ was also high. This indicates that regional transport of precursors does not have a large impact on $PM_{2.5}$ air quality in Fairbanks but transport of existing PM does.

Throughout the ALPACA campaign period, meteorology influenced pollution levels either by introduction of background air or through trapping of emissions during inversions. The WRF inputs we used in our model simulations captured the majority of the meteorological conditions. Specific instances of poorly-represented meteorology, such as the cold valley flow experienced at the UAF Farm site, may be better represented with additional measurements for nudging and analysis of these unique events. A further limitation of this study is that we downscaled hemispheric and global simulations to generate boundary and initial condition inputs for our simulations over the central Alaska domain. The coarse resolution of these simulations may have missed small-scale emission sources outside the central Alaska domain, such as black carbon or sea spray aerosol (Ioannidis et al., 2023, 2025), that that would affect our domain due to transport. A series of nested simulations (e.g., on 35 or 12 km grid resolution) would reduce the discrepancy between the global/hemispheric simulations and our 1.33 km domain and improve the ability of the boundary conditions to represent smaller, local emission sources such as windblown dust. In addition, the NHEMI 2019 CMAQ simulation contained daily rather than hourly output, which may have resulted in further underestimates of some species with strong diurnal cycles.

Our results demonstrate the importance of regional transport on wintertime air quality in high latitude locales. Although population centers in the Arctic are often far from other areas of pollution, regional transport can still have a significant impact on local air quality. Based on our results, better simulation of air quality in the Arctic on fine spatial resolution depends on BCons of the representative year, adjusting emissions for location-specific factors such as temperature, and having meteorology that captures the air exchange between the free troposphere and the polluted boundary layer. When modeling air quality for attainment purposes in such locations, it is important to consider the effects of regional transport and to choose representative boundary condition inputs.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Availability Statement

Daily hemispheric CMAQ 2019 model output from the EQUATES project is available through the CMAQ Data Warehouse (U.S. EPA, 2021). Hourly hemispheric 2022 CMAQ and 3 hourly global 2022 GEOS-Chem model output is available through the U.S. EPA's AWS repository (U.S. EPA, 2024b). All observational data from the ALPACA campaign used in this paper can be accessed from the Arcticdata repository (Barret et al., 2024; Simpson et al., 2023; Stutz et al., 2025). Daily $PM_{2.5}$ measurements made at the NCore, Hurst Road, and A Street sites can be found through the U.S. EPA's daily data download portal (U.S. EPA, 2024a). The CMAQ model source code is available for download from Zenodo (US EPA ORD, 2024).

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Acknowledgments

We thank the entire ALPACA science team for designing the experiment, acquiring funding, making measurements, and ongoing analysis of the results. The ALPACA project is organized as a part of the International Global Atmospheric Chemistry (IGAC) project under the Air Pollution in the Arctic: Climate, Environment and Societies (PACES) initiative with support from the International Arctic Science Committee (IASC), the National Science Foundation (NSF), and the National Oceanic and Atmospheric Administration (NOAA). We thank University of Alaska Fairbanks and the Geophysical Institute for logistical support, and we thank Fairbanks for welcoming and engaging with this research. We thank Barron Henderson for providing the 2022 NHEMI and global model simulations and guidance on using aqmbc. We also thank Kathleen Fahey, Rob Gilliam, and George Pouliot at the U.S. EPA Office of Research and Development for developing the emissions and WRF meteorological inputs for the ALPACA 2022 project, as well as providing modeling support. Alicia Hoffman, Steve R. Arnold, and Natalie Brett were supported by the UK Natural Environment Research Council (Grant ref. NE/W00609X/1). Natalie Brett, Kathy S. Law, Jean-Christophe Raut, Tatsuo Onishi, Brice Barret, Brice Temime-Roussel, Barbara d'Anna, and Slimane Bekki acknowledge support from the Agence Nationale de Recherche (ANR) Climate-relevant Aerosol Sources and Processes in the Arctic (CASPA) project (Grant ANR-21-CE01-0017), the Institut polaire français Paul-Émile Victor (IPEV) (Grant 1215), and the CNRS-INSU programme Les Enveloppes Fluides et l'Environnement (LEFE) ALPACA-France projects. Roman Pohorsky and Julia Schmale received funding from the Swiss Polar Institute (Technogrant 2019) and the Swiss National Science Foundation (Grant 200021_212,101). Julia Schmale holds the Ingvar Kampstad Chair for Extreme Environments Research sponsored by Ferring Pharmaceuticals. William R. Simpson and Meeta Cesler-Maloney acknowledge support from NSF Grants NNA-1927750 and AGS-2109134.

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