https://doi.org/10.5194/acp-2019-642
Preprint. Discussion started: 30 August 2019

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On the Limit to the Accuracy of Regional-Scale Air Quality Models

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Abstract. Regional-scale air pollution models are routinely being used world-wide for research, forecasting air quality, and regulatory purposes. It is well known that there are both reducible and irreducible uncertainties in the meteorology-atmospheric chemistry modeling systems. Inherent or irreducible uncertainties stem from our inability to properly characterize stochastic variations in atmospheric dynamics and from the incommensurability associated with comparisons of the volume-averaged model estimates with point measurements. Because stochastic variations in atmospheric dynamics and emissions forcing influencing the air pollutant concentrations are difficult to explicitly simulate, one can expect to find a percentile value from the distribution of measured concentrations to have much greater variability than that of the model. This paper presents an observation-based methodology to determine the expected errors from regional air quality models even when the model design, physics, chemistry, and numerical analysis techniques as well as its input data were "perfect". To this end, the short-term synoptic-scale fluctuations embedded in the daily maximum 8-hr ozone time series are separated from the longer-term forcings using a simple recursive moving average filter. The inherent variability attributable to the stochastic nature of the atmosphere is determined based on 30+ years of historical ozone time series data measured at various monitoring sites in the contiguous United States. The results reveal that the expected root mean square error at the median and 95th percentile is about 2 ppb and 5 ppb, respectively, even for "perfect" air quality models driven with "perfect" input data. Quantitative estimation of the limit to the model's accuracy will help in objectively assessing the current state-of-the-science in regional air pollution models, measuring progress in their evolution, and providing meaningful and firm targets for improvements in their accuracy relative to ambient measurements.

1 Introduction

Confidence in model estimates of pollutant distributions is established through direct comparisons of modeled concentrations with corresponding observations made at discrete locations for retrospective cases. It is well known that there are both reducible and irreducible uncertainties in the meteorology-atmospheric chemistry modeling systems. Pinder et al. (2008) discussed the reducible (i.e., structural and parametric) uncertainties that are attributable to the errors in model input data (e.g., meteorology, emissions, initial and boundary conditions) as well as our incomplete or inadequate understanding of the relevant atmospheric processes (e.g. chemical transformation, planetary boundary layer evolution, transport and dispersion, modeling domain, grid resolution, deposition, rain, clouds). Inherent or irreducible uncertainties stem from our inability to properly characterize the

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stochastic variations in atmospheric dynamics (Gilliam et al., 2015), from the incommensurability associated with comparing the volume-averaged model estimates with point measurements (e.g., McNair et al., 1996; Swall and Foley, 2009), and our inability to precisely quantify the space and time variations in atmospheric emissions and other atmospheric variables and processes. Also, without completely knowing the 3-dimensional initial physical and chemical state of the atmosphere, its future state cannot be simulated accurately (Lamb, 1984; Lamb and Hati, 1987; Lewellen and Sykes, 1989; Pielke, 1998; Gilliam et al., 2015). Given the presence of the irreducible uncertainties, precise replication of observed concentrations or their changes by the models cannot be expected (Dennis et al., 2010).

Whereas an air quality model's prediction represents some time/space-averaged concentrations, an observation at any given time at a monitoring location reflects an individual event or specific realization out of a population that will almost always differ from the model estimate even if the model and its input data were perfect (Rao et al., 1985). Consequently, comparisons of modeled and observed concentrations paired in space and time indicate biases and errors in simulating absolute levels of pollutant concentrations at individual monitoring sites (Porter et al., 2015). The scientific discussion on modeling uncertainty reduction goes back more than three decades with the current practice including data assimilation, ensemble modeling, and model performance evaluation (e.g., Fox, 1981, 1984; Lamb, 1984; Pielke, 1998; Lewellen and Sykes, 1989; Lee et al., 1997; Carmichael et al., 2008; Hogrefe et al., 2001a, 2001b; Grell and Baklanov, 2011; Gilliam et al., 2006; Baklanov et al., 2014; Bocquet et al., 2015). While ever-improving process knowledge and increasing computational power will continue to help reduce the structural and parametric uncertainties in air quality models, the inherent uncertainty cannot be eliminated because our inability to properly characterize the stochastic nature of the atmosphere will always result in some mismatch between the model results and measurements; this could lead to speculation on the inferred accuracy of the future states simulated by the regional air quality models (Porter et al., 2015; Astitha et al., 2017; Luo et al., 2019).

In most applications of regional-scale air quality models, statistical metrics such as bias, root mean square error (RMSE), correlation, index of agreement are being used to judge the quality of model predictions and determine if the model is suitable for forecasting or regulatory purposes (e.g., Fox, 1981, 1984; Solazzo et al., 2011; Appel et al., 2012; Simon et al., 2012; Foley et al., 2014; Ryan et al., 2016; Emory et al. 2016; Zhang, 2016; U.S. EPA, 2018). While significant improvements in the formulation, physical and chemical parameterizations, and numerical techniques have been implemented in atmospheric models over the past three-decades, it is not clear if the improvement claimed in the model's performance relative to the routine network measurements is statistically significant based on these metrics (Hogrefe et al., 2008). Also, no assessments have been made to date on the errors to be expected in regional-scale air quality models. To this end, we analyzed the daily maximum 8-hr (DM8HR) ozone data at monitoring locations across the contiguous United States (CONUS) during the 1981-2014 time period along with the 21-year fully coupled WRF-CMAQ simulations covering the 1990-2010 period as detailed below.

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2 Data and Methods

Ground-level DM8HR ozone data covering the CONUS during May to September in each year were obtained from the U.S. Environmental Protection Agency's (EPA) Air Quality System (AQS) (see https://github.com/USEPA/CMAQ/tree/5.0.2). A valid ozone season consists of at least 80% data coverage during May to September at each station. A total 185 monitoring stations with at least 30 valid years (to provide enough variety of synoptic conditions, denoted hereafter as 30+ in this paper) from the year 1981 to 2014 are analyzed. Also, fully coupled WRF-CMAQ model simulations over the CONUS for the 1990-2010 period were utilized in this study to demonstrate a new perspective on model performance evaluation. Time-varying chemical lateral boundary conditions are nested from the 108 km hemispheric WRF-CMAQ simulation from 1990 to 2010 (Xing et al., 2015). Evaluation of the 21-year long WRF-CMAQ simulation using 36-km grid can be found in Gan et al. (2015).

It has been shown that time series of the daily maximum 8-hr ozone concentrations contain fluctuations operating on different time scales, reflecting the short-term forcing induced by the passage of weather systems across the country and long-term forcing induced by emissions, El-Nino-Southern Oscillation (ENSO), climate change, and other slow-varying processes such as seasonal and sub-seasonal variations in the atmospheric deposition and stratosphere-troposphere exchange processes (Rao et al., 1996, 1997; Vukovich, 1997; Hogrefe et al., 2000; Porter et al., 2015; Astitha et al., 2017). Variations in ambient ozone can be thought of comprising of the baseline of pollution that is created by various emitting sources and modulated by the prevailing synoptic weather conditions (Rao et al., 2011). Thus, the level of the baseline (BL) concentration and the strength of the synoptic component (SY) should be viewed as the necessary and sufficient conditions for how high ozone levels can reach on a given day (Astitha et al., 2017). Scale separation can be achieved by applying filtering methods such as the Empirical Mode Decomposition (Huang et al., 1998), Elliptic filter (Poularika, 1998), Kolomogorov-Zurbenko (KZ) filter (Rao and Zurbenko, 1994), Adaptive Filter Technique (Zurbenko, et al., 1996), and Wavelet (Lau and Weng, 1995). Because Empirical Mode Decomposition and KZ filter yielded similar results for the DM8HR time series data, only the results from the KZ filter are presented here. Further, the KZ filtering is a simple method and works well in the presence of missing data (Hogrefe et al., 2003). In this study, we used the KZ(5,5) with a window size of 5 days and 5 iterations in the same manner as in Porter et al. (2015), Rao et al. (2011), and Luo et al. (2019). The size of the window and the number of iterations determine the desired scale separation. The KZ(5,5) filtering process helps separate the synoptic-scale weather-induced variations embedded in the May-September DM8HR time series data (short-term component, noted as SY) from the long-term baseline component (noted as BL).

$$BL(t) = KZ(5,5) \quad (1)$$
$$SY(t) = O_3(t) - KZ(5,5) \quad (2)$$
$$O_3(t) = SY(t) + BL(t) \quad (3)$$

where O3(t) is the original time series of the observed DM8HR ozone concentration, BL(t) is the baseline component and SY(t) is the synoptic component at any given time. Because we are working with daily maximum 8-hr ozone data, the Nyquist interval is 2-days, indicating that the dynamical features having time scales less than 2 days (e.g., intra-day forcing from fast

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changing emissions and chemical transformations, boundary layer evolution, diurnal forcing due to night vs. day differences) are not resolvable in this analysis (see Fig. 2 in Dennis et al., 2010). The 50% cut-off frequency for the KZ(5,5) is ~24 days, and, hence, time scales less than those associated with synoptic-scale weather fluctuations are embedded in the short-term or SY forcing. The KZ filtering is applied to both DM8HR observations and modeled DM8HR time series. Once the baseline is separated from the original DM8HR time series from all monitoring stations, then the synoptic forcing in the historical ozone time series data is used to estimate the variability in ozone concentrations that can be expected because of the chaotic/stochastic nature of the atmosphere by taking into account the relationship between the strength of synoptic forcing and mean of baseline ozone at each location over CONUS. This methodology was applied to both measured and modeled ozone concentrations (see details in Luo et al., 2019). Whereas the objective of Luo et al. (2019) was on transforming the deterministic modeling results into a probabilistic framework for assessing the efficacy of different emission control strategies in achieving compliance with the ozone standard, this paper is aimed at quantifying the errors to be expected at each monitoring site over CONUS even from "perfect" regional ozone models driven with "perfect" input data from the ever-present stochastic nature of the atmosphere.

3 Results and Discussion

3.1 Analysis of ambient ozone data

To illustrate the concept of the ozone baseline, DM8HR time series measured in 2010 at Altoona, PA is presented in Fig. 1a together with the embedded baseline concentration as extracted by the KZ(5,5) filter. It is evident that high ozone levels are always associated with the elevated baseline. The difference between the raw ozone time series and baseline, denoted as the short-term or synoptic forcing (SY), is displayed in Fig. 1b along with time series of white noise. By superimposing AR(1) process on the ozone baseline, Rao et al. (1996) demonstrated that the number of observed ozone exceedances above a given threshold at a monitoring site can be reproduced. A comparison between the SY component and white noise process, presented in Fig. 1b, reveals that the SY component having finite variance and zero mean resembles near-stochastic process. Hence, the baseline concentration is to be viewed as the deterministic part and SY is considered the stochastic component in the ambient ozone time series.

Once the scale separation is achieved with the KZ(5,5), we superimposed the SY forcing imbedded in 30+ years of historical DM8HR ozone time series measured at a given location on the baseline component of the ozone time series at that location to generate 30+ reconstructed or pseudo ozone distributions. Illustrative results at a suburban location in Altoona, PA are presented for 2010 base year in Fig. 2a; it should be noted that the linear relationship between the strength of SY and the magnitude of the BL has been taken into account in generating 30+ years of adjusted SY forcing as illustrated in Luo et al. (2019). As expected, there is excellent agreement between the average of 30+ values (solid blue line) and observed ozone in 2010 at each percentile of the concentration distribution function (red line). Also, the original cumulative distribution function (CDF) in 2010 (red line) is constrained within the 30+ CDFs of pseudo-observations (Fig. 2a); note, it is equally likely for any

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of the 30+ CDFs to occur due to the stochastic nature of the atmosphere even though the individual event in 2010 yielded the CDF shown in red. As mentioned before, ozone mixing ratio at any given probability point on the red line in Fig. 2a reflects a specific event while ozone values at the same probability in different CDFs (light blue lines) reflect the population stemming from the chaotic nature of the atmosphere. In other words, there are 30+ dynamically consistent ozone time series attributable to the 2010 emissions loading for examining the inherent variability. It is evident in Fig. 2a that there is larger variability at the lower and upper percentiles than that in inter-quartile range, revealing that the tails of the concentration distribution function are subject to large inherent uncertainty. Using these 30+ pseudo-observation ozone mixing ratios and the actual observed ozone values at each percentile, statistical metrics such as Bias, RMSE, coefficient of variation (CV=standard deviation/mean), normalized mean error (NME) and normalized mean bias (NMB) are presented in Fig. 2b and c. As expected, the lower and upper tails of the distribution are prone to large errors. These results demonstrate the presence of larger natural variability at the upper 95th percentile, which is of primary interest in regulatory analyses.

Ozone time series at 185 monitoring stations covering CONUS, having at least 80% data completeness, are analyzed in the above manner and the results are displayed as box plots in Fig. 3. Note the presence of large variability in the CV, NME, and NMB, and Bias at lower and upper percentiles (Fig. 3). The RMSE expected for the ozone mixing ratios in the interquartile range is ~1.5 ppb, but it is >5 ppb for the upper 95th percentile (Fig. 3b). The spatial distribution of RMSE at the 50th and 95th percentiles is displayed in Figures 4a and 4b, respectively. The RMSE at the upper 95th percentile is very high at some monitoring sites in California and Michigan (Fig. 4b). Monitoring stations at high elevations, residing well above the nocturnal boundary layer, tend to exhibit lower variability than those situated in the urban areas, near large water bodies, and complex terrain due to the dominance of local conditions.

3.2 Analysis of modeled ozone concentrations

The analysis in the previous section quantified the inherent stochastic variability represented by the SY component using long-term records of ozone observations. In this section, we analyze long-term records of model simulations in an attempt to quantify the error associated with not explicitly representing stochastic variations in atmospheric dynamics and emission variability in the current generation regional air quality models. The model simulations were performed with the fully coupled WRF-CMAQ system with a 36-km horizontal grid cell size and covered the 21-year period from 1990 to 2010 (Gan et al., 2015). To provide an illustration of the differences between observed and modeled time series over this period, Figure 5a displays a scatter plot of the strength of the SY component vs. the mean of the baseline (BL) component for both observations and model simulations at the Altoona, PA site. While both observations and WRF-CMAQ simulations show a strong correlation between these two variables, it is evident that at this monitoring location the standard deviation (i.e., strength) of the SY component is substantially lower for the WRF-CMAQ simulations for a given mean of the BL component (i.e., for any given year). The year-to-year variation in the observed and modeled mean of BL and strength of SY forcing, displayed in Fig. 5b, reveals that the model overestimated BL and underestimated the strength of SY forcing. The 36-km grid may be better reproducing the large-scale

https://doi.org/10.5194/acp-2019-642
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synoptic forcing associated with the translation of weather systems than the meso-scale weather and urban influences that are embedded in the observed SY component. Meteorological modeling with higher horizontal grid resolution might be able to capture the land-sea breeze, lake-sea breeze, and terrain influences that observations are seeing at certain monitoring locations.

An understanding of the expected error even when the model's physics, chemistry, numerical solver, and the input data are "perfect" would help model developers in making decisions on model improvements. To this end, we assume that the model perfectly reproduces the 'true' BL depicted by the observed BL. We then use this 'perfect' modeled BL and reconstruct 'pseudo-simulated' ozone time series, similar to what was done in Fig. 2, except for using the SY component from the 21 years of coupled WRF-CMAQ simulations. Fig. 6a shows the CDF of actual observed ozone (red line) overlaid on 21 pseudo-simulated ozone CDFs (blue lines) at the Altoona, PA site while Figs. 6b and 6c display absolute and normalized performance metrics. Figure 6a confirms that the coupled WRF-CMAQ SY components have less intra-annual (sub-seasonal) variability than observed SY components, causing an overestimation at the low end and an underestimation at the high end of the observed CDF for all 21 years of reconstruction; these results imply that the model's results at the upper and lower percentiles will always tend to be unreliable or prone to large errors even when the baseline concentration is predicted perfectly. The U-shape of the absolute and relative error curves in Figures 6b and c is similar to the corresponding curves in Figure 2, but the larger magnitude at the high and low end of the distribution indicates that the effects of the underestimated intra-annual (sub-seasonal) SY variability (note that the distribution of modeled values in Fig. 6a is much flatter (i.e., having higher Kurtosis) than that of the observations) outweigh those errors attributable to the stochastic variability presented in Figure 2. The shape of the absolute and normalized bias curves deviates from those shown for the pseudo-observations in Figures 2b-c and, thus, also reveals the effect of the underestimation of the sub-seasonal SY variability. Figures 6d-f present differences between the curves shown in Figures 6a-c and a version of Figure 2a-c computed from 1990-2010 rather than 30+ years of observations. Panels e and f show that at the 50th percentile, the differences in the error curves are close to zero due to the fact that both the pseudo-simulations and pseudo-observations used the same observed BL component. At the upper percentiles, the differences reach 3 – 5 ppb, providing an estimate of the reducible error in simulating extreme values at this location because of the differences in the observed and WRF-CMAQ SY components at this location; high-resolution meteorological modeling may help address these reducible errors.

Figs. 7a and 7b show the RMSE at the median and 95th percentile for the 'pseudo-simulated' ozone values at each monitoring site. For the 50th percentile, the RMSE values range from 0.2 ppb to 3.2 ppb over CONUS with a median value of 1 ppb while at the 95th percentile, the RMSE values range from 1 ppb to 14.9 ppb with a median value of 3.8 ppb across all sites over CONUS. The values are highest along the California coast and near Great Lakes, possibly due to errors in boundary conditions and inadequacies in simulating the land-sea breeze and land-lake breeze regimes, respectively, with modeling at 36 km grids. As model improvements are made, one can quantitatively assess how close the predictions of the improved model are to the

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expected or target RMSE at each monitoring site for each percentile for the given base year simulation (see Fig.4a and b for expected errors from "perfect" models with "perfect" input at the median and 95th percentile).

4 Conclusions

Weather is a stochastic process that impacts the prediction of air pollutants, and regardless of how accurate the regional air quality model is, this stochastic component cannot be consistently reproduced. In this study, we demonstrate how to quantify this irreproducible stochastic component by isolating the synoptic forcing imbedded in 30+ years of historical observations and assess the performance of the 36 km fully coupled WRF-CMAQ model in simulating 21 years of ozone concentrations over CONUS. Observation-based analysis reveals that on average, the irreducible error attributable to the stochastic nature of the atmosphere ranges from ~2 ppb at the 50th percentile to ~ 5 ppb at the 95th percentile. To improve regional-scale ozone air quality models, attention should be paid to accurately simulate the baseline concentration by focusing on the quality of the emission inventory and the model's treatment for the slow-changing atmospheric processes. Also, errors in reproducing the sub-seasonal variability can possibly be reduced with high-resolution meteorological modeling. Nonetheless, these results demonstrate the presence of large variability in the upper tail of the DM8HR O_3 concentration cumulative distribution even with "perfect" models using "perfect" input data. Having this quantitative estimation of practical limits for model's accuracy helps in objectively assessing the current state of regional-scale air quality models, measuring progress in their evolution, and providing meaningful and firm targets for improvements in their accuracy relative to measurements from routine networks.

Code availability: Source code for version 5.0.2 of the Community Multiscale Air Quality (CMAQ) modeling system can be downloaded from https://github.com/USEPA/CMAQ/tree/5.0.2. For further information, please visit the U.S. Environmental Protection Agency website for the CMAQ system: https://www.epa.gov/cmaq.

Data availability: All ozone observations used in this article are available from https://aqs.epa.gov/aqsweb/airdata/download_files.html (AQS). Paired ozone observation and CMAQ model data used in the analysis will be made available at https://edg.epa.gov/metadata/catalog/main/home.page. Raw CMAQ model outputs are available on request from the U.S EPA authors.

Competing interests: The authors declare that they have no conflict of interest.

Author Contribution: STR conceptualized the idea. STR, CH, VG, and RM designed the analysis approach. CH and RM post-processed previously conducted model simulations. HL performed data analyses and prepared the illustrations. STR prepared the manuscript with contributions from all co-authors.

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Disclaimer: The views expressed in this paper are those of the authors and do not necessarily represent the view or policies of the U.S. Environmental Protection Agency.

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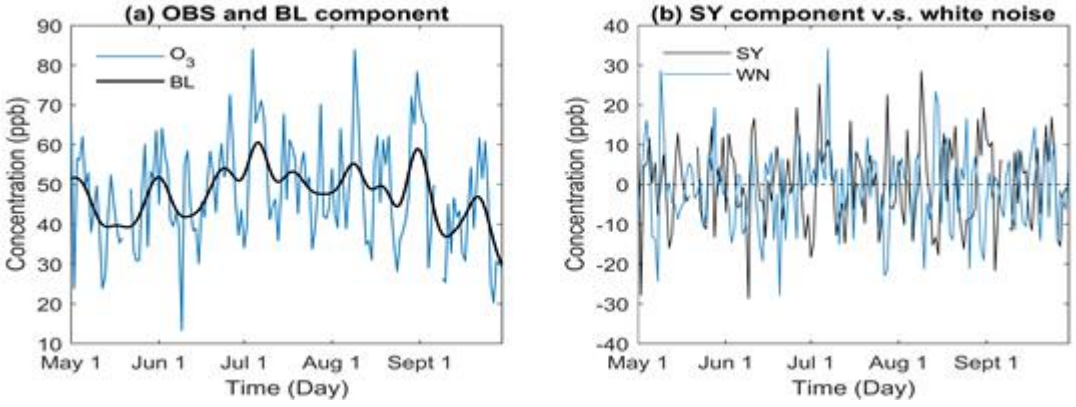

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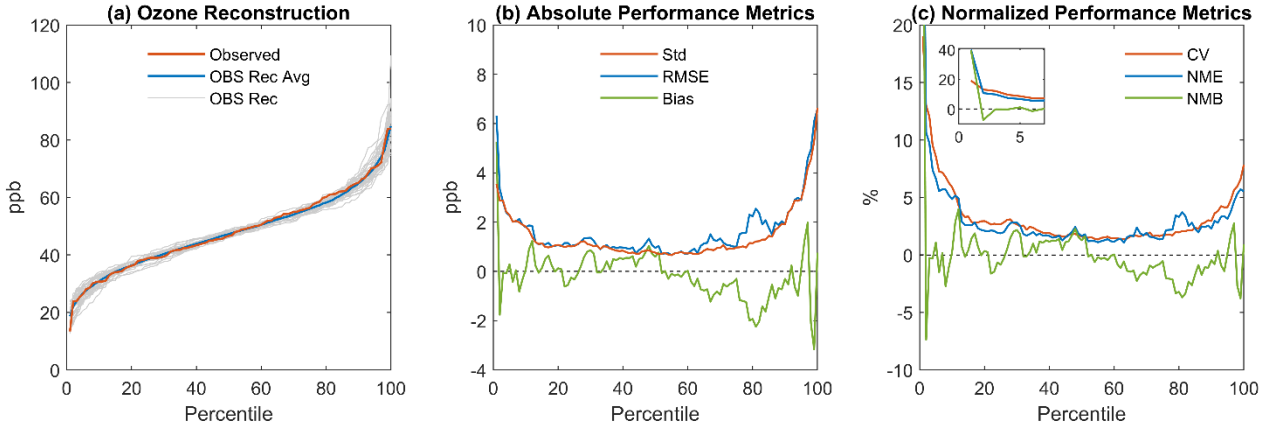

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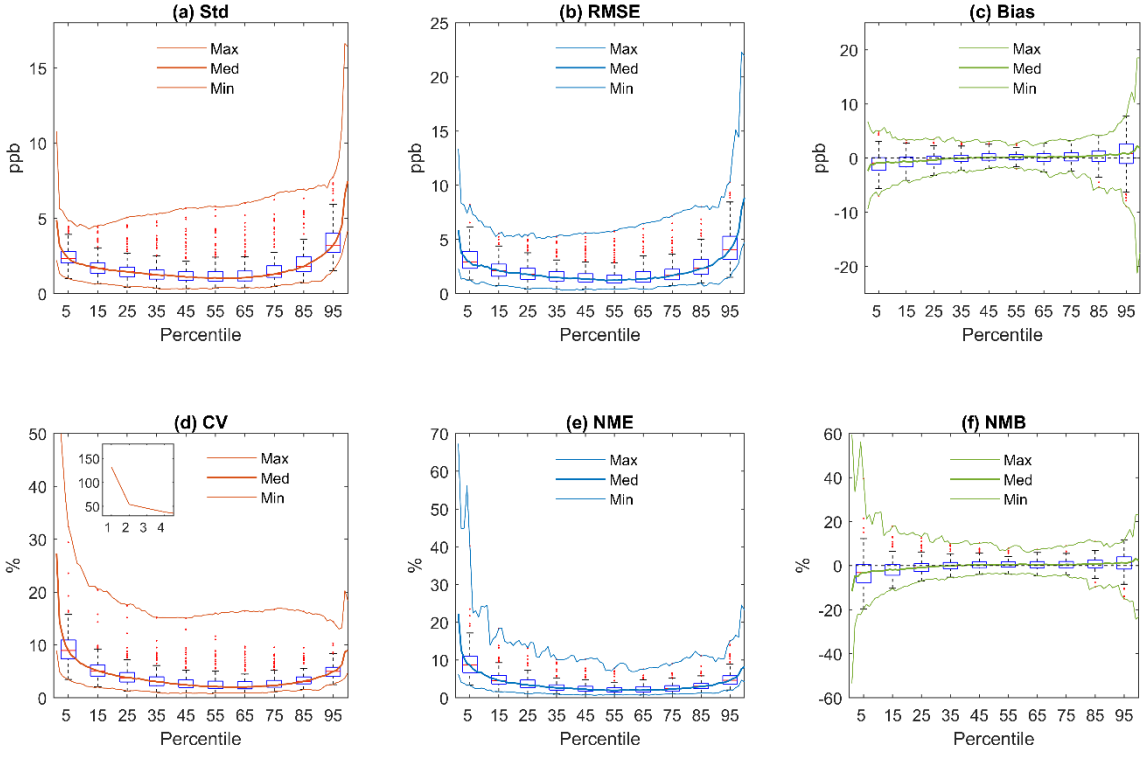

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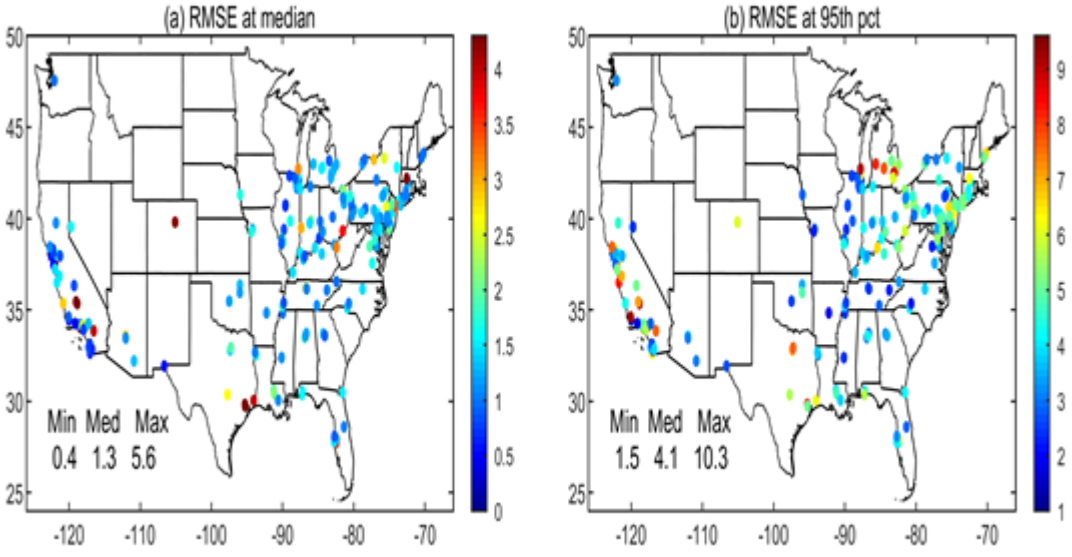

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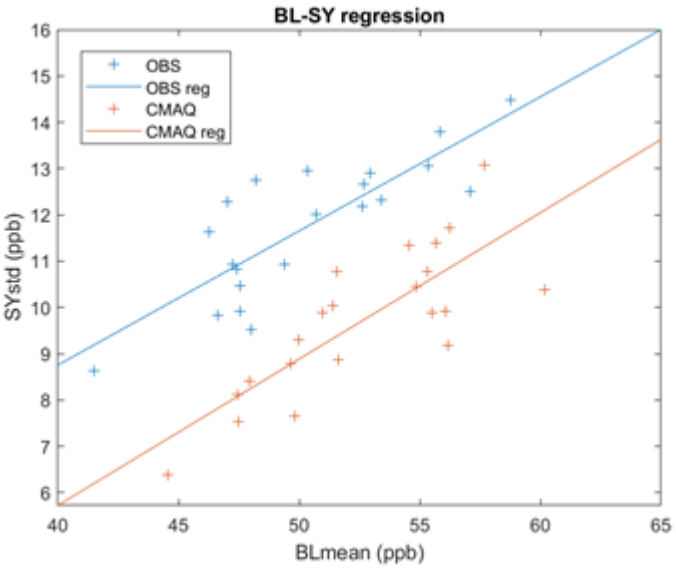

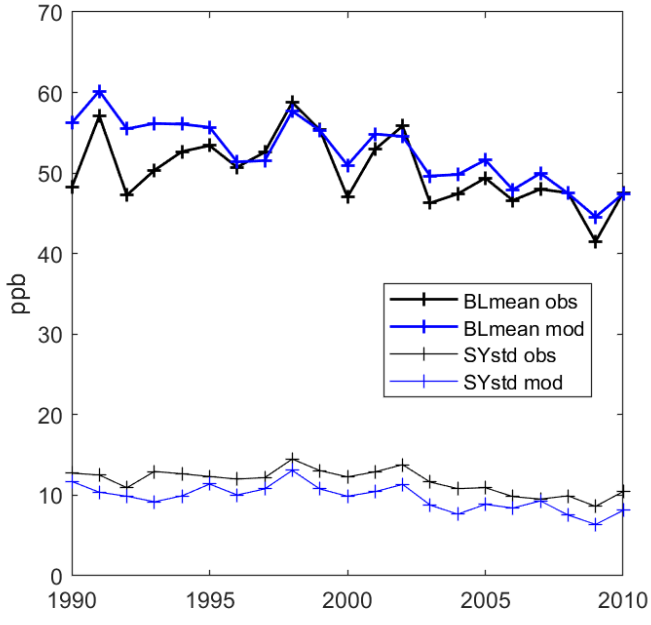

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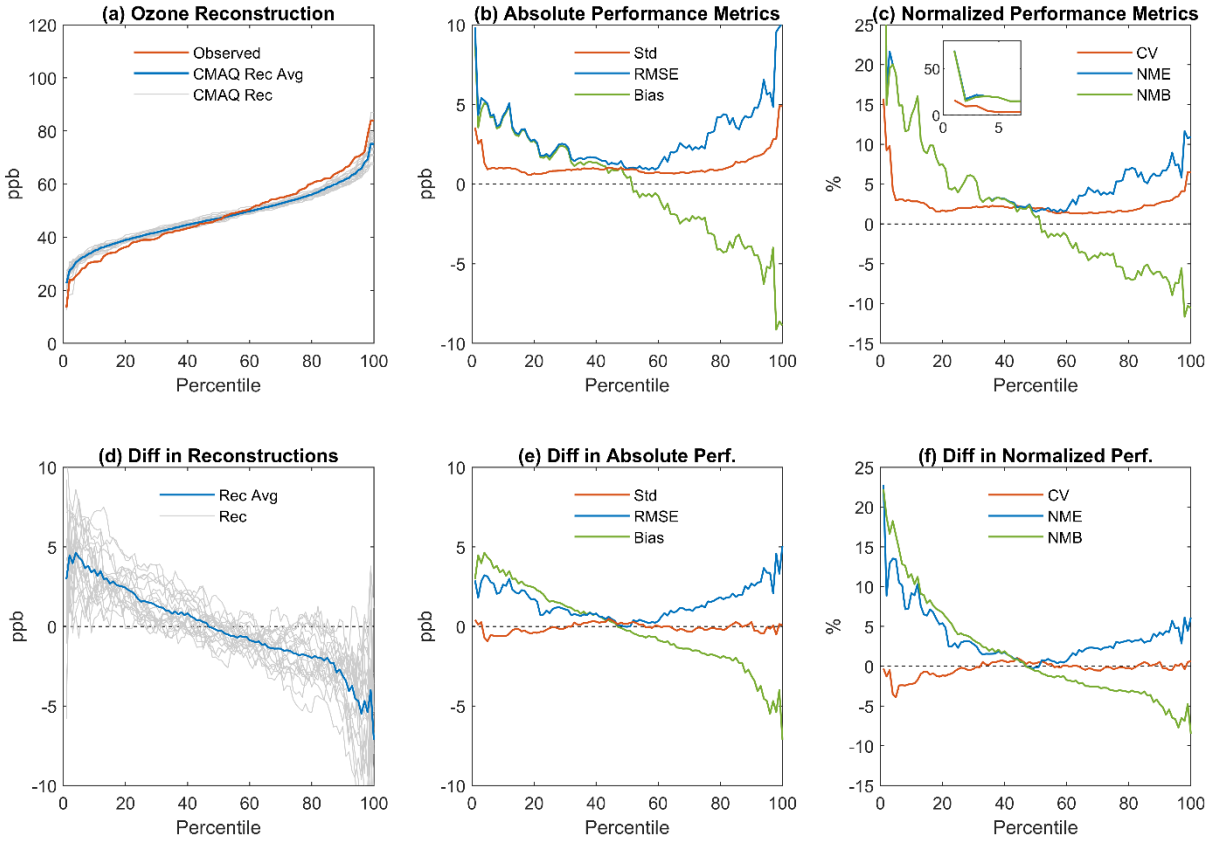

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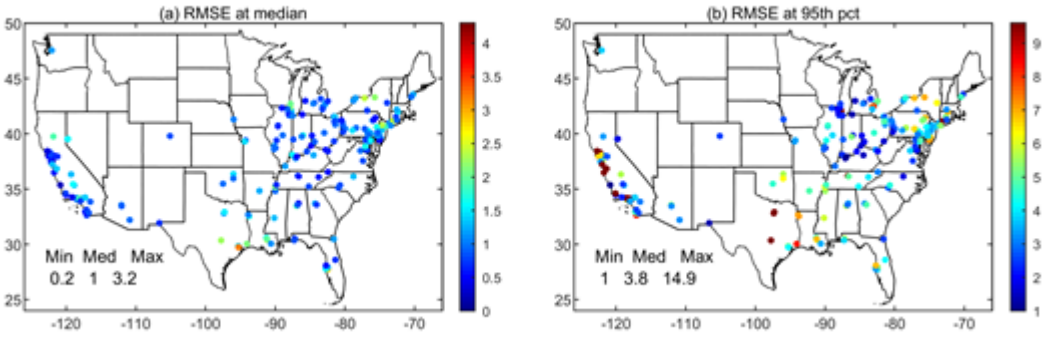

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